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Contaminants of emerging concern in a large temperate estuary

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Abstract

This study was designed to assess the occurrence and concentrations of a broad range of contaminants of emerging concern (CECs) from three local estuaries within a large estuarine ecosystem. In addition to effluent from two wastewater treatment plants (WWTP), we sampled water and whole-body juvenile Chinook salmon (*Oncorhynchus tshawytscha*) and Pacific staghorn sculpin (*Leptocottus armatus*) in estuaries receiving effluent. We analyzed these matrices for 150 compounds, which included pharmaceuticals, personal care products (PPCPs), and several industrial compounds. Collectively, we detected 81 analytes in effluent, 25 analytes in estuary water, and 42 analytes in fish tissue. A number of compounds, including sertraline, triclosan, estrone, fluoxetine, metformin, and nonylphenol were detected in water and tissue at concentrations that may cause adverse effects in fish. Interestingly, 29 CEC analytes were detected in effluent and fish tissue, but not in estuarine waters, indicating a high potential for bioaccumulation for these compounds. Although concentrations of most detected analytes were present at relatively low concentrations, our analysis revealed that overall CEC inputs to each estuary amount to several kilograms of these compounds per day. This study is unique because we report on CEC concentrations in estuarine waters and whole-body fish, which are both uncommon in the literature. A noteworthy and unexpected finding was the preferential bioaccumulation of CECs in free-ranging juvenile Chinook salmon relative to staghorn sculpin, a benthic species with relatively high site fidelity.

Graphical abstract

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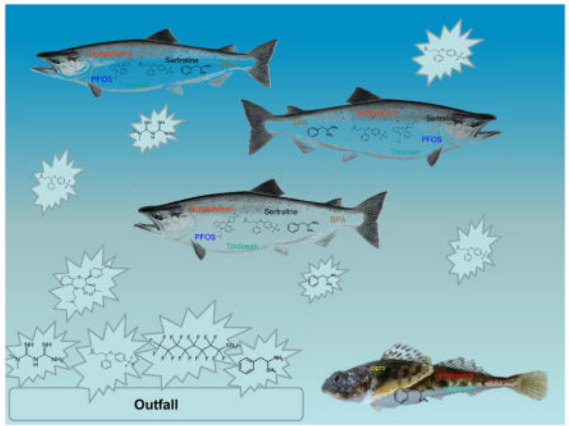

Keywords

Estuary; fish; wastewater effluent; pharmaceuticals; personal care products

1. Introduction

Contaminants of emerging concern (CECs) constitute a wide range of chemicals for which there is limited data on occurrence, environmental fate, and toxicity. Represented in this class of environmental contaminants are pharmaceutical and personal care products (PPCPs) and a number of industrial compounds such as polybrominated diphenyl ethers (PBDEs), perfluorinated compounds (PFCs), alkylphenols, bisphenol A, phthalates, and current-use pesticides. Many of these compounds are present in our rivers, estuaries, and coastal areas from wastewater treatment plant (WWTP) effluent discharging via outfalls to these water bodies. Other sources of CECs to waterways include discharges from industrial sources and aquaculture operations, in addition to runoff from impervious surfaces, landfills, biosolids application, and agricultural and farming activities (Gaw et al., 2014).

Most of these CECs are potent human and animal medicines that are used for various purposes, many of which are then excreted as the parent compound or as metabolites that flow into WWTPs. Some of these compounds are eliminated or reduced in concentration by treatment practices that vary among facilities or are sorbed to biosolids and removed from the waste stream (Lubliner et al., 2010; Oulton et al., 2012). By contrast, some CECs are poorly removed by WWTP processing or are discharged to surface waters, including streams, estuaries, or open marine waters due to secondary bypass or combined sewer overflows, (Lubliner et al., 2010; Phillips et al., 2012).

There are several important factors to consider in assessing the environmental risk of CECs in estuarine waters, as well as other aquatic habitats. These include; the extent of product usage among local human populations, physical-chemical parameters (i.e. water solubility, hydrolysis, photodegradation, and adsorption to sediment and biosolids), rates of bioaccumulation, chemical potency, and potential toxicity to aquatic organisms and aquatic-dependent wildlife. Among these aforementioned factors, bioaccumulation and comparative toxicity to aquatic species constitutes the largest data gap in assessing ecological risk.

Over 4,000 approved drug products are currently available (U.S. Food and Drug Administration, 2015) under various formulations and approximately 1100 are unique prescription and over-the-counter compounds comprising a large number of chemical classes and mechanisms of action (MoA). A consensus value of 324 drug targets has been proposed by Overington et al. (2006) for all classes of therapeutic drugs. A recent study of 12 fish species from a variety of families concluded that 65 – 86% of human drug targets are conserved in diverse fish species (Brown et al., 2014); therefore it is reasonable to assume that many of these drugs will also affect fish. Of the hundreds of chemicals that are likely present in the Puget Sound ecosystem, only a small percentage are currently monitored or regulated and there is little or no toxicity information for the vast majority of these compounds. Many of these are common household chemicals that pass through wastewater treatment, have been approved for use and/or consumption by the general public, and are generally considered to be non-toxic. However, the higher-than-expected levels for some of these chemicals in aquatic organisms and possibly aquatic-dependent wildlife along with critical gaps in toxicological and risk assessment data underscores their importance for further investigation in the context of environmental and public health concerns (Roos et al., 2012; Arnold et al., 2014).

Relatively comprehensive analyses of CECs in the marine or estuarine ecosystem within the United States are uncommon. Notable exceptions for U.S. waters include the analysis of CECs in effluent and marine waters in southern California (Vidal-Dorsch et al., 2012) and Charleston Harbor (Hedgespeth et al., 2012), receiving waters in four estuaries along the Texas coast (Scott et al., 2015), San Francisco Bay (Klosterhaus et al., 2013), and Lubliner et al., (2010) who reported on effluent concentrations from WWTPs in Puget Sound, Washington. As far as we know, there are no studies that tested for a large suite of CECs in whole-body fish in marine waters.

Our approach in the present study involved a review of the literature that resulted in a prioritized list of 102 PPCPs, 17 hormones, and 31 industrial compounds to serve as a representative subset of CECs that we identified as a potential concern in the estuarine waters of Puget Sound, Washington, USA. Our primary goal was to determine the occurrence and concentrations of CECs in WWTP effluent, estuary water, and two fish species occupying different habitats with different life histories and compare among locations and matrices.

2. Methods

2.1. Selection of field sites

We selected three local estuaries as focal points for our study, including two estuaries that receive effluent from WWTPs and one as a reference site that is not known to have direct inputs from WWTP effluent. One contaminated site was Sinclair Inlet, which receives effluent from the Bremerton Westside WWTP (Figure 1). The effluent outfall is located approximately 170 meters from shore at a depth of 10 meters below mean lower low water (MLLW) in Sinclair Inlet. Sinclair Inlet has one other known source of effluent from the South Kitsap Water Reclamation Facility with a design flow of 16 million liters/d (MLD) (South Kitsap Water Reclamation Facility 2013). The other contaminated site selected was the Puyallup River estuary, which receives effluent from the Tacoma Central WWTP (Figure 1). The discharge outfall is at 40 meters MLLW and approximately 370 meters northwest from the mouth of the Blair Waterway in Commencement Bay. The Puyallup River basin contains 8 additional WWTPs with a combined permitted effluent volume of 63 MLD with flows generally running much lower (Pierce County 2010). The Nisqually estuary was selected as a minimally-contaminated reference site, and has been used in numerous studies as a reference site (Meador 2014). Table 1 contains additional details for each site. Two fish species that commonly occur in Puget Sound estuaries were selected for assessing bioaccumulation of CECs. Specifically, Pacific staghorn sculpin (*Leptocottus armatus*) was selected for biomonitoring because this species found widely in Puget Sound and U.S. west coast temperate waters, generally exhibits high site fidelity, and may reside in estuaries for extended periods (Tasto, 1976). Juvenile Chinook salmon (*Oncorhynchus tshawytscha*) were selected based on their residence time (up to several weeks) in local estuaries where contaminants are often concentrated (Healey 1991). Chinook salmon were selected over other salmonids that do not exhibit this life history trait (Meador 2014). We also collected hatchery-reared juvenile Chinook salmon from the Voight's Creek hatchery on the Puyallup River for comparison to fish collected in the estuary. Fish were collected under a Washington State Scientific Collection Permit 13—046 and ESA Section 10(a)(1)(A) permit 17798. All methods for obtaining, transporting, and tissue sampling of fish were approved by the University of Washington Institutional Animal Care and Use Committee (protocol number 4096-01). Details of all sampling methods used in this study are reported in Yeh et al., (2013).

2.2. Sampling for CEC analytes in WWTP effluents and water

The effluent from Bremerton West WWTP was sampled on 9 September 2014 and the effluent flow was 13.2 MLD. The maximum monthly design flow from October – April is stated to be 58.7 MLD and permitted at 86 MLD (Bremerton Westside Factsheet, 2013). The effluent from Tacoma Central WWTP, Tacoma, WA was collected on 17 September 2014 and the flow on that day was 56.8 MLD. The maximum month design flow for wet weather is listed as 143.8 MLD (Tacoma Central WWTP Factsheet, 2004) and the permitted capacity is 228 MLD (Pierce County, 2010). These values do not include secondary treatment bypass during high volume flows or peak flows, which may exceed average flows by 2-fold. For the two week period prior to sampling, Tacoma experienced 2.03 inches of rain and Bremerton received 2.89 inches of rain (Weatherunderground, 2015).

At each WWTP, a total of 11 one-liter amber glass bottles were filled with effluent sampled at the final stage of processing, just before discharge into the outfall leading to the estuary. Similarly, at each field site a total of 11 one-liter amber glass bottles were filled with estuarine water at a depth of 2 m below the surface with a swing-sampling pole designed to collect water below the surface. We generally followed Washington Department of Ecology (2006) for obtaining water samples. Estuary water quality parameters including dissolved oxygen, conductivity, salinity, and temperature of the water column were measured at a depth of 2 m below the surface using the YSI Model 85 handheld probe (YSI Incorporated, Yellow Springs, OH). Similarly, the pH of the water column was measured using the Eutech Multi-Parameter PCSTestr 35 (Oakton Instruments, Vernon Hills, IL). One water sample was taken at each site and the estuary parameters were measured within minutes of water collection. No field blanks were collected.

2.3. Fish sampling

Juvenile Chinook salmon were obtained at each field site with a beach seine and were categorized as wild or hatchery origin based on the presence of an adipose fin. Artificially reared salmon are marked by removal of the adipose fin by each hatchery. Staghorn sculpin were also obtained by beach seining; however 4 – 5 individuals from the Puyallup estuary were obtained by shrimp traps set at 8 – 9 m below the surface. Each species was collected as close as possible to the outfall area (Figure 1), which in most cases was several hundred meters away. Fish were kept alive after collection in the field and transported to the laboratory for processing. Fish were transported in site water that was aerated and temperature was maintained at 11 °C with ice packs. Samples were taken approximately 3 – 6 hours after capture and whole bodies of all fish were frozen at −80 °C after processing.

Fish were euthanized with tricaine methanesulfonate (MS-222; Argent Chemical Laboratories, Redmond, WA) for processing. To avoid analysis of stomach contents that were considered external to the fish, the entire alimentary canal and stomach contents of all fish analyzed for chemistry were cleaned of material by rinsing with distilled water. The contents were discarded and the cleaned tissue included with the whole bodies for analysis. Chemical analyses for CEC analytes were conducted on composite samples consisting of 3 – 12 whole-body salmon or 3 – 5 whole-body sculpin.

Juvenile Chinook salmon from the nearby Gorst Creek rearing ponds that empty directly into the head of Sinclair Inlet (far west end) were released unusually early in the year (Mike Huff, hatchery manager, personal communication) and were probably out of the area at the time of sampling. As a result, the juvenile salmon sampled were likely from outside the area, as noted in previous studies of this local estuary (Fresh et al., 2006), but were nonetheless exposed to WWTP effluent while residing in Sinclair Inlet. All collected fish were scanned for the presence of coded wire tags (CWTs) by personnel from the U.S. Fish and Wildlife Service (USFWS). Heads of fish with detected CWTs were removed and read by USFWS personnel. Only four CWTs were found in Chinook salmon obtained from Sinclair Inlet and all were from nearby Grover’s Creek Hatchery. Two CWTs were detected in Chinook salmon obtained from Puyallup estuary and both were from the White River Hatchery. Two CWTs were also detected in Chinook salmon from the Nisqually estuary, which indicated the Kalama Creek and Clear Creek Hatcheries as the source.

2.4. Analytical methods

Concentrations of CEC analytes were determined by AXYS Analytical, Ltd. (Sidney, British Columbia, Canada) using LC/MS/MS techniques. Table S1 gives a complete list of the 150 CEC analytes with their analytical methods and reporting limits (RLs). Of the 150 analytes, 147 were analyzed in water samples and 122 were analyzed in fish tissue. Based on the low RL values obtained in these samples, the analytical methods employed were generally highly sensitive for most compounds and represent the state-of-the-art approaches in quantitating this diverse group of compounds in environmental media.

All analytes were measured in water and tissue, except hormones, hexabromocyclododecanes (HBCDDs), and phthalate esters. Hormones were only determined in water because many of these compounds occur naturally in tissue and the available phthalate ester method was developed for water. Because phthalates are difficult to quantify in various matrices due to high control and analytical blank values, we opted to analyze ester metabolites of these compounds, which are less problematic. HBCDDs were analyzed in tissue only. Two of the compounds (bisphenol A and triclosan) were determined by two different analytical methods, once as part of a general analytical method and again by a compound-specific method (Table S1). No corrections were applied to the analytical values (e.g. percent recovery). All sampling objectives and quality control parameters outlined in Yeh et al. (2013) were achieved in this study. Many of the quality assurance and quality control parameters for the chemical analyses can be found in U.S. Environmental Protection Agency (2007), which have improved post-publication of this document.

3. Results

Of the 150 targeted analytes for this study, 92 (61%) were detected in effluent, estuarine water, or fish and only 58 (39%) were not detected in any of these matrices (Tables 2, S2, and S3). Additional information and data highlighting chemical output rates from effluent, physical-chemical properties, known half-lives, available partition coefficients, undetected compounds, and reporting limits can be found in Appendix A (Tables S1 – S4). Site and fish data are listed in Table 1. The available data for partitioning as determined by the bioconcentration factor (BCF) and organic-carbon normalized sediment-water partition coefficient (K_{oc}) and listed in Table S4. Most values in this table are estimated based on various schemes, many of which are based on water solubility and an octanol-water partition coefficient (K_{ow}) dependent regression. These approximations likely underestimate actual values for ionizable organic compounds.

3.1. Occurrence and concentrations of CECs in WWTP effluents

We detected 81 analytes in WWTP effluent (Table S4) representing 55% of the total analyzed. Several of these analytes (15) were detected at concentrations greater than 1,000 ng/L (low ppb range) and 8 of those analytes were detected in estuarine water. A few compounds were observed in estuarine waters but not effluent, including sulfadimethoxine, sulfamethoxazole, testosterone, and mono-*n*-butyl phthalate, the latter a metabolite of dibutyl phthalate. In general, the detection frequency and concentrations were similar for a given type of media (e.g. effluent or estuary water) among impacted sites, although there were several notable differences (Table S4). For effluent, 77 analytes were detected in the Tacoma effluent, with 15 being unique for this type of matrix and location. The Bremerton WWTP effluent contained 66 detected compounds, with 4 (PFOS, PFBS, PFHxS, and androstenedione) being unique to this effluent. Several of the 15 analytes detected in Tacoma effluent and not the Bremerton effluent were observed at elevated concentrations (>20 ng/L). For the 62 compounds detected in both WWTP effluents there was no clear pattern of dominance with respect to concentration. However, comparing between the Bremerton and Tacoma effluent we found substantially higher concentrations in the Bremerton effluent compared to the Tacoma effluent for DEET (684 v. 23 ng/L), caffeine (1,170 v. 152 ng/L), BPA (350 v. 4,290 ng/L), and estrone (58 v. 4.5 ng/L) (Table S4), which may indicate regional differences in usage.

3.2. Occurrence and concentrations of CECs in estuary waters

In the present study, we detected 25 CEC analytes in estuarine waters (Table S4). The estuary samples from both Sinclair Inlet and the Puyallup estuary contained 16 – 17 analytes with 5 or 6 analytes unique to each estuary. The Nisqually reference site contained 10 detectable analytes, including comparatively high concentrations of 4-nonlyphenol (4-NP), and monobutyl phthalate (Table S4). All analytes detected in effluent were considered as a source to estuarine waters in terms of mass per day. Based on both the effluent flow rate at the time of collection and measured concentrations, the total amount of detected analytes flowing into their respective estuarine waters ranged from 0.8 and 6.6 kg/d for the Bremerton Westside and Tacoma Central WWTPs (Table 2). During "maximum design flows" occurring October – April, CEC inputs from these WWTP could be substantially higher at 3.5 and 16.8 kg/d, which is based on flow data obtained from Bremerton Westside Factsheet (2013) and Tacoma Central WWTP Factsheet (2004). These values would not account for episodic releases of influent during peak flows that bypass secondary treatment. Based on the data presented in Lubliner et al. (2010), influent concentrations can be 1 – 2 orders of magnitude higher than effluent concentrations for many PPCPs.

3.3. CECs in sculpin and salmon tissues

A number of compounds were found in fish and not in effluent or estuary water. These include PFDA, PFOSA, enalapril, benztropine, fluocinonide, sulfadaizine, sulfamerazine, virginiamycin M1, and ormetoprim (Table 2). Interestingly, ormetoprim is widely used in hatcheries to treat fish under the trade name Romet™, and likely was in some hatchery fish at the time of release. Sulfadimethoxine is also a component of Romet™ and was found only in salmon; however it was detected in effluent and estuary water, and therefore it is not known if tissue levels were due to estuarine or hatchery exposure. The compounds HBCDD (not analyzed in water), PFDA, and PFSOA have been detected in fish or WWTP effluent in Puget Sound or its watershed (Washington Department of Ecology, 2010; Johnson and Friese, 2009) and are likely from industrial sources in the area. Conversely, even though phthalate ester metabolites were not analyzed for tissue samples they likely occurred in whole-body fish because of their relatively high K_{ow} and elevated concentrations in estuary water. Sulfadiazine has been reported in effluent by Verlicchi et al. (2012). The source of the remaining compounds sulfamerazine, fluocinonide, and virginiamycin is unclear. Virginiamycin is an antibiotic approved for large animal use and may occur in estuaries from runoff. A review of the literature did not reveal any studies reporting detectable concentrations for these compounds in either effluent or fish tissue.

Collectively, we detected 42 compounds in whole-body fish (Tables 2 and S5). CECs in juvenile Chinook salmon were detected at greater frequency and higher concentrations compared to staghorn sculpin. Figure 2 shows the concentrations of detected analytes in fish, estuary water, and effluent sorted by occurrence from high to low concentrations in salmon tissue. In general, juvenile Chinook salmon from the Puyallup estuary contained a greater frequency of detected analytes (25) and higher concentrations (most > 1 ng/g) than that observed for Chinook collected in Sinclair Inlet. Notable compounds occurring at comparatively high concentrations in juvenile Chinook from the Puyallup estuary include amphetamine, azithromycin, diltiazam, diphenhydramine, fluoxetine, gemfibrozil, miconazole, norfluoxetine, sertraline, sulfadimethoxine, triclosan, triclocarban, virginiamycin, and nonylphenol and its metabolites. Chinook collected in Sinclair inlet contained 19 detected analytes and most were lower in concentrations compared to Puyallup Chinook with some exceptions (e.g., PFOS, caffeine, and fluocinonide). Nisqually Chinook salmon contained 13 detected analytes; however most exhibited low concentrations, except for nonylphenol. Chinook salmon from both effluent sites contained several CECs at concentrations substantially higher than those observed for Nisqually Chinook. We detected 7 analytes in juvenile Chinook collected from the Voight's Creek Hatchery. Two of these analytes (benztropine and enalapril) were found only in these fish, which have been detected in WWTP effluent or lake water in other studies (Verlicchi et al., 2012; Ferrey, 2013). Three of the other detected compounds (BPA, nonlyphenol, and DEET) were unusually elevated in these fish. Values for nonylphenol and bisphenol A in fish tissue were as high or higher than levels found in estuary fish and may have come from leaky septic systems in the area or other discharge upstream.

Among sculpin, concentrations of detected analytes were relatively similar between the 2 effluent sites, both in terms of chemical concentrations and frequency of occurrence. Sculpin from the Nisqually estuary reference site contained 9 detected analytes, including several at comparatively high concentration. The predominant analytes in sculpin harvested from the Nisqually included nonylphenol, caffeine, ciprofloxacin, and metformin. However, based on water concentrations, sculpin from the effluent sites were exposed to higher numbers and concentrations of contaminants than those collected in the Nisqually estuary, many of which were likely not bioaccumulated to levels above the analytical detection limit.

Based on the relatively rapid half-life for several of the compounds tested for (Table S2) and the lag time between capturing fish in the field and sacrifice in the lab (3 – 6 h), many of the analytes examined in this study may have been higher, some substantially, in feral fish. Therefore the reported concentrations may underestimate, sometimes by a large margin, the concentrations accumulated by fish in these estuaries or even fall below detection after capture if elimination is particularly rapid.

4. Discussion

The greater Puget Sound area contains 106 publicly owned WWTPs that discharge at an average total flow about 1,347 MLD (Washington Department of Ecology, 2010). Our study examined 2 of these with a combined total of 71 MLD. The output for these 2 WWTPs alone was on the order of kg quantities of detected CECs per day into estuarine waters of Puget Sound. Considering the low percentage of commercially available PPCPs analyzed in this study and the amount of effluent discharged to Puget Sound waters, it is apparent that a substantial load of potentially harmful chemicals are introduced into streams and nearshore marine waters daily. If the concentrations from the 2 studied effluents are representative of that from other WWTPs in Puget Sound, then it is reasonable to assume that inputs to streams and nearshore waters are substantial and likely on the order of 100 kg/d (≈ 36,500 kg annually) and even higher if secondary treatment bypass, permitted flows, maximum outputs, unmeasured compounds, septic system contributions, and transboundary contributions are considered.

Based on our water and fish data, the Nisqually estuary was more contaminated than expected, which highlights the difficulties of establishing suitable non-polluted reference sites for these ubiquitously distributed CECs (Ferguson et al., 2013). It is noteworthy that for all 3 estuaries investigated in the present study, a few analytes (e.g., cocaine, ciprofloxacin, and ranitidine) were found only in estuary water at our reference site, even though compounds were present in effluent from the contaminated sites. Although the source of these compounds to the Nisqually estuary is unknown, the Nisqually River, Nisqually Reach, and McAllister Creek are all included on the 303(d) list of water bodies that do not meet water quality standards for fecal coliform bacteria, which may be caused by leaking septic systems (Washington Department of Ecology, 2007; Washington Department of Ecology, 2015). Even though a number of analytes were surprisingly elevated in water and tissue (e.g., nonylphenol, diphenhydramine, ciprofloxacin, DEET, and metformin), overall the frequency of occurrence and concentrations of these contaminants in the Nisqually estuary were generally low relative to the effluent-impacted sites. While it is unknown if these chemicals alone or in combination are sufficiently elevated to result in adverse effects, we are conducting other studies that link exposure to CECs with adverse physiological outcomes in sculpin and salmon.

4.1. CECs in water and fish tissue

Compared to other marine studies, our results for effluent were generally similar to those reported by Vidal-Dorsch et al. (2012) and Hedgespeth et al. (2012) for the few overlapping analytes. As for surface waters, our values for the few analytes in common for each study were generally greater than the reported values in Vidal-Dorsch et al. (2012), but lower than the values observed in Charleston Harbor, San Francisco Bay, and the Gulf of Mexico estuaries (Hedgespeth et al., 2012; Klosterhaus et al., 2013; Scott et al., 2015). For the 16 effluent CECs in common between Lubliner et al. (2010) and the present study for Puget Sound, most of the analytes reported in the present study were observed at higher concentrations, which could be a result of increased rates of usage for these CECs or differences in the treatment processes among plants.

As discussed, our results indicate a large number of analytes in effluent were below their respective limits of analytical detection in estuarine waters. These chemicals may have been present at extremely low levels in water and fish but could not be quantified; however, this does not imply the absence of potential toxic effects as noted by Schlenk et al. (2012) for mixtures of CECs. It is noteworthy that our estuarine water samples were collected several hundred meters from the effluent outfalls and at a depth of only 2 meters, thus reported concentrations likely underestimate those occurring in deeper water and closer to outfalls. The effluent plume is expected to move horizontally with currents before substantial mixing occurs (Environment Canada, 2003).

To better understand the characteristics of our WWTP effluents relative to those in other locations, we compared our effluent concentrations to those reported by Kostich et al. (2014)

for the 50 largest WWTPs in the U.S., none of which discharged to marine waters or were located in the Pacific Northwest. The Kostich et al. (2014) data for 53 pharmaceuticals and 7 metabolites were summarized statistically and compared to our measured values in the two effluents. As observed in Table 2, the results of our comparison to the Kostich et al. (2014) data overlapped on 45 compounds. The CEC analyte concentrations observed in our study were generally higher than most values for a given compound measured in the 50 WWTP effluents, which is reflected in the percentile ranking of our values to those presented in Kostich et al. (2014). Our concentrations were greater than the 90th percentile for values from all 50 WWTPs (most >99th percentile) for 34 of those 45 analytes. For 10 of the common analytes, all 50 effluent values in that study were below their reporting limits, but were detected in our study (OH-amitriptyline, atorvastatin, benztropine, lincomycin, paroxetine, promethazine, simvastatin, sulfadimethoxine, testosterone, and warfarin). Conversely, we report non-detectable concentrations for acetaminophen, sulfamethazine, and theophylline in effluent whereas detectable concentrations were reported in the Kostich et al. (2014) dataset. While our observed concentrations were among the highest reported for effluent in the United States, higher concentrations have been reported in secondary effluent in other countries (Verlicchi et al., 2012).

The concentrations obtained in our one-time sampling event for each estuary are likely representative of samples taken for other time points throughout the year and not expected to exhibit substantial temporal variability. One study on CECs in the marine environment examined temporal variability of effluent and receiving water concentrations and observed little difference among the 4 seasons for the 56 analytes examined (Vidal-Dorsch et al., 2012). Some seasonality was observed by Hedgespeth et al. (2012) in their study of 19 CEC compounds in effluent and surface water, who noted higher frequencies of detection in winter compared to summer, a similar phenomenon observed by Daneshvar et al. (2010). Higher frequency of detection and greater concentrations during winter months are likely due to colder temperatures inhibiting bacterial metabolism and reduced photolysis (Vieno et al., 2005; Daneshvar et al., 2010; Hedgespeth et al., 2012), which may offset any dilution due to potential stormwater inputs. Additionally, for some PPCPs there is likely a seasonal component for usage rates by consumers. For example, some chemicals such as antihistamines may be more prevalent during spring and summer months, whereas others such as DEET, are expected to be lower during winter.

Despite the widespread occurrence of CECs and importance whole-body tissue concentration in risk assessment and regulatory frameworks (Sappington et al. 2011), we found no comprehensive studies reporting on whole-body tissue concentrations for these compounds in field-collected fish. Clearly, this is an important data gap in assessing the environmental risk of CECs. Choosing one representative tissue for assessing toxic effects and bioaccumulation is generally more problematic than analyzing whole bodies. Whole-body concentrations are likely a better surrogate for toxic dose and bioaccumulation compared to individual organ concentrations due to greater comparability among species toxicity metrics and bioaccumulation factors and because of the inherent variability for target-organ specificity and lipid content, in addition to confounding effects and seasonal differences (Meador et al., 2008). Many studies provide data on organ-specific concentrations, which are generally higher than reported for whole-body concentrations.

Ramirez et al. (2009) examined PPCPs in fish tissue from 5 effluent-dominated streams and one recent review (Daughton and Brooks, 2011) summarized the known data for wild fish. The report of Ramirez et al. (2009) and the present study have 5 analytes in common that were detected in fish tissue (norfluoxetine, sertraline, diphenhydramine, diltiazem, and triclosan). Ramirez et al. (2009) detected carbamazepine in tissue, whereas we detected this compound only in effluent and estuary water. The 2 studies are not directly comparable because Ramirez et al. (2009) reported concentrations for fish fillets and liver. Another interesting comparison is the San Francisco Bay data for co-located water and mussel tissue concentrations (Klosterhaus et al., 2013). Even though most of their estuarine water concentrations were higher than our values, our fish tissue concentrations were higher, sometimes substantially, compared to mussel tissue, with notable exceptions for carbamazepine, DEET, and NP2EO.

4.2. CEC physicochemical characteristics and bioaccumulation

Compounds with $\log_{10}$Kow value > 2 were more likely to bioaccumulate in fish; however, compounds with relatively short half-lives (less than 24 h) would not be expected to appreciably bioaccumulate due to elevated rates of clearance and/or metabolism. Unfortunately, a review of the literature revealed few values for chemical half-lives in fish. For most compounds with elimination data for both humans and fish, the reported half-life for humans was much shorter than that observed for fish (Table S2). It should be noted that human half-life values are for plasma and they may be representative of whole-body half-life only if the compounds moved freely among tissues and were not sequestered or stored in other tissues. Therefore human plasma half-lives are likely not directly comparable to whole-body half-lives for fish, but may be a reasonable estimation for relative persistence in tissue.

4.2.1. Bioaccumulation of CECs in sculpin and salmon—As discussed by Daughton and Brooks (2011), pharmaceuticals are generally more polar and less hydrophobic than most environmental contaminants considered in risk assessments and therefore do not preferentially associate with sediment or tissue. While these compounds remain mostly dissolved in water, they can be bioaccumulated by organisms through ventilation, ingested water, and prey and therefore may interact with receptor targets resulting in pharmacological effects if concentrations are high enough. Even though predicted bioaccumulation and bioconcentration factors estimated with K_{ow} values are relatively low, it is well known that many ionic compounds do not bioaccumulate according to these predicted values (Meador 2000; Fu et al., 2009; Daughton and Brooks 2011). One study that measured plasma bioconcentration factors in fish found large variation among sites that was not attributed to aqueous concentration, pH, exposure time, or temperature (Brown et al., 2007), indicating the difficulty of predicting tissue concentrations.

Of the 69 PPCPs detected in water or fish in the present study, 70% are ionizable organic compounds. Bioaccumulation of polar and ionizable compounds is generally not predictable with the current target lipid model (Di Toro et al., 2000) that is premised solely on hydrophobic partitioning to organismal lipid. Instead of passive diffusion across membranes that can be easily modeled, predictions of bioaccumulation for many CECs demand an

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evaluation based on toxicokinetics, passive diffusion, and active transport, which can vary widely among species (Daughton and Brooks 2011; Meredith-Williams et al., 2012). Active transport is likely an important mechanism to consider because a large number of drugs are known to be taken up across biological membranes by one of several known transporters (Dobson and Kell 2008).

Various estimates for the percentages of commercially available drugs that are ionizable range from 63 – 95% (Manallack 2007) indicating this as an important factor for determining bioaccumulation, toxicity, and environmental fate. Specifically, organic compounds with pKa values several units above or below the pH of seawater (pH ≈ 8 – 8.1) are expected to be ionic and may not readily accumulate in fish, unless there is active transport across gill or gastrointestinal membranes. Wells (1988) estimated that 75% of pharmaceuticals are weak bases, indicating that pKa is a crucial factor for assessing bioaccumulation and toxicity in marine waters especially when pH – pKa > −3 to 1 (Rendal et al., 2011).

A number of compounds in Table S2 have relatively high $\log_{10}K_{ow}$ values (>3) and pKa values similar to seawater (pH approx. 8.0), indicating a high potential for bioaccumulation in aquatic organisms. It is not known if these high K_{ow} compounds would exhibit even higher bioaccumulation as a result of active transport over that predicted based on thermodynamics (e.g., the target lipid model). In the present study, most of the compounds that were detected in fish are characterized by high K_{ow} values (Table S2), with the exception of amphetamine, caffeine, ciprofloxacin, DEET, ranitidine, and sulfadimethoxine. It is unknown if pK_a would play a role in bioaccumulation for these low K_{ow} compounds. It should be noted that BCF values may be a poor estimator of bioaccumulation for some of these compounds in the field. For example, the steady-state BCF values for caffeine, carbamazepine, and diphenhydramine determined in the laboratory for mosquito fish (*Gambusia holbrooki*) were 2, 1.4, and 16, respectively, whereas the BCF values for this species naturally exposed to these compounds in a pond were 29, 108, and 821 (15 – 77× greater), indicating that dietary exposure is likely important for bioaccumulation (Wang and Gardinali, 2012).

As noted by Rendal et al., (2011), organic bases such as fluoxetine, norfluoxetine, propanolol, lidocaine, sertraline, and trimipramine, exhibit increasing toxicity for algae and fish with rising pH, with large differences between pH 6.5 and 8.5. As shown for fluoxetine ($pK_a = 9.8$) each unit increase in pH from 7 – 9 caused both the $\log_{10}K_{ow}$ and levels of unionized fluoxetine to increase 10-fold (Nakamura et al., 2008). These data indicate a much greater potential for bioaccumulation in aquatic environments with greater than neutral pH, such as marine systems. This was confirmed by Nakamura et al. (2008) who showed a substantial increase in the fluoxetine BCF for fish (30-fold) in addition to a 28-fold decrease in the LC50 (more toxic) as pH increased from 7 to 9.

Even though observed and predicted BCFs for many CECs are relatively low (e.g., 3 – 10, Table S4), salmon and sculpin collected in the present study contained higher than expected concentrations when based on analytes detected in estuary water. These higher than predicted tissue concentrations could be due to additional sources, such as upriver inputs or foodweb magnification. One study demonstrated large differences in bioconcentration factors among invertebrates exposed to a number of pharmaceuticals with species varying 10 – 100 fold (Meredith-Williams et al., 2012). Notably, these authors reported a BCF of 185,900 for fluoxetine in the amphipod (*Gammarus pulex*), which may contribute to higher than expected fish tissue concentrations. Such differences are often due to variable uptake and elimination kinetics among species, similar to those described for invertebrates exposed to tributyltin, which is both polar and ionizable (Meador 1997). The unexpectedly large differences in tissue concentrations for juvenile Chinook salmon and staghorn sculpin in this study are unknown; however such differences noted above for invertebrate prey, in addition to variability in ventilation and ingestion rates between fish species, potential metabolic differences, and degree of mobility may explain the disparity. As noted in Meador (2014), Chinook salmon can exhibit high rates of ingestion and gill ventilation.

4.4. Classes of compounds

Noteworthy groups of compounds are highlighted due to the high frequency of occurrence and potential to cause adverse effects in fish.

4.4.1. Pharmaceuticals

4.4.1.1. Hormones: Many pharmaceuticals are considered endocrine disrupting (ED) compounds affecting reproductive function (Diamanti-Kandarakis et al., 2009). Hormones are the most potent EDs affecting fish at low ng/L concentrations and several were detected in effluent or estuarine waters (androstenedione, estrone, and testosterone). Estrone (E1) was elevated in the Bremerton effluent and the measured value (58 ng/L) is in the 85th percentile of all measured effluent values from U.S. WWTPs as summarized by Kostich et al. (2013). Dammann et al. (2011) reported increased levels of vitellogenin, altered secondary sexual characteristics, and enhanced aggression in male fish exposed at aqueous concentrations of estrone ranging from 15 – 54 ng/L, exhibiting a similar potency as 17-α-ethinylestradiol (EE2). Dietary uptake may be a substantial source of these compounds for fish species. The predicted E1 BCF for fish is 54 (Table S4); however *Daphnia magna* exhibited a BCF for E1 of 228 (Gomes et al., 2004), which may be representative of bioaccumulation in other invertebrates and could lead to enhanced tissue concentrations in fish.

4.4.1.2. Antibiotics: In our study, 16 antibiotic compounds were detected in water and fish tissue. Excess antibiotics in the water may affect the natural composition of bacteria externally and internally in fish (Daughton and Brooks 2011, Carlson et al., 2015). Possible effects include the suppression of beneficial bacteria and enhancement of pathogenic bacterial resistance to antibiotics. A number of authors have raised the possibility that continuous discharge of antibiotics into surface waters may increase the occurrence of antibiotic resistant strains of bacteria (Kristiansson et al., 2011, Berglund 2015). Several macrolide antibiotics (-mycins, Table S2), were detected and they summed to approximately 500 – 980 ng/L in effluent, 5 ng/L in estuarine water, and 13 – 34 ng/g in whole-body fish. Because a number of these antibiotics work by the same MoA (e.g., macrolide antibiotics at a specific site on subunit 50S of the bacterial ribosome), their effect concentration may be considered together through dose addition (Meador 2006).

4.4.1.3. Central nervous system agents: A large number (25) of detected compounds in this study are used to modulate neurological function in humans. These include serotonin selective re-uptake inhibitors (SSRIs) in addition to central nervous system stimulants, narcotics, and analgesics. These compounds have been widely prescribed to treat anxiety, epilepsy, and hypertension in humans (Table S2). Many of these chemicals may also affect behavioral function in fish and invertebrates, even at the relatively low concentrations found in contaminated receiving waters (Painter et al., 2009; Brooks, 2014). Surprisingly, algal growth was very sensitive to fluoxetine (Brooks et al., 2003).

Two of the antidepressants, sertraline and fluoxetine, are especially noteworthy because these were observed in juvenile Chinook (Table 2) at concentrations higher than those reported by Brooks et al. (2005) for 3 species of fish from an effluent-dominated stream. A number of studies have examined effects of sertraline and fluoxetine in fish and report a large range in aqueous concentrations causing adverse effects. For example, Schultz et al. (2011) reported increased mortality and histological alterations in the testes for sertraline and fluoxetine and increased vitellogenin production in male fathead minnows exposed to very low concentrations (1.6 – 5.2 ng/L of sertraline and 28 ng/L for fluoxetine), which are substantially lower than effluent concentrations reported in the present study. In Schultz et al. (2011), reported concentrations of these compounds in brain tissue were very low (0.17 ng/g for fluoxetine and 0.02 – 0.06 ng/g for sertraline). While the concentrations of these SSRIs were below detection limits in estuarine water in the present study, our whole body concentrations for these compounds were elevated (5 – 17 ng/g) for juvenile Chinook salmon. Because neural tissue preferentially accumulates sertraline and fluoxetine and exhibits concentrations that are higher than other tissues (Brooks et al., 2005; Schultz et al., 2010), whole-body concentrations are likely lower than that expected for brain tissue, suggesting that brain tissue of juvenile Chinook salmon in our study contained very high levels of these antidepressants. Additionally, the metabolite norfluoxetine binds the serotonin reuptake transporter with a similar affinity as fluoxetine and is considered as potent as the parent compound. Based on these characteristics, it is reasonable to sum the concentrations of these compounds to determine the potential for adverse effects for this MoA (Daughton and Brooks, 2011).

4.4.1.4. Metabolic regulators: A number of compounds that target metabolic abnormalities (e.g. metabolic regulators) such as diabetes, elevated lipids, and hyperglycemia were observed in effluent, estuarine water, and fish tissue. These include atorvastatin, gemfibrozil, glipizide, glyburide, metformin, and simvastatin and they have the potential to act as metabolic disruptors affecting growth, lipid homeostasis, and energy balance in nontarget organisms when introduced to the environment (Casals-Casas and Desvergne, 2011). Other chemicals that are known metabolic disruptors were also detected at high concentrations in the present study, including bisphenol A, nonylphenols, phthalates, and perfluorinated compounds.

Metformin, a medicine to treat diabetes, was the analyte detected at the highest concentration in effluent (29,300 – 82,700 ng/L) with very high concentrations in estuary water (up to 832 ng/L). The high metformin concentration observed in sculpin from the Nisqually estuary (27.8 ng/g) was surprising given the very low K_{ow} for this compound. A recent study (Niemuth and Klaper 2015) demonstrated reduced growth in male fathead minnow (*Pimephales promelas*) and extensive disruption of reproductive parameters in both sexes of this species exposed to metformin at 40,000 ng/L. Another recent study demonstrated significant increases in mRNA transcripts for vitellogenin, estrogen receptor-alpha, gonadotropin releasing hormone 3, and cytochrome P450 3A4-like isoform in juvenile fathead minnow exposed to concentrations in water as low as 1 ng/mL (Crago et al., 2016).

4.4.2. Personal care products—Triclosan and triclocarban were detected in effluent and salmon tissue. Only triclosan was detected in estuary water (Sinclair Inlet) and was present at 5.2 ng/L, which would theoretically result in a fish tissue concentration of 0.47 ng/g, given the observed fish BCF of 90 for this compound. A high concentration was observed in salmon tissue from the Puyallup estuary (mean = 24.4 ng/g), which may be due to foodweb magnification from algae and invertebrate species that exhibit relatively high bioaccumulation factors (500 – 1,000) (Hontela and Habibi, 2014). High tissue concentrations are also expected in higher trophic level species such as marine mammals (Hontela and Habibi, 2014). Given the observed BCF for triclosan, our reported tissue concentration in salmon would be equivalent to a water exposure concentration of 271 ng/L. Triclosan is weakly estrogenic in fish (Hontela and Habibi, 2014), but has been shown to significantly increase aggressive behavior in fathead minnows when exposed to a mixture of triclosan (560 ng/L) and triclocarban (179 ng/L) (Schultz et al., 2012). These aforementioned concentrations are only about 2-fold higher than the modeled exposure concentration (271 ng/L) expected to result in the observed salmon tissue concentration.

4.4.3. Industrial chemicals—Nonylphenol (NP) was one of the more ubiquitous compounds in our study and was observed in every sample (except Sinclair Inlet estuary water) at relatively high concentrations in water (14 – 41 ng/L) and tissue 8 – 76 ng/g). The ethoxylates of nonylphenol (NP1EO and NP2EO) were also detected in most effluent and tissue samples. The U.S. Environmental Protection Agency (2005) chronic water quality criterion (WQC) for nonylphenol in marine systems is 1.7 ng/mL, a value that approximates the observed effluent concentration for the Tacoma WWTP reported here. Also, the U.S. Environmental Protection Agency (2010) provides toxic equivalency factors (TEFs) for aquatic species exposed to nonylphenol ethoxalates and these are considered to be about 50% as potent as NP (NP = 1; NP1EO and NP2EO = 0.5). When these TEFs are applied to the observed effluent concentrations, the combined concentrations of NP and these 2 ethoxylates exceed the WQC approximately 2-fold. No toxicity data for alkylphenols in fish tissue could be found for comparison to our observed values.

Several studies indicate adverse effects for fish exposed to alkylphenols at environmentally-relevant concentrations. One study reported severe reductions in growth for rainbow trout (*O. mykiss*) exposed separately to 1 ng/mL of NP and NP2EO at concentrations as low as 1 ng/mL that persisted for several weeks to months after exposure was terminated (Ashfield et al., 1998). Our measured concentrations for each of these compounds in effluent was higher than this growth impairment concentration and combined were approximately 3-fold higher. The second study observed a negative correlation between catch data for Atlantic salmon and the application of a pesticide to various tributaries within a river basin during smolt development for a one year period (Fairchild et al., 1999). Based on the analysis of Fairchild et al. (1999), the authors concluded that NP (an adjuvant for the pesticide application) was responsible for excess mortality during this life stage. Similar effects were also noted by Fairchild et al. (1999) for spray events over several years for another anadromous species (Blueback herring, *Alosa aestivalis*).

4.5. Implications for potential adverse ecological effects in Puget Sound

As discussed, the observed water and tissue concentrations of numerous analytes detected in the two effluent impacted estuaries in Puget Sound have the potential to cause adverse effects in both fish species in this study. Endocrine and metabolic disruption may have important impacts on adult fish, such as staghorn sculpin examined here; however, metabolic disruption is even more critical for actively growing juvenile salmonids. A recent study concluded that juvenile Chinook salmon migrating through contaminated estuaries in Puget Sound exhibited a two-fold reduction in survival compared to those migrating through uncontaminated estuaries (Meador, 2014). Some of the lowest survival rates for juvenile Chinook salmon were seen for estuaries that have WWTPs discharging into the estuary or nearshore areas where this species rears before heading into open water.

Some of the compounds observed in Chinook salmon and staghorn sculpin tissue may also accumulate in larger fish that prey on these species, in addition to aquatic-dependent wildlife including birds and marine mammals (Diehl et al., 2012). Although a few studies have examined potential bioaccumulation, biomagnification, or potential adverse effects for these higher trophic-level aquatic predators (Arnold et al., 2014; Gaw et al., 2014), these are relatively uncommon. Another relatively unexplored aspect concerns the bioaccumulation and adverse effects of these compounds on estuarine invertebrates and algae, which are an important component of the foodweb for fish. In addition to enhanced bioaccumulation via dietary uptake, reductions in prey species could impact growth rates of fish residing in these estuaries.

A noteworthy outcome of the present study is the occurrence of several compounds in water and tissue that have the potential to affect fish growth, behavior, reproductive impairment, immune function, and antibiotic resistance. One recent review provides a summary of studies on the effects of endocrine disruptors on immune system in fish (Milla et al., 2011). Many of these agents, such as metformin, may impact multiple systems such as growth and reproductive pathways. It is expected that few, if any, of these compounds would result in direct mortality to estuarine organisms; however, all of the above mentioned responses could lead to indirect mortality or reduced population fitness. As noted by Spromberg and Meador (2005) and Meador (2014) even a minor inhibition in juvenile salmonid immune function or growth likely results in a major impact on survivability during their first year in marine waters.

5. Conclusions

The CECs investigated in the current study were selected based upon their widespread use, in addition to the likelihood of continued use and potential for increased contamination in the future. Accordingly, regulation and assessment of the ecological and human health risks of these compounds continue to warrant high interest as human populations increase. It should be noted that the results of the present study represent a snapshot of concentrations that exist at our sites and that may vary day-to-day and seasonally. Surprisingly, a large percentage of the chemicals detected in Puget Sound effluents are among the highest concentrations reported in the U.S., which may be a function of per capita usage of these compounds or the treatment processes used at these WWTPs. The fact that we observed multiple pharmaceuticals capable of interacting with a variety of molecular targets in our two fish species, leads to the potential for mixture interactions on critical physiological processes. These interactions can be additive, synergistic, or inhibitory, which are difficult to assess in the field or laboratory. Future work developing and applying mechanism-based biomarkers linked to physiological outcomes resulting from exposure to CECs would help close this data gap and lead to better predictions of adverse ecological impacts.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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Highlights

- Provides data on a broad range of contaminants of emerging concern in marine waters and whole-body fish, which is uncommon in the literature.
- Detected analytes in effluent higher than most concentrations reported for large wastewater treatment plants in the United States.
- Whole-body fish tissue concentrations of CECs elevated for some analytes and occur at greater than expected concentrations.
- Several compounds were observed at concentrations that may result in adverse responses in biota.
- Loading of CEC analytes to estuaries considered substantial and expected to increase with human population growth.

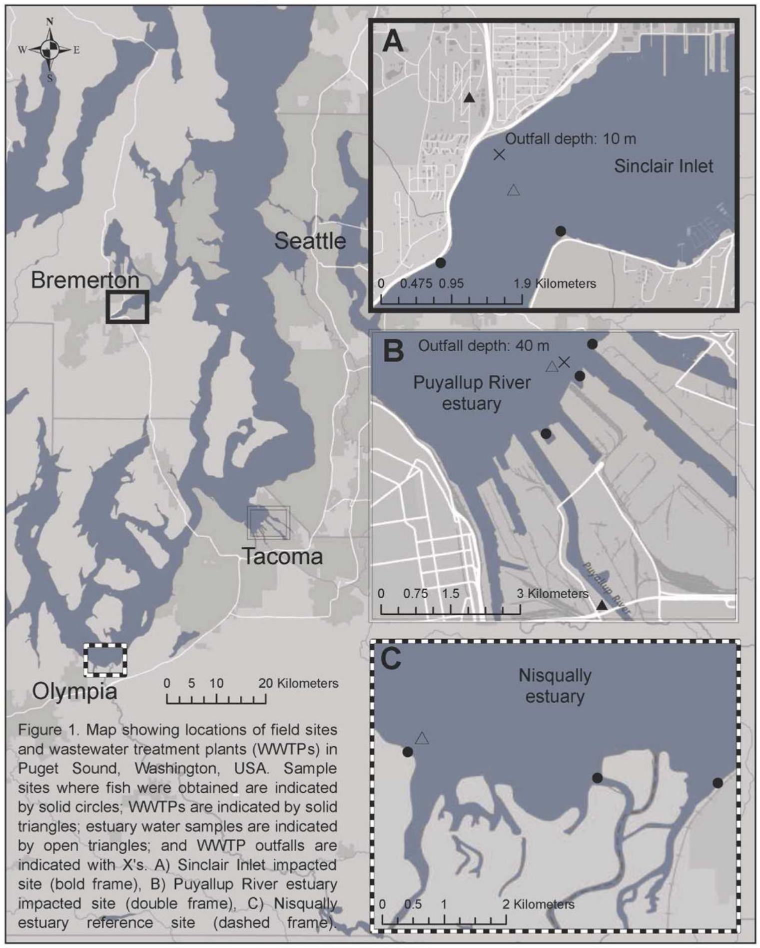


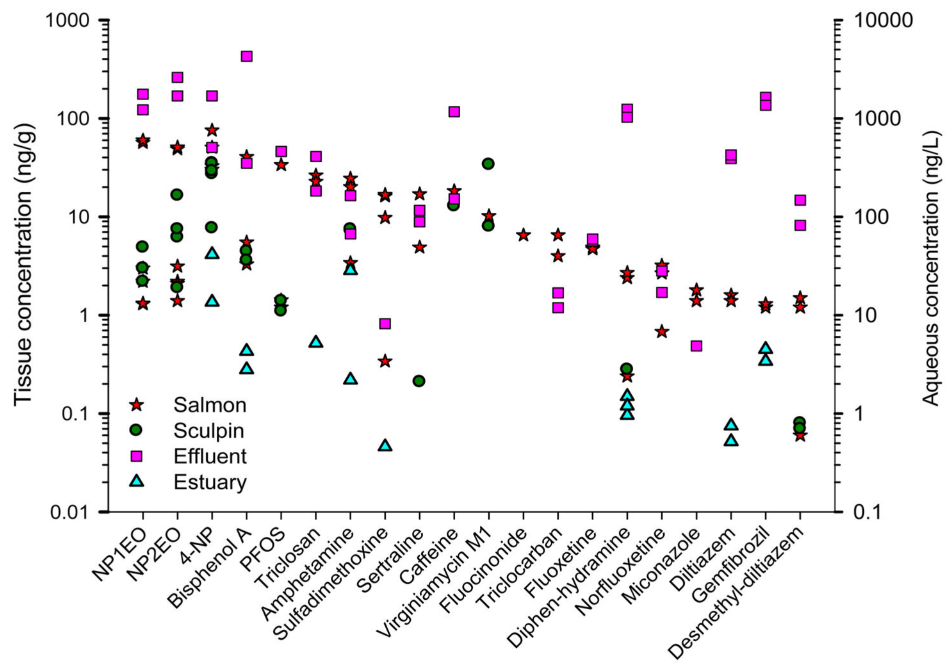
Figure 1. Map showing locations of field sites and wastewater treatment plants (WWTPs) in Puget Sound, Washington, USA. Sample sites where fish were obtained are indicated by solid circles; WWTPs are indicated by solid triangles; estuary water samples are indicated by open triangles; and WWTP outfalls are indicated with X's. A) Sinclair Inlet impacted site (bold frame), B) Puyallup River estuary impacted site (double frame), C) Nisqually estuary reference site (dashed frame).

Figure 1.

Map of Puget Sound and estuaries sampled with locations of wastewater treatment plant outfalls and sampling sites.

Map showing locations of field sites and wastewater treatment plants (WWTPs) in Puget Sound, Washington, USA. Sample sites where fish were obtained are indicated by solid circles; WWTPs are indicated by solid triangles; estuary water samples are indicated by open triangles; and WWTP outfalls are indicated with X's. A) Sinclair Inlet impacted site (bold frame), B) Puyallup River estuary impacted site (double frame), C) Nisqually estuary reference site (dashed frame).

Figure 2a



Tissue concentration (ng/g)
Aqueous concentration (ng/L)
1000
100
10
1
0.1
0.01
10000
1000
100
10
1
0.1
Salmon
Sculpin
Effluent
Estuary
NP1EO
NP2EO
4-NP
Bisphenol A
PFOS
Triclosan
Amphetamine
Sulfadimethoxine
Sertraline
Caffeine
Virginiamycin M1
Fluocinonide
Triclocarban
Fluoxetine
Diphen-hydramine
Norfluoxetine
Miconazole
Diltiazem
Gemfibrozil
Desmethyl-diltiazem

Figure 2b

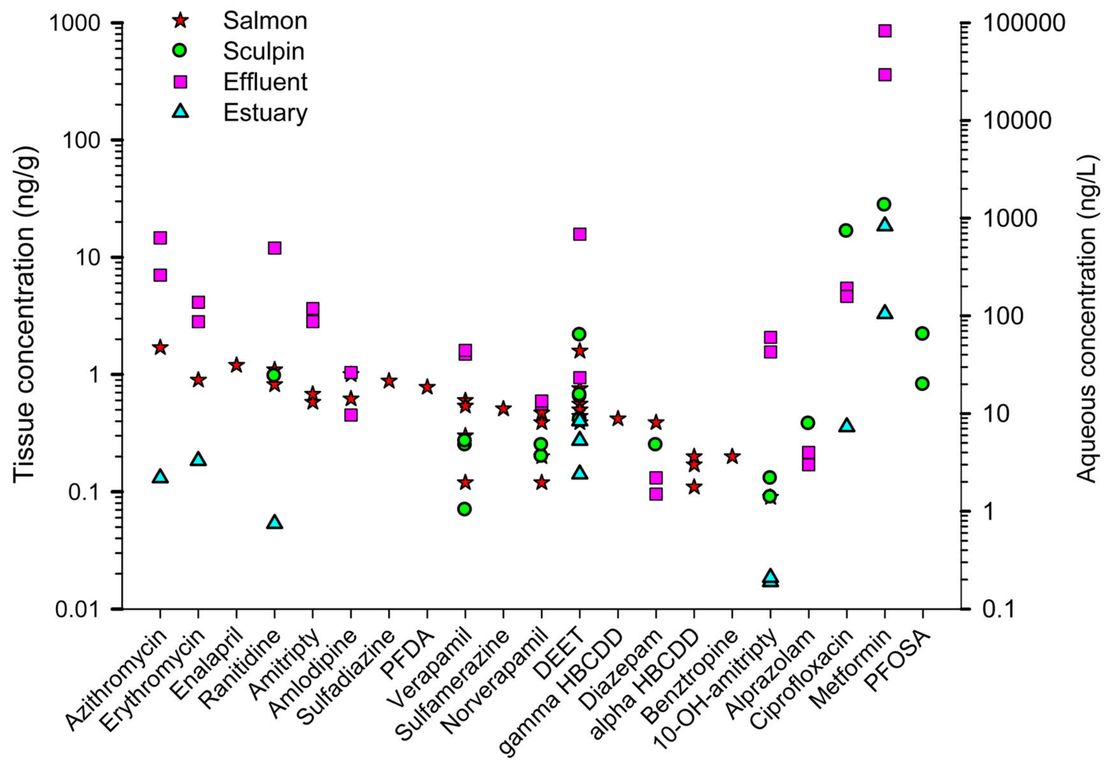


Figure 2.

Plot showing occurrence of detected analytes in fish, estuary water, and effluent. Data are ordered from high to low concentrations in juvenile chinook. All replicate data shown for each matrix.

Table 1

Sampling locations, water and fish collection data, composition of chemistry composites, and estuary parameters.

	Puyallup estuary	Sinclair Inlet	Nisqually estuary	Voight's Creek Hatchery
	Collection data			
Coordinates	47°16′35.4″N 122°24′58.0″W	47°32′24.4″N 122°39′44.3″W	47°05′56.4″N 122°42′01.8″W	47°04′58.8″N 122°10′40.8″W
Sample dates fish	21 Aug 2013, and 4 Sept 2013 (La); 16 June 2014 (Ot); 29 June 2014, and 7 and 13 Aug 2014 (La)	9 and 11 June 2014 (Ot); 27 July 2014 (La)	27 Aug 2013 (La); 19 June 2014 (Ot); 4 Aug 2014 (La)	29 May 2014 (Ot)
n fish collected	Ot: 75 La: 18§ and 31$^{\wedge}$	Ot: 38 La: 40$^{\wedge}$	Ot: 72 La: 24§ and 47$^{\wedge}$	Ot: 56
Mean (SD) salmon wt. (g)	5.4 (2.4)	13.4 (8.2)	6.8 (1.5)	5.4 (0.9)
Mean (SD) sculpin wt. (g)	La§: 60.7 (29.4) La$^{\wedge}$: 22.7 (20.5)	La$^{\wedge}$: 18.8 (6.6)	La§: 36.7 (14.3) La$^{\wedge}$: 16.1 (5.0)	N/A
% hatchery chinook	70%	71%	100%	100%
Salmon CF mean (sd)	0.94 (0.14)	0.90 (0.19)	0.96 (0.12)	1.09 (0.12)
Sample dates water	21 Aug 2013 (EW); 17 Sept 2014 (EF)	22 July 2014 (EW); 9 Sept 2014 (EF)	27 Aug 2013 (EW)	N/A
<u>Chemistry composites</u>				
N Fish/chem composite, lipids %	Ot A: 10, 4.3% Ot B: 12, 3.2% La§: 3, 1.6% La$^{\wedge}$: 5, 1.9%	Ot A: 3, 3.3% Ot B: 3, 1.5% La$^{\wedge}$: 3, 1.7%	Ot: 9, 2.5% La§: 4, 2.1% La$^{\wedge}$: 3, 1.6%	Ot: 12, 5.1%
Mean (SD) salmon wt. (g)	Ot A: 5.5 (1.3) Ot B: 4.1 (0.6)	Ot A: 14.1 (4.7) Ot B: 16.9 (9.0)	5.6 (0.7)	5.4 (0.9)
Mean (SD) sculpin wt. (g)	La§: 47.5 (50.2) La$^{\wedge}$: 9.4 (1.6)	La$^{\wedge}$: 30.9 (6.2)	La§: 48.1 (31.8) La$^{\wedge}$: 16.8 (2.8)	N/A
<u>Estuary parameters</u>				
pH	8.04	8.45	7.62	—
Salinity (ppt)	23.5	27	15.5	0
Temp (°C)	12.5	12.5	13.5	10
Oxygen (mg/L)	8.2	15	10.6	12

EW= estuary water, EF=effluent, Ot= *Oncorhynchus tshawytscha* (Chinook salmon), La= *Leptocottus armatus* (staghorn sculpin).

§=2013 sampling year sculpin,

^=2014 sampling year sculpin. CF is condition factor (=weight (g)^{2}/length (mm)^{3}). Percent hatchery fish based on the presence of an adipose fin. Estuary parameters determined at time of water sampling. SD is standard deviation.

Table 2

Range of observed concentrations for CECs detected in water or fish.

Analytes	Range for effluent (ng/L)	Range for estuary water (ng/L)	Range for salmon (ng/g)	Range for sculpin (ng/g)	WWTP output (g/d)#		Percentile ranking for effluent
					Bremerton	Tacoma	
Albuterol	36 – 41	12			0.54	2.03	> 90th
Alprazolam	3.0 – 4.0			0.38	0.04	0.23	> 95th
Amitriptyline	88 – 119		0.58 – 0.68		1.58	4.97	> 99th
10-OH-amitriptyline	43 – 60	0.19 – 0.21	0.09	0.13	0.80	2.43	*> 99th
Amlodipine	9.7 – 26		0.62 – 1.0		0.13	1.49	> 99th
Amphetamine	67 – 164	2.2 – 29	3.4 – 25	7.3 – 25	2.17	3.81	> 99th
Androstenedione	8.4				0.11		
Atenolol	1,700 – 2,440	3 – 22			22.5	138.5	> 95th
Atorvastatin	68					3.87	*> 99th
Azithromycin	261 – 629	2.2	1.7		8.33	14.8	
Benzoylecgonine	151 – 293	0.50 – 0.80			3.88	8.57	
Benztropine	0.57 – 0.93		0.20		0.01	0.03	*> 99th
Bisphenol A	350 – 4,290	2.8 – 4.3	3.3 – 41	3.6 – 4.5	4.64	243	
Caffeine	152 – 1170		18	13	15.5	8.63	
Carbamazepine	510 – 735	1.9			6.76	41.7	> 99th
Cimetidine	194					11.0	> 99th
Ciprofloxacin	158 – 192	7.3		17	2.54	8.97	> 80th
Clarithromycin	52 – 181				0.69	10.3	
Cocaine	9 – 59	0.30			0.78	0.48	
Codeine	290 – 178				2.36	16.5	
Cotinine	115 – 340				4.50	6.53	
DEET	23.3 – 684	2.4 – 5.3	0.39 – 1.6	0.41 – 2.2	9.06	1.32	
Diazepam	1.5 – 2.2		0.39	0.25	0.03	0.09	
Dehydronifedipine	13 – 15				0.20	0.73	
Diltiazem	390 – 425	0.52 – 0.75	1.4 – 1.6		5.17	24.1	> 99th
Diltiazem desmethyl	82 – 148		0.06 – 1.5	0.07 – 0.08	1.96	4.64	> 99th

Analytes	Range for effluent (ng/L)	Range for estuary water (ng/L)	Range for salmon (ng/g)	Range for sculpin (ng/g)	WWTP output (g/d)#		Percentile ranking for effluent
					Bremerton	Tacoma	
Dimethylxanthine 1,7	873 – 2060				27.3	49.6	
Diphenhydramine	1030 – 1240	0.96 – 1.5	0.24 – 2.7	0.28	16.4	58.5	
Enalapril	5.9		1.2			0.34	> 80th
Erythromycin	87 – 138	3.3	0.90		1.83	4.96	
Estrone	4.5 – 58				0.77	0.25	> 85th
Fluocinonide			6.5				
Fluoxetine	57 – 60		4.9		0.75	3.38	> 99th
Furosemide	994 – 1290				17.1	56.4	> 95th
Gemfibrozil	1360 – 1640	3.4 – 4.5	1.3		21.7	77.2	> 90th
Glipizide	22 – 23				0.29	1.24	*> 99th
Glyburide	7.6 – 11				0.14	0.43	*> 99th
α-HBCDD			0.10 – 0.20				
γ-HBCDD			0.42				
Hydrochlorothiazide	411 – 578				7.66	23.3	≈ 5 th
Hydrocodone	69 – 74				0.98	3.93	> 80th
Ibuprofen	116 – 1060				14.0	6.59	> 80th
2-OH-ibuprofen	1,160 – 4,550				60.3	65.9	> 95th
Lincomycin	27					1.55	*> 99th
MBP		289 – 491					
MEHP	0.40					0.02	
Meprobamate	513 – 623				8.25	29.1	
Metformin	29,300 – 82,700	105 – 832		28	388	4695	
Metoprolol	805 – 835				10.7	47.4	> 90th
Miconazole	4.9		1.8			0.28	
Naproxen	106 –701				9.29	6.02	
Norfluoxetine	17 – 28		0.68 – 3.2		0.37	0.97	> 99th
Norverapamil	13 – 14		0.12 – 0.47	0.20 – 0.30	0.17	0.77	> 95th
4-NP	506 – 1690	41	30 – 76	7.7 – 35	6.70	95.9	
NP1EO	1,220 – 1,760		1.3 – 60	3 – 4.9	23.3	69.3	

Analytes	Range for effluent (ng/L)	Range for estuary water (ng/L)	Range for salmon (ng/g)	Range for sculpin (ng/g)	WWTP output (g/d)#		Percentile ranking for effluent
					Bremerton	Tacoma	
NP2EO	1,690 – 2,610		1.4 – 51	1.9 – 17	34.6	95.9	
Ofloxacin	108 – 387				5.13	6.13	> 90th
Ormetoprim			44 – 1,600				
Oxycodone	158 – 231				2.09	13.1	> 95th
Paroxetine	6.6 – 42				0.56	0.37	*> 99th
PFBA	6.7					0.38	
PFBS	13					0.17	
PFDA			0.78				
PFHpA	3 – 7.5				0.10	0.17	
PFHxA	15 – 53				0.71	0.86	
PFHxS	55				0.73	0.00	
PFNA	2					0.11	
PFOA	7.6 – 12				0.16	0.43	
PFOS	461		1.2 – 34	1.1 – 1.4	6.11	0.00	
PFOSA				0.82 – 2.2			
PFPeA	3.4 – 4.7				0.06	0.19	
Promethazine	3.8					0.21	*> 99th
Propoxyphene	0.7 – 1.9				0.02	0.04	> 80th
Propranolol	76 – 109				1.00	6.19	> 95th
Ranitidine	494	0.75	0.82 – 1.1	0.97		28.1	> 95th
Roxithromycin	3.8					0.22	
Sertraline	89 – 116		17	0.20	1.54	5.05	> 95th
Simvastatin	34					1.95	*> 99th
Sulfadiazine			0.88				
Sulfadimethoxine	8.2	0.46	0.34 – 17			0.47	*> 99th
Sulfamerazine			0.51				
Sulfamethoxazole	1380	1.5 – 4.2				78.4	> 90th
Testosterone		1.9					
Thiabendazole	24 – 27				0.36	1.35	

Analytes	Range for effluent (ng/L)	Range for estuary water (ng/L)	Range for salmon (ng/g)	Range for sculpin (ng/g)	WWTP output (g/d)#		Percentile ranking for effluent
					Bremerton	Tacoma	
Triamterene	151 – 156				2.00	8.86	> 95th
Triclocarban	12 – 17		6.5		0.16	0.96	
Triclosan	250 – 538	5.2	26		7.13	14.2	
Trimethoprim	742 – 852	2.3			9.83	48	> 99th
Valsartan	2010 – 3000	5.4			26.6	170	> 80th
Verapamil	40 – 44		0.30 – 0.60	0.07 – 0.27	0.54	2.52	> 80th
Virginiamycin M1			10	8 – 34			
Warfarin	6.2					0.35	*> 99th
Detected analytes	81	25	37	21			
Sum kg/d for sample flow					0.82	6.66	
kg/d at maximum flow					3.5	17	

All blank values indicate a concentration <RL. Range shows minimum and maximum for each matrix (effluent, estuary water, or fish tissue) and type (sculpin or salmon). All single values indicate at least one site with a quantifiable concentration. Tissue concentrations are whole-body wet weight. Grams/day (g/d) for each analyte shown based on measured concentration (Table S4) and flow rate on the date of collection (personal communication from plant operators). Also shown is predicted kg/d for flow at the time of sampling and maximum flow. Our effluent concentrations expressed as percentile ranking compared to Kostich et al. (2013, 2014) who analyzed 56 active pharmaceutical ingredients in the 50 largest WWTPs in the U.S.

* = all values from Kostich et al. (2014) below detection but detected in the present study. See Table S1 for all analyte abbrevations and text for details.

NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7
BIOLOGICAL OPINION

Title:	Biological Opinion on EPA Pesticides General Permit for Discharge of Pollutants into U.S. Waters
Consultation Conducted By:	Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce
Action Agency:	Environmental Protection Agency
Publisher:	Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce
Consultation Tracking number:	FPR-2016-9154
Digital Object Identifier (DOI):	https://doi.org/10.7289/V54Q7S33

NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7 BIOLOGICAL OPINION

Action Agency: Environmental Protection Agency

Activity Considered: Reissuance of the Pesticides General Permit for Discharge of Pesticide Pollutants into Waters of the United States

Consultation Conducted By: Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service

Approved: Donna S. Wieting

Donna S. Wieting
Director, Office of Protected Resources

Date: OCT 17 2016

Public Consultation Tracking System number: FPR-2016-9154

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1 INTRODUCTION

Section 7(a)(2) of the Endangered Species Act (ESA) requires Federal agencies to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. When a Federal agency's action "may affect" a protected species, that agency is required to consult formally with NOAA's National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), together, the Services, depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS or the USFWS concurs with that conclusion (50 CFR §402.14(b)).

Section 7(b)(3) of the ESA requires that at the conclusion of consultation, NMFS and/or USFWS provide a biological opinion (opinion) stating how the Federal agencies' actions will affect ESA-listed species and their designated critical habitat under their jurisdiction. If the analyses concludes that the action will jeopardize an ESA-listed species or adversely modify designated critical habitat, section 7(b)(3) of the ESA directs the consulting agency to provide reasonable and prudent alternatives that the action agency can implement to avoid jeopardy or adverse modification or indicate whether there are no reasonable and prudent alternatives. If an incidental take is expected, section 7(b)(4) of the ESA requires the consulting agency to provide an incidental take statement that specifies the impact of any incidental taking and includes reasonable and prudent measures to minimize such impacts.

This document represents NMFS' opinion on the U.S. Environmental Protection Agency's (EPA's) reissuance of its Pesticides General Permit (PGP) authorizing discharges of biological pesticides and residues from chemical pesticides (together, pesticide pollutants) to waters of the U.S. and the implications of these discharges for threatened and endangered species and their designated critical habitat under NMFS' jurisdiction. The EPA uses general permits issued under section 402, the National Discharge Elimination System (NPDES) of the Clean Water Act (33 U.S.C. 1342 et seq.; CWA), to authorize routine discharges by multiple dischargers. Coverage for discharges under a general permit is granted to applicants after they submit a notice of intent to discharge (NOI[1]). Once the NOI is submitted and any review period specified under the PGP has closed, the applicant is authorized to discharge under the terms of the general permit. Under the PGP, however, some dischargers are automatically covered without submitting an NOI. The PGP authorizes discharges only of pesticide pollutants from pesticides that EPA has registered for use under the Federal Insecticide Fungicide and Rodenticide Act (FIFRA), 7 U.S.C. 136-136y.

The opinion and incidental take statement were prepared by NMFS' Endangered Species Act Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 CFR §402. This document represents NMFS' opinion on the effects of these actions on endangered and threatened species and designated critical habitat that has been designated for those species. A complete record of this consultation is on file at NMFS' Office of Protected Resources in Silver Spring, Maryland.

[1] There are many types of NOIs, throughout this document NOI refers to the notice of intent to discharge into waters of the U.S. in the action area.

1.1 Background

On October 14, 2011, NMFS' Office of Protected Resources issued a biological opinion on EPA's first CWA PGP authorizing discharges of FIFRA-approved pesticides to waters of the U.S.. NMFS concluded that EPA's issuance of the PGP was likely to jeopardize listed species and likely to adversely modify designated critical habitat. The EPA issued its final NPDES PGP on October 31, 2011. The PGP is valid until October 31, 2016.

1.2 Consultation History

The current PGP expires on October 31, 2016. Below we summarize meetings and communications on the ESA section 7 consultation process on a proposed new PGP. Pre-consultation discussions began in 2015. Formal consultation was initiated on May 25, 2016.[2]

On June 4, 2015, EPA Office of Water[3] met with the Services to provide an update on the status of the 2016 PGP, share information collected under the 2011 PGP, and share a draft schedule for the new permit issuance.

On June 16, 2015, EPA shared data extracted from the PGP annual reports and Best Management Practices worked out for PGP discharges in the state of Idaho.

On July 29, 2015, EPA met via conference call with NMFS to coordinate development of EPA's biological evaluation (BE).

On August 3, 2015, EPA and NMFS met via conference call to discuss the analysis framework for the BE.

On August 7, 2015 NMFS shared a draft analysis framework for the BE.

On August 18, 2015, EPA and NMFS met via conference call to discuss the draft analysis framework.

On October 1, 2015, NMFS submitted comments on the PGP NOI form to EPA

On January 6, 2016, EPA met with NMFS via conference call to discuss the creation of a webmap of locations where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur.

On February 26, 2016, EPA transmitted its request to initiate formal consultation for EPA's reissuance of the PGP. EPA's BE, submitted with the request, contained information gathered under the requirements of the 2011 PGP.

On April 21, 2016, NMFS transmitted a letter identifying additional information needed before formal consultation could begin.

On April 22, 2016, NMFS transmitted a draft description of the action and *Action Area* for review by EPA.

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[2] In 2013, The National Academy of Sciences issued *Assessing Risks to Endangered and Threatened Species from Pesticides* (National Research Council, 2013;http://www.nap.edu/catalog.php?record_id=18344). In response to the report's recommendations EPA, NMFS, and the US Fish and Wildlife Service have been working closely and have developed interim approaches for ESA consultations on EPA's FIFRA decisions. Because the agencies have not reached final decisions, neither EPA's BE nor this opinion rely on the interim approaches.

[3] Both EPA's Office of Water and Office of Pesticide Programs participated in this consultation. We will refer simply to "EPA," unless there is a reason to identify which office participated.

On May 25, 2016, EPA supplied NMFS with the additional information needed to initiate formal consultation.

On August 9, 2016, NMFS transmitted draft reasonable and prudent alternatives (RPAs) and a draft incidental take statement (ITS) to EPA for review.

On September 15, 2016, EPA provided comments on the draft RPAs and ITS.

On October 13, 2016, NMFS provided EPA revised draft RPAs and ITS.

2 The Assessment Framework

Section 7(a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat.

"*Jeopardize the continued existence of* means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species." 50 CFR 402.02.

"Destruction or adverse modification means a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features. " 50 CFR 402.02. An ESA Section 7 assessment involves the following steps:

Description of the Proposed Action (Section 3), *Interrelated and Interdependent Actions* (Section 4), and *Action Area* (Section 5)*:* We describe the proposed action and those aspects (or stressors) of the proposed action that may have direct or indirect effects on the physical, chemical, and biotic environment, we identify any interrelated and interdependent actions, and describe the action area with the spatial extent of those stressors.

Status of Species and Designated Critical Habitat (Section 6).We identify the ESA-listed species and designated critical habitat that are likely to co-occur with those stressors in space and time and evaluate the status of those species and habitat. In this Section, we also identify those *Species and Designated Critical Habitat Not Considered Further in the Opinion* (Section 6.1), because these resources will either not be affected or are not likely to be adversely affected.

Environmental Baseline (Section 7). We describe the environmental baseline in the action area including: past and present impacts of Federal, state, or private actions and other human activities in the action area; anticipated impacts of proposed Federal projects that have already undergone formal or early section 7 consultation, impacts of state or private actions that are contemporaneous with the consultation in process.

Effects of the Action: Risk Assessment (Section 8.1) and *Programmatic Analysis* (Section 8.2): To determine the effects of the action, we conduct two separate analyses. First, in the Risk Assessment, we evaluate the potential adverse effects of discharges under the PGP on ESA-listed species and designated critical habitat under NMFS' jurisdiction, without consideration of the protective measures of the PGP. To do this, we begin with problem formulation that identifies and integrates the stressors of the action with the species status (Section 6) and the

Environmental Baseline (Section 7) and formulate risk hypotheses. The risk hypotheses identify assessment endpoints of concern for listed species and designated critical habitat. To evaluate the risk hypotheses, we consider the exposure by individual members of listed species (exposure analysis) and essential features of designated critical habitat, and what expected responses might be (response analysis). If the assessment endpoints of the individuals or the essential features indicate adverse effects, we evaluate whether those responses will affect populations or subpopulations of species or the designated critical habitat (risk characterization.). Second, since we conclude that population level effects to species and adverse effects to essential features of designated critical habitat are likely to occur as a result of the pesticide discharges, we conduct a Programmatic Analysis. In this analysis, we evaluate whether the process and the protective measures in the PGP are enough to allow EPA to insure that its program is not likely to jeopardize listed species or destroy or adversely modify designated critical habitat. To do so, we consider seven questions focused on EPA's knowledge and ability to respond.

Integration and Synthesis (Section 9): In this section we integrate the analyses in the opinion to summarize the consequences to ESA-listed species and designated critical habitat under NMFS' jurisdiction.

Cumulative Effects (Section 10): Cumulative effects are the effects to listed species and designated critical habitat of future state or private activities that are reasonably certain to occur within the action area. 50 CFR 402.02. Effects from future Federal actions that are unrelated to the proposed action are not considered because they require separate section 7 consultation..

Conclusion (Section 11); With full consideration of the status of the species and the designated critical habitat, we consider the effects of the action within the action area on populations or subpopulations and on essential habitat features when added to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:

Reduce appreciably the likelihood of survival and recovery of each ESA-listed species in the wild by reducing its numbers, reproduction, or distribution, and state our conclusion as to whether the action is likely to jeopardize the continued existence of that species; or

Appreciably diminish the value of designated critical habitat for the conservation of a ESA-listed species, and state our conclusion as to whether the action is likely to destroy or adversely modify designated critical habitat.

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated critical habitat, then we must identify reasonable and prudent alternative(s) (RPAs) to the action, if any, or indicate that to the best of our knowledge there are no reasonable and prudent alternatives. See 50 C.F.R. § 402.14.

In addition, we include an incidental take statement (ITS) that specifies the impact of the take, reasonable and prudent measures (RPMs) to minimize the impact of the take, and terms and conditions to implement the RPMs. ESA Section 7(b)(4); 50 CFR 402.14(i). We also provide discretionary conservation recommendations that may be implemented by EPA. 50 CFR 402.14(j). Finally, we identify the circumstances in which reinitiation of consultation is required. 50 CFR 402.16.

To comply with our obligation to use the best scientific and commercial data available, we collected information identified through searches of ISI Web of Science, Medline, scientific publisher databases (e.g., Elsevier), government databases (e.g., EPA's National Service Center for Environmental Publications), and literature cited sections of peer reviewed articles, species listing documentation, and reports published by government and private entities. This opinion is based on our review and analysis of various information sources, including:

EPA' s Biological Evaluation (BE) for the PGP;

the PGP and its fact sheet;

toxicity data provided by EPA;

annual reports and NOI submitted under the 2011 PGP;

NPDES program compliance and enforcement data;

Section 7 consultations on pesticide re-registrations;

status reviews, recovery plans, and listing notices for ESA-listed species and designated critical habitat;

reports on the status and trends of water quality; and

peer reviewed research.

These resources were used to identify information relevant to the potential stressors and responses of ESA-listed species and designated critical habitat under NMFS' jurisdiction that may be affected by the proposed action to draw conclusions on risks the action may pose to the continued existence of these species and the value of designated critical habitat for the conservation of ESA-listed species.

3 Description of the Proposed Action

The EPA proposes to re-issue the PGP to authorize point source discharges of pesticide pollutants directly to waters of the U.S. by pesticide Operators. An *Operator* is any entity who performs the application of a pesticide or who has day-to-day control of the application (i.e., *Applicators*) or any entity with control over the decision to perform pesticide applications including the ability to modify those decisions (i.e., *Decision-makers*). All Applicators and Decision-makers are Operators, and Operators can be either or both an Applicator and a Decision-maker. When an Operator is both Applicator and Decision-maker, the Operator must comply with all requirements for both. Some Decision-Makers must submit NOIs prior to discharge, as described in Appendix A of the PGP and Table 1, with discharge authorized within 30 days after filing the NOI. Any proposed discharge to waters of the U.S. containing *NMFS' Listed Resources of Concern* requires a NOI. The PGP contains an exception for response to a Declared Pest Emergency, but still requires the filing of a NOI within 30 days of beginning a discharge. If the PGP does not require a NOI, then the discharge is authorized without notice to EPA.

The PGP defines *pesticides* as (1) any substance or mixture of substances intended for preventing, destroying, repelling, or mitigating any pest, (2) any substance or mixture of substances intended for use as a plant regulator, defoliant, or desiccant, and (3) any nitrogen stabilizer. *Pesticide pollutants* are defined as all biological pesticides[4] and those chemical pesticides that leave a residue. A *pesticide residue* is that portion of a pesticide application that is discharged from a point source to waters of the U.S. and no longer provides its pesticidal purpose. Pesticide residues also include any degradates of the pesticide. The EPA, in its BE for the PGP assumed that "all chemical pesticides will leave a residue once the product has performed its intended purpose."

The PGP authorization to discharge pesticide pollutants into waters of the U.S. is available to eligible Operators in those States and Territories where the EPA is the permitting authority: American Samoa, District of Columbia, Guam, Idaho, Johnston Atoll, Massachusetts, Midway Island, New Hampshire, New Mexico, Northern Mariana Islands, Puerto Rico, and Wake Island. The proposed general permit will also authorize discharges of pesticide pollutants into waters of the U.S. resulting from pesticide applications on Federal lands located in Colorado, Delaware, Vermont, and Washington, as well as Indian lands nationwide. The statutory authority for the PGP is the NPDES of the Clean Water Act (33 U.S.C. 1342 et seq.; CWA), which is administered by EPA's Office of Water. The purpose of the proposed general permit is to satisfy the goals and policies of the CWA (33 U.S.C. 1251).

Although the PGP would authorize discharges of pesticide pollutants into waters of the U.S. under the CWA, these pesticides and their use patterns have been evaluated, registered, and regulated under the FIFRA as amended by the Food Quality Protection Act of 1996, which is administered by the EPA's Office of Pesticide Programs.

The EPA is requiring that discharges of pesticide pollutants to waters of the U.S. resulting from the four use patterns be subject to the terms of the PGP under the CWA. This provides EPA with the authority to enforce CWA requirements that may not have been addressed and may be in addition to requirements under FIFRA. Operators must comply with all other applicable federal and state laws and regulations that pertain to the application of pesticides. For example, the PGP does not negate the requirements under the FIFRA and its implementing regulations to use registered pesticides consistent with the product's labeling. Violation of certain FIFRA requirements, such as exceeding label rates, would also be a violation of the PGP and therefore a violation of the CWA.

This proposed permit does not affect the existing CWA exemptions for irrigation agriculture return flows or agricultural stormwater runoff. These discharges are excluded from the definition of a point source under Section 502 (14) of the CWA. Agricultural stormwater runoff and irrigation agriculture return flows do not require NPDES permits. Therefore, runoff from irrigation agriculture return flows and agricultural stormwater are not considered in this opinion.

3.1 Authorized Discharges

The proposed PGP authorizes point-source discharges of pesticide pollutants into aquatic habitats from the application of pesticides directly to or at waters edge for waters of the U.S. as a result of the following four use patterns:

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[4] EPA Office of Water is relying on existing regulatory definitions in 40 CFR 174.3, 158.2000(a)(1), and 158.2100(b) developed under FIFRA to define the term "biological pesticides" to include microbial pesticides [40 CFR 158.2100(b)], biochemical pesticides [40 CFR 158.2000(a)(1)], and plant-incorporated protectants. [40 CFR 174.3]

Mosquito and Other Flying Insect Pest Control: to control public health/nuisance and other flying insect pests that develop or are present during a portion of their life cycle in or above standing or flowing water. Public health/nuisance and other flying insect pests in this use category include mosquitoes and black flies.

Weed and Algae Pest Control: to control weeds, algae, and pathogens that are pests in water and at water's edge, including ditches and/or canals.

Animal Pest Control: to control animal pests in water and at water's edge. Animal pests in this use category include lampreys, other fish, insects, mollusks, and pathogens.

Forest Canopy Pest Control: application of a pesticide to a forest canopy to control the population of a pest species (e.g., insect or pathogen) where, to target the pests effectively, a portion of the pesticide unavoidably will be applied over and deposited to water.

3.2 Limitations on Coverage

The PGP restricts coverage for discharges to waters impaired by pesticides, Tier 3 waters, discharges covered or previously covered under another NPDES permit, and discharges to waters used by species and designated critical habitat protected under the ESA.

3.2.1 Discharges to Pesticide-impaired Waters

Discharges from a pesticide application to waters of the U.S. are not eligible for coverage under the PGP if the water is identified by EPA as impaired by either the specific active ingredient in that pesticide or its degradate, as listed at www.epa.gov/OWOW/tmdl/. Impaired waters are those that have been identified as not meeting applicable state or tribal water quality standards pursuant to section 303(d) of the CWA. These include waters with EPA-approved or EPA-established total maximum daily loads (TMDLs) and waters for which EPA has not yet approved or established a TMDL. If there is evidence that shows the water is no longer impaired, Operators may submit this information to EPA and request coverage under the PGP.

3.2.2 Discharges to Tier 3 Waters

In most cases the PGP does not cover discharges to waters designated by a state or tribe as Tier 3 (Outstanding National Resource Waters) for antidegradation purposes under Title 40 of the Code of Federal Regulations (CFR) 131.12(a)(3). Discharges to Tier 3 waters may be covered if the purpose of the pesticide application is to restore or maintain water quality or to protect public health or the environment and the application will not degrade water quality or will only degrade water quality on a short-term or temporary basis. In such cases a NOI is required and must specifically identify the Tier 3 water by the name, as listed, at www.epa.gov/npdes/pesticides.

3.2.3 Discharges Currently or Previously Covered by another Permit

Discharges are not eligible for coverage under the PGP if the discharge is already covered by another NPDES permit, or the discharge was included in a permit that in the past five years has been or is in the process of being denied, terminated, or revoked by EPA (this does not apply to the routine permit reissuance every five years).

3.2.4 Discharges to Waters with *NMFS' Listed Resources of Concern*[5]

The proposed 2016 PGP includes the same procedures, including a requirement to submit a NOI at least 30 days prior to discharge, as the 2011 PGP to assist in protecting *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP. The current definition found in Appendix A of the draft PGP is:

> *Federally-listed endangered and threatened species and federally-listed critical habitat for which NMFS' 2011 Endangered Species Act Section 7 Consultation Opinion on the United States Environmental Protection Agency's Proposed PGP concluded the draft 2011 PGP, absent any additional mitigating measures, would either jeopardize the continued existence of such species or destroy or adversely modify such critical habitat. The opinion included a Reasonable and Prudent Alternative, implemented through the PGP, to avoid likely jeopardy to ESA-listed species or adverse modification of critical habitat. Additional information, including maps noting where these resources overlap with PGP areas of coverage is available at www.epa.gov/npdes/pesticides.*

NMFS notes that this definition does not protect species or designated critical habitat listed after issuance of the 2011 permit and does not protect species anticipated to be listed and potentially affected by 2016 PGP-authorized discharges over the course of the five-year permit term.[6]

The draft PGP submitted for public comment in January of 2016 explains that EPA is currently conducting consultation with the Services under the ESA and that the results of consultation with the Services may result in additional or altered conditions to the final 2016 PGP.

The draft permit states in part 1.6 that Operators must comply with all conditions and/or requirements that address discharges from activities also covered under this PGP resulting from:

ESA section 7 consultation that Operators have completed with USFWS and/or NMFS, and/or

ESA section 10 permits issued to Operators by USFWS and/or NMFS.

As proposed, NOIs filed by Decision-makers will contain a section that directs the decision maker to self-certify whether pesticide application activities will overlap with the distribution of endangered or threatened species or designated critical habitat under NMFS' jurisdiction, and if so: 1) if their pesticide applications have undergone ESA section 7 consultations or if the Operator has received an ESA section 10(a)(1)(b) permit and; 2) a list of those endangered or threatened species, or designated critical habitat whose distributions overlap with treatment areas. A Decision-maker required to submit a NOI, either because the proposed discharge is to waters of the U.S. containing *NMFS' Listed Resources of Concern* or for one of the other discharges requiring a NOI, self-certifies their eligibility to discharge under the PGP under one of six eligibility criteria (A-F). These criteria are:

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[5] The draft PGP definition of "NMFS Listed Resources of Concern" is out of date in that it is limited to ESA-listed species and designated critical habitat at the time of issuance of the 2011 PGP and occurring in waters where EPA has permitting authority. For purposes of this opinion, we use the term "ESA-listed species and designated critical habitat under NMFS' jurisdiction" to include also the species listed and designated critical habitat designated since 2011.

[6] NMFS requested and included in a RPA to this opinion that EPA revise its definition of "NMFS listed Resources of Concern" to include species listed or designated critical habitat designated since the 2011 PGP was issued in an e-mail dated April 18, 2016.

Criterion A. Pesticide application activities will not result in a point source discharge to one or more waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP.

Criterion B. Pesticide application activities for which permit coverage is being requested will discharge to one or more receiving waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP, but consultation with NMFS under section 7 of the ESA has been concluded for pesticide application activities covered under the PGP. Consultations can be either formal or informal, and would have occurred only as a result of a separate federal action. The consultation addressed the effects of pesticide discharges and discharge-related activities on federally-listed threatened or endangered species and federally-designated critical habitat, and must have resulted in either:

i. A opinion from NMFS finding no jeopardy to federally-listed species and no destruction/adverse modification of federally-designated critical habitat; or

ii. Written concurrence from NMFS with a finding that the pesticide discharges and discharge-related activities are not likely to adversely affect federally-listed species or federally-designated critical habitat.

Criterion C. Pesticide application activities for which permit coverage is being requested will discharge to one or more waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP, but all "take" <u>of these resources</u> associated with such pesticide application activities <u>has</u> been authorized through NMFS' issuance of a permit under section 10 of the ESA, and such authorization addresses the effects of the pesticide discharges and discharge-related activities on federally-listed species and federally-designated critical habitat. The term "take" means to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct. See Section 3 of the Endangered Species Act, 16 U.S.C. § 1532(19).

Criterion D. Pesticide application activities were, or will be, discharged to one or more waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP, but only in response to a Declared Pest Emergency Situation. Decision-makers must provide EPA with their rationale supporting the determination whether the discharge is likely to adversely affect *NMFS' Listed Resources of Concern*, including the description of appropriate measures to be undertaken to avoid or eliminate the likelihood of adverse effects.

Criterion E. Pesticide application activities for which permit coverage is being requested in the NOI will discharge to one or more waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP. Eligible discharges include those where the Decision-maker includes in the NOI written correspondence from NMFS that pesticide application activities performed consistent with appropriate measures will avoid or eliminate the likelihood of adverse effects to *NMFS' Listed Resources of Concern*. Eligibility under this criterion is contingent upon the Decision-maker following the measures described in correspondence from NMFS designed to avoid or eliminate the likelihood of adverse effects.

Criterion F. Pesticide application activities for which permit coverage is being requested in the NOI will discharge to one or more waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP. Eligible discharges include those from pesticide application activities that are demonstrated by the Decision-maker as not likely to adversely affect *NMFS' Listed Resources of Concern* or that the pest poses a greater threat to the *NMFS' Listed Resources of Concern* than does the discharge of the pesticide. Decision-makers must provide EPA with their documentation demonstrating the basis for their finding.

3.2.5 Review of Notices of Intent to Discharge

For discharges to those areas with *NMFS' Listed Resources of Concern*, as defined in Appendix A, NMFS will provide EPA with a determination as to whether it believes the eligibility criterion of "not likely to adversely affect ESA-listed species or designated critical habitat" has been met, could be met with conditions that NMFS identifies, or has not been met. EPA expects to rely on NMFS' determination in deciding whether to withhold authorization. If NMFS does not provide EPA with this information within 30 days of EPA posting on the Internet receipt of a complete and accurate NOI, the discharges will be authorized 30 days after EPA posted on the Internet receipt of a complete NOI.

NOI for discharges in response to a Declared Pest Emergency Situation are to be submitted no later than 30 days after beginning discharge. For Declared Pest Emergency Situation in waters of the U.S. containing *NMFS' Listed Resources of Concern*, NMFS will have 30 days after submission of an NOI to provide EPA with a determination as to whether it believes the eligibility criteria of "not likely to adversely affect ESA-listed species or designated critical habitat" has been met, could be met with conditions that NMFS identifies, or has not been met. EPA expects to rely on NMFS' determination in deciding whether to disallow continued permit coverage or if additional conditions are necessary. If NMFS does not provide EPA with a recommendation within 30 days of EPA posting on the Internet receipt of a complete and accurate NOI, authorization for these discharges will continue. If EPA identifies additional permit conditions or prohibitions, or includes additional permit conditions or prohibitions recommended by NMFS, as necessary to qualify discharges for particular Operators as eligible for coverage beyond 60 days under the PGP for the Declared Pest Emergency Situation, those conditions remain in effect for the life of the PGP.

EPA may authorize certain discharges in less than 30 days, but no fewer than 10 days, for any discharges authorized under Criterion B, C, or E for which NMFS has already evaluated the effects of these discharges. In sum, PGP coverage is available only for discharges that are not likely to adversely affect ESA-listed species, except as provided for in a separate ESA section 7 consultation (Criterion B), covered under a permit issued under section 10 of the ESA (Criterion C), or in the event of a declared pest emergency (Criterion D). The PGP does not specify how EPA will evaluate NOIs for accuracy or if the listed resource distribution and application area overlap determinations contained therein are correct.

3.3 Obtaining Authorization

Decision-makers that are required to submit an NOI to discharge pesticide pollutants under PGP that do not discharge to waters where *NMFS' Listed Resources of Concern* occur would be authorized no earlier than 10 days after EPA posts a receipt of a complete and accurate NOI. NOI for discharges in response to a Declared Pest Emergency Situation are to be submitted no later than 30 days after beginning discharge. NOIs are required from the following types of Decision-makers (see Table 1):

Decision-makers exceeding the annual treatment area threshold in Table 1

Decision-makers specifically in the business of pest control;

Decision-makers discharging to Tier 3 waters; and

Decision-makers discharging to waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP.

NOIs are also required to be submitted by public, quasi-public, and private entities with land resource stewardship responsibilities or having as an integral responsibility for controlling pests regardless of the size of the area treated. The specific entities required to submit NOIs, regardless of whether an annual treatment area threshold is exceeded, are:

Any Agency for which pest management for land resource stewardship is an integral part of the organization's operation (e.g., state departments of natural resources and federal agencies such as the U.S. Forest Service)

Mosquito control districts (or similar pest control districts, such as vector control districts)

Irrigation control districts (or other similar public or private entities supplying irrigation waters)

Weed control districts (or other similar special purpose districts created with a responsibility of pest control)

Table 1. Decision-makers required to submit NOIs under the 2016 PGP.

PGP Part/ Pesticide Use	Which Decision-makers Must Submit NOIs?	For Which Pesticide Application Activities?
All four use patterns identified in Part 1.1.1	Any Decision-maker with an eligible discharge to a Tier 3 water (Outstanding National Resource Water) consistent with Part 1.1.2.2	Activities resulting in a discharge to a Tier 3 water
All four use patterns identified in Part 1.1.1	Any Decision-maker with an eligible discharge to waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP	Activities resulting in a discharge to waters of the U.S. containing *NMFS' Listed Resources of Concern*, as defined in Appendix A of the PGP
1.1.1(a) - Mosquito and Other Flying Insect Pest Control	Any Agency for which pest management for land resource stewardship is an integral part of the organization's operations.	All mosquito and other flying insect pest control activities resulting in a discharge to waters of the U.S.
	Mosquito control districts, or similar pest control districts	All mosquito and other flying insect pest control activities resulting in a discharge to waters of the U.S.
	Local governments or other entities that exceed the annual treatment area threshold identified here	Adulticide treatment if more than 6,400 acres during a calendar year
1.1.1(b) - Weed and Algae Pest Control	Any Agency for which pest management for land resource stewardship is an integral part of the organization's operations.	All weed and algae pest control activities resulting in a discharge to waters of the U.S.
	Irrigation and weed control districts, or similar pest control districts	All weed and algae pest control activities resulting in a discharge to waters of the U.S.
	Local governments or other entities that exceed the annual treatment area threshold identified here	Treatment during a calendar year if more than either: 20 linear miles OR 80 acres of water (i.e., surface area)
1.1.1(c) - Animal Pest Control	Any Agency for which pest management for land resource stewardship is an integral part of the organization's operations.	All animal pest control activities resulting in a discharge to waters of the U.S.
	Local governments or other entities that exceed the annual treatment area threshold identified here	Treatment during a calendar year if more than either: 20 linear miles OR 80 acres of water (i.e., surface area)
1.1.1.(d) - Forest Canopy Pest Control	Any Agency for which pest management for land resource stewardship is an integral part of the organization's operations.	All forest canopy pest control activities resulting in a discharge to waters of the U.S.
	Local governments or other entities that exceed the annual treatment area threshold identified here	Treatment if more than 6,400 acres during a calendar year

The annual treatment area for the uses Mosquitoes and Other Flying Insect Pest Control and Forest Canopy Pest control is additive over the calendar year. That is to say, each pesticide application activity is counted as a separate area treated. For example, applying pesticides three times a year to the same 3,000 acre site should be counted as 9,000 acres of treatment area for purposes of determining if such an application exceeds an annual treatment area threshold.

The annual treatment area for Weed and Algae Control and Animal Pest Control is not additive over the calendar year. The treatment area is either the linear extent or the surface area of waters of the U.S. (or at water's edge) treated. For these uses each treatment area is counted only once, regardless of the number of pesticide application activities performed on that area in a given year. Also, treatment of linear features (e.g., a canal or ditch) is measured as the length of the feature regardless of whether treating at water's edge/bank on one side or both sides of that feature.

Certain Operators are automatically covered under the PGP and are not required to submit an NOI. These include Operators who are for-hire applicators, but are not Decision-makers, as defined in Appendix A of the PGP and Decision-makers who apply pesticides to relatively small areas below the defined annual thresholds listed in Table 1. If a Decision-maker who was previously not required to submit an NOI discovers that they will exceed a treatment threshold, that Decision-maker must submit an NOI at least 10 days prior to exceeding the threshold in order to be authorized by the PGP. The 2016 PGP also provides automatic authorization of eligible discharges that result from the application of a pesticide as part of *pesticide research and development*, as defined in Appendix A of the PGP. EPA emphasizes in its BE that even if an NOI is not required, Operators that are covered automatically under the PGP are still subject to all applicable requirements contained within the PGP. This is not explicitly stated in the PGP itself.

3.4 Continuation of the PGP

If the 2016 PGP is not reissued or replaced before its expiration date, it will be administratively continued in accordance with 40 CFR 122.6 and remain in force and effect. If an Operator was authorized to discharge under the PGP before the expiration date, any discharges authorized under the PGP will automatically remain covered by the PGP until:

A Decision-maker is authorized for coverage under a reissued permit or a replacement of the PGP, following the timely and appropriate submittal of a complete NOI requesting authorization to discharge under the new permit and in compliance with the requirements of the NOI;

A Decision-maker submits a Notice of Termination and that notice is processed and posted on the Internet;

An NPDES individual permit for a discharge resulting from application of a pesticide that would otherwise be covered under the PGP is issued or denied;

EPA issues a formal permit decision not to reissue this general permit, at which time EPA will identify a reasonable period for covered dischargers to seek coverage under an alternative NPDES general permit or an NPDES individual permit. Coverage under the PGP will cease when coverage under another permit is granted/authorized; or

EPA has informed the Operator that its discharge is no longer covered under the PGP.

3.5 Alternative Permits

EPA may require, or an Operator may request, that authorization to discharge be applied for and obtained under either an NPDES individual permit or an alternative NPDES general permit. If coverage under an alternative permit is required by EPA, the applicant will be notified in writing with a brief statement of the reasons for the decision, information on what permit to apply for, and, if the Operator is authorized under the PGP, the notice will include a deadline to apply for an alternative permit and will include a statement that on the effective date of the NPDES individual permit, coverage under this general permit will terminate. Operators wanting coverage under an NPDES individual permit must submit an individual permit application with reasons supporting the request to EPA. EPA may issue an NPDES individual permit or authorize the discharges under an alternative NPDES general permit. Authorization to discharge under the PGP is terminated on the effective date of the NPDES individual permit or alternative NPDES general permit.

3.6 Severability

Invalidation of a portion of the PGP will not render the whole permit invalid. EPA's intent is that the PGP will remain in effect to the extent possible; if any part of the PGP is invalidated, the remaining parts of the PGP will remain in effect unless EPA issues a written statement otherwise.

3.7 Technology-Based Effluent Limitations

For the purpose of the PGP, "Operator" is defined to mean any entity associated with the application of pesticides which results in a discharge to waters of the U.S. that meets either of the following two criteria: (1) any entity who performs the application of a pesticide or who has day-to-day control of the application (i.e., they are authorized to direct workers to carry out those activities); or (2) any entity with control over the decision to perform pesticide applications including the ability to modify those decisions. Operators identified in (1) above are referred to in this permit as Applicators while Operators identified in (2) are referred to in this permit as Decision-makers. As defined, more than one Operator may be responsible for complying with this permit for any single discharge from the application of pesticides.

Both Applicators and Decision-makers are required to comply with manufacturer specifications, industry standards and recommended industry practices related to the application of pesticides, relevant legal requirements and other provisions that a prudent Operator would implement to reduce and/or eliminate pesticide discharges to waters of the U.S. Both must use only the amount of pesticide and frequency of pesticide application necessary to control the target pest, using equipment and application procedures appropriate for this task.

Responsibilities of applicators include:

To the extent not determined by the Decision-maker, using only the amount of pesticide and frequency of pesticide application necessary to control the target pest, using equipment and application procedures appropriate for this task.

Maintain pesticide application equipment in proper operating condition, including requirement to calibrate, clean, and repair such equipment and prevent leaks, spills, or other unintended discharges.

Assess weather conditions (e.g. temperature, precipitation, and wind speed) in the treatment area to ensure application is consistent with all applicable federal requirements.

All Decision-makers required to submit an NOI must employ Pest Management Measures to minimize the discharge of pesticide pollutants to waters of the U.S. from the application of pesticides. Prior to their first pesticide application under the PGP and prior to the first pesticide application for each calendar year thereafter, Decision-makers must conduct a problem identification to evaluate the extent and source of the pest problem and to determine the conditions under which pest control will be necessary (i.e., the action threshold[7]). Once the pest problem has been framed, the Decision-maker must consider pest management options: whether to take no action, take action to prevent the need for control, to apply mechanical, physical or cultural methods (as appropriate), to use biological control agents, or to apply pesticides. To determine when the action threshold(s) is met the Decision-maker must assess the pest management area by conducting surveillance of the target pest/life stage in an area that is representative of the pest problem, evaluating existing surveillance data and environmental conditions, or evaluating data from adjacent areas. The Decision-maker must reduce impact on the environment and on non-target organisms by applying the pesticide only when the action threshold(s) has been met.

The Decision-maker must use larvicides as a preferred pesticide for mosquito or flying insect pest control when the larval action threshold(s) has been met. In situations or locations where it is not practical or feasible to achieve effective control through the use of larvicides, the Decision-maker may use adulticides for mosquito or flying insect pest control when the adult action threshold(s) has been met. For pesticide control of forest canopy pests, the Decision-maker must also evaluate the use of pesticides against the most susceptible developmental stage of the pest.

3.8 Water Quality Based Effluent Limits

All Operators must control discharges as necessary to meet applicable state and tribal water quality standards. If at any time the Operator becomes aware that the PGP discharge causes or contributes to a failure to meet such standards, the Operator must take corrective actions.

3.9 Monitoring

During any pesticide application or post-application surveillance of discharges authorized under the PGP, Applicators or all Operators must, when considerations for safety and feasibility allow, visually assess the area to and around where pesticides are applied for possible and observable adverse incidents, as defined in Appendix A, caused by application of pesticides, including the unanticipated death or distress of non-target organisms and disruption of wildlife habitat, recreational or municipal water use.

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[7] Action Threshold: the point at which pest populations or environmental conditions necessitate that pest control action be taken based on economic, human health, aesthetic, or other effects. An action threshold may be based on current and/or past environmental factors that are or have been demonstrated to be conducive to pest emergence and/or growth, as well as past and/or current pest presence. Action thresholds are those conditions that indicate both the need for control actions and the proper timing of such actions.

3.10 Pesticide Discharge Management Plan

A Pesticide Discharge Management Plan (PDMP) documents how a Decision-maker will implement the Technology Based and Water Quality based effluent limitations (including Pest Management Measures), response procedures, and information supporting eligibility considerations under other federal laws. Certain Decision-makers required to submit an NOI are not required to develop a PDMP. These include Decision-makers who are:

working for private enterprises meeting the Small Business Administration size standard (13 CFR 121.201);

working for a local government serving a population of 10,000 or less;

responding to a Declared Pest Emergency Situation; or

submitting an NOI for the sole purpose of obtaining authorization for discharges to waters of the U.S. containing *NMFS' Listed Resources of Concern*.

All other Decision-makers that are required to submit an NOI must develop a PDMP prior to filing their NOI. The PDMP includes the name, contact information, and specific responsibilities for all PDMP team members, one of which should be trained in procedures for stopping, containing, and cleaning up leaks, spills, and other releases to waters of the U.S. Operators who may cause, detect, or respond to a spill or leak must be trained in response procedures and have necessary spill response equipment available. Contact information for state/federal permitting agency, nearest emergency medical facility, and nearest hazardous chemical responder must be in locations that are readily accessible and available.

The PDMP will include the Pest Management Measures (i.e., problem identification and evaluation of pest management options) to be implemented under the PGP. The problem identification describes:

the management area, target pest or pests, pest source or sources, and data used to identify the problem;

the development and planned implementation of action thresholds;

the geographic boundaries of the area to which the plan applies;

the location of the waters of the U.S. affected;

any Tier 3 (Outstanding National Resource Waters), and

any water(s) identified as impaired by a substance which either is an active ingredient or a degradate of such an active ingredient of a pesticide that may be applied under the PGP.

The evaluation of pest management options within the PDMP takes into consideration impacts to water quality and to non-target organisms, feasibility, cost effectiveness, and any relevant previous Pest Management Measures.

The PDMP will also describe spill and adverse incident response procedures that must include procedures for expeditiously stopping, containing, and cleaning up leaks, spills, and other releases to waters of the U.S., procedures for responding to any adverse incident resulting from pesticide applications, and procedures for notifying appropriate personnel within the Decision-maker's agency/organization, emergency response agencies, and regulatory agencies.

Decision-makers will modify their PDMP whenever necessary to address any of the triggering conditions for corrective action in Part 6.1 of the PGP, or when a change in pest control activities significantly changes the type or quantity of pollutants discharged. Changes to the PDMP must be made before the next pesticide application that results in a discharge, if practicable, or if not, no later than 90 days after any change in pesticide application activities.

Decision-makers must retain a copy of the current PDMP, along with all supporting maps and documents, including supporting documentation for their determination with regard to endangered and threatened species and designated critical habitat protection, at the address provided in the NOI. The PDMP and all supporting documents must be readily available, upon request, and copies of any of these documents provided, upon request, to EPA, any state, tribal, or local agency governing discharges or pesticide applications within their respective jurisdictions, and representatives of the Services. Any Confidential Business Information, as defined in 40 CFR Part 2, may be withheld from the public provided that a claim of confidentiality is properly asserted and documented in accordance with 40 CFR Part 2; however, Confidential Business Information must be submitted to EPA, if requested, and may not be withheld from staff within EPA, USFWS, and NMFS cleared for Confidential Business Information review.

3.11 Corrective Action

Pest Management Measures must be reviewed and revised before or, if not practicable, as soon as possible after the next pesticide application if:

An unauthorized release or discharge associated with the application of pesticides (e.g., spill, leak, or discharge not authorized by this or another NPDES permit) occurs.

Any Operator observes or is otherwise made aware of evidence that a person or non-target organism has likely been exposed to a pesticide residue, and has suffered a toxic or adverse effect (i.e., adverse incident).

Operators become aware, or EPA concludes, that Pest Management Measures are not adequate/sufficient for the discharge to meet applicable water quality standards.

Any monitoring activities indicate failure to meet applicable technology-based effluent limitations.

An inspection or evaluation of activities by an EPA official, or local, state, or tribal entity, reveals that modifications to the Pest Management Measures are necessary to meet the effluent limitations in the PGP.

Unauthorized releases and adverse incidents require specific notification and documentation by the Operator. Other events triggering corrective action require documentation within 30 days of discovery. Documentation must include the date of discovery, how the problem was identified, the condition triggering the need for corrective action, any water quality monitoring data used in identifying that condition, a summary of the corrective action taken or planned, including initiation date and anticipated completion date, measures taken to prevent recurrence of the problem, and whether the modifications to the PDMP were required.

3.12 Unauthorized Release or Discharges

An Operator must notify the National Response Center immediately upon becoming aware of a leak, spill, or other release into waters of the U.S. containing a hazardous substance or oil in an amount equal to or in excess of a reportable quantity established under either 40 CFR Part 110, 40 CFR Part 117, or 40 CFR Part 302 occurs over any 24-hour period. State or local requirements may necessitate also reporting spills or leaks to local emergency response, public health, or drinking water supply agencies. If the leak results in an adverse incident, adverse incident reporting is required (see section 3.13 below). If an adverse incident did not result from the unauthorized discharge, within 30 days of becoming aware of the release the Operator must document and retain information on the corrective action taken or planned, expected initiation and completion of corrective actions, measures to prevent recurrence, and whether modifications to the PDMP are required.

3.13 Adverse Incidents

The phrase “toxic or adverse effect” includes effects on non-target plants, fish, or wildlife that are unusual or unexpected as a result of exposure to a pesticide residue, and may include observation of dead, immobile, or nonresponsive non-target aquatic organisms, abnormal or erratic movement by non-target aquatic organisms, or stunting, wilting, or desiccation of non-target submerged or emergent aquatic plants. An Operator must immediately notify the appropriate EPA Incident Reporting Contact of any adverse incident within 24 hours of becoming aware of the incident. If the adverse incident has affected ESA-listed species or designated critical habitat, the Operator must also notify NMFS in the case of incidents involving ESA-listed anadromous or marine species or designated critical habitat, or USFWS in the case of incidents involving ESA-listed terrestrial or freshwater species or designated critical habitat.

Adverse incident reporting under the PGP are in addition to (i.e., do not replace) reporting requirements under FIFRA section 6(a)(2) and its implementing regulations at 40 CFR Part 159. Under the PGP, notification must include contact and permit identification, a description of the activity and parties contributing to the adverse incident, a description of how the incident was detected, and the response measures taken or planned. When the incident involved ESA-listed species, the Operator must also identify the species affected. The Operator is required provide a written report to the appropriate EPA Regional office within 30 days of the initial notification. The report must include the information provided by the Operator when EPA was initially notified along with:

Date and time the Operator contacted EPA notifying the Agency of the adverse incident, who the Operator spoke with at EPA, and any instructions received from EPA;

Location of incident, including the names of any waters affected and appearance of those waters (sheen, color, clarity, etc.);

A description of the circumstances of the adverse incident including species affected, estimated number of individual and approximate size of dead or distressed organisms;

Magnitude and scope of the affected area;

Pesticide application rate; intended use site (e.g., on the bank, above waters, or directly to water); method of application; and the name of pesticide product and EPA registration number;

Description of the habitat and the circumstances under which the adverse incident occurred (including any available ambient water data for pesticides applied);

If laboratory tests were performed, an indication of which test(s) were performed, and when; additionally, a summary of the test results must be provided within 5 days after they become available, if not available at the time of submission of the 30-day report; and

Description of actions to be taken to prevent recurrence of adverse incidents.

Where multiple Operators are authorized for a discharge that results in an adverse incident, notification and reporting by any one of the Operators constitutes compliance for all of the Operators, provided a copy of the required written report is also provided to all of the other authorized Operators within 30 days of the reportable adverse incident.

Incidents that require revision of Pesticide Management Measures may be a violation of the PGP. Corrective action does not absolve liability for any violation and failure to make changes to the Pesticide Management Measures in a timely fashion constitutes an additional permit violation. EPA will consider the appropriateness and promptness of corrective action in determining enforcement responses to permit violations and may impose additional requirements and schedules of compliance, including requirements to submit additional information concerning the condition(s) triggering corrective action or schedules and requirements more stringent than specified in the PGP.

Adverse incident reporting is not required when the Operator is aware of facts indicating the adverse incident was not related to toxic effects or exposure from the pesticide application, has been notified by EPA that the reporting requirement has been waived for this incident or category of incidents (such notification must be retained), receives information of an adverse incident that is clearly erroneous, or the incident occurs to pests that are similar in kind to potential target pests identified on the FIFRA label.

3.14 Record Keeping and Annual Reporting

All Operators must retain any records required under the PGP for at least 3 years after the Operator's coverage expires or is terminated. Required records must be documented as soon as possible but no later than 14 days following completion of each pesticide application. Operators must make available to EPA, including an authorized representative of EPA, all records kept under the PGP, upon request, and provide copies of such records, upon request.

3.14.1 Records Required of All Operators

Adverse Incident Reports

If any incidents were identified but determined not to be reportable, the rationale for making that determination must also be retained as a record.

Corrective action documentation

Spill, leak, or other unpermitted discharge documentation

Documentation for each treatment area to which pesticides are discharged, including:

A description of treatment area, its location and size and identification of any waters of the U.S. to which pesticide(s) are discharged;

The pesticide use pattern(s);

Target pest(s);

The name and EPA registration number of each pesticide product applied to each treatment area;

Quantity of each pesticide product applied to each treatment area;

Pesticide application date(s);

Equipment calibration documentation (by Applicators and Decision-makers that are also Applicators); and

Whether or not visual monitoring was conducted during pesticide application and/or post-application.

If visual monitoring was not conducted, records must indicate why monitoring did not occur.

If monitoring did occur, records must describe any possible or observable adverse incidents caused by application of pesticides.

3.14.2 Records Required of All Decision-makers

Any Decision-maker required to submit an NOI must retain:

A copy of the NOI;

A copy of any correspondence with EPA specific to coverage under the PGP;

A copy of the EPA acknowledgment letter with the assigned permit tracking number;

Records containing the names and contact information of companies hired to apply pesticides;

An explanation for the need to control target pests; and

A description of the pest management measures implemented prior to the first pesticide application.

3.14.3 Records Required of Decision-makers for a Large Entity

Decision-makers that submitted an NOI for a large entity[8] must also retain:

Copies of annual reports submitted to EPA;

A copy of the PDMP documenting:

the action threshold(s) derived for pest management measures;

the method(s) and/or data used to determine when an action threshold(s) has been met; and

any modifications made to the PDMP during the term of the 2016 PGP.

3.15 Annual Reporting Requirements

Decision-makers who submit an NOI for any large entity and those submitting an NOI for small entities[9] making discharges to waters of the U.S. Containing *NMFS' Listed Resources of Concern* are required to submit an annual report to EPA for each year of coverage under the

[8] Large Entity: Any public entity that serves a population greater than 10,000 or private enterprise that exceeds the Small Business Administration size standard identified at 13 CFR 121.201

[9] Small Entity: Any public entity that serves a population less than 10,000 or private enterprise that meets the Small Business Administration size standard identified at 13 CFR 121.201

PGP. If a Decision-maker's obligation to submit an annual report changes at some point during a calendar year (i.e., ceases due to permit termination or, for large entities, is triggered by exceeding an annual treatment area threshold), the Decision-maker must submit an annual report for that portion of the year during which the entity was covered under the PGP and required to provide annual reporting. Once a Decision-maker meets the obligation to submit an annual report, the Decision-maker must submit the annual report each calendar year thereafter for the duration of coverage under this general permit, whether or not the Decision-maker has discharges from the application of pesticides in any subsequent calendar year.

The annual report must contain the following information:

Name and contact information for the Decision-maker and any other contact person;

NPDES permit tracking number(s);

For each area treated in that year:

Description of treatment area, including location and size (acres or linear feet) of treatment area and identification of any waters of the U.S., either by name or by location, to which pesticide(s) are discharged;

Pesticide use pattern(s) (i.e., mosquito and other flying insects, weed and algae, animal pest, or forest canopy) and target pest(s);

Company name(s) and contact information for pesticide applicator(s), if different from the Decision-maker;

Total amount of each pesticide product applied for the reporting year by the EPA registration number(s) and by application method (e.g., aerially by fixed-wing or rotary aircraft, broadcast spray, etc.);

If the Annual Report is from a large entity, the report must indicate whether this pest control activity was addressed in the PDMP prior to pesticide application;

If the Annual Report is from a small entity, the report must indicate the dates of pesticide application;

If applicable, any adverse incidents as a result of these treatment(s), as described in Part 6.4.1; and

If applicable, description of any corrective action(s), including spill responses, resulting from pesticide application activities and the rationale for such action(s).

3.16 Standard Permit Conditions

The PGP includes an appendix explaining permit holders duty to comply with permit provisions that outlined the administrative penalties, civil penalties, and criminal penalties for negligent violations, intentional violations or endangerment, and the making of false statements. This appendix also includes the following permit holder requirements:

Reapply for coverage if discharge activities are to continue after the PGP has expired.

Take all reasonable steps to minimize or prevent any discharge in violation of the PGP, which has a reasonable likelihood of adversely affecting human health or the environment.

Properly operate and maintain all facilities and systems of treatment and controls to achieve compliance with the conditions of the PGP.

Provide EPA or an authorized representative any information or access for inspection that EPA may request to determine whether cause exists for modifying, revoking and reissuing, terminating coverage, or to determine compliance with the PGP.

Retain records of all reports required by the PGP, and records of all data used to complete the NOI for the PGP, for a period of at least 3 years from the date the PGP expires or the date the Operator's authorization is terminated. That period may be extended by request of EPA at any time.

Comply with all signatory and reporting requirements of the PGP.

4 Interrelated and Interdependent Actions

NMFS must consider interrelated and interdependent actions of the proposed action. Interdependent actions are actions having no independent utility apart from the proposed action [50 CFR §402-02]. They are typically a consequence of the proposed action. For example, if our consultation were evaluating the effects of building a road, an interdependent action would be the planned construction of homes and other structures that would not be accessible without the presence of that road. Interrelated actions are actions that are part of a larger action and depend on the larger action for their justification [50 CFR §402-02]. They are actions that are typically associated with the proposed action. In the case of the PGP, no chemical pesticide residue can be discharged without a discharge of a chemical pesticide. NMFS therefore includes discharges of all pesticides, whether or not included in the PGP definition of pesticide pollutants, as interrelated actions and assesses the effects from these charges as part of the effects of the action. In this opinion, we will use the term "pesticide" or "pesticides" to refer to all pesticides, and use the term "pesticide pollutant" only when referring specifically to the terms of the PGP.

5 Action Area

The action area for this consultation consists of all waters of the U.S. in states, territories, and possessions receiving discharges authorized by EPA under the PGP. Because NMFS only has jurisdiction over marine, estuarine, and anadromous endangered and threatened species and designated critical habitat for those species, this consultation addresses the potential effects of PGP-authorized discharges to waters of the U.S. occurring in coastal areas and inland waters used by ESA-listed marine, estuarine, and anadromous species under NMFS' jurisdiction where EPA has permitting authority. This includes the entire states of Massachusetts, New Hampshire, and Idaho, the District of Columbia, Puerto Rico, the Pacific territories, federally operated facilities in Washington and Delaware, and Indian country lands[10] nationwide. At this time, waters of the U.S. are defined as (40 CFR 122.2):

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[10] The term "Indian country" means: (a) all land within the limits of any Indian reservation under the jurisdiction of the United States Government, notwithstanding the issuance of any patent,

- All waters which are currently used, were used in the past, or may be susceptible to use in interstate or foreign commerce, including all waters which are subject to the ebb and flow of the tide and all interstate waters, including interstate "wetlands."
- All other waters such as intrastate lakes, rivers, streams (including intermittent streams), mudflats, sandflats, "wetlands," sloughs, prairie potholes, wet meadows, playa lakes, or natural ponds the use, degradation, or destruction of which would affect or could affect interstate or foreign commerce including any such waters:
    - Which are or could be used by interstate or foreign travelers for recreational or other purposes;
    - From which fish or shellfish are or could be taken and sold in interstate or foreign commerce; or
    - Which are used or could be used for industrial purposes by industries in interstate commerce.
- All impoundments of waters otherwise defined as waters of the U.S. under this definition.
- Tributaries of those waters described above.
- The territorial sea.
- "Wetlands" adjacent to waters (other than waters that are themselves wetlands).
- Waters of the U.S. extend to the outer reach of the three mile territorial sea, defined in section 502(8) of the Clean Water Act as the belt of the seas measured from the line of ordinary low water along that portion of the coast which is in direct contact with the open sea and the line marking the seaward limit of inland waters, and extending seaward a distance of three miles.

Although degradates and metabolites of some of the pesticide considered in this opinion might be transported more than three miles from our coastline at some concentration, the data we would need to follow a pesticide as it is transported from a particular application site to reservoirs in coastal waters and the open ocean are not available to us. Similarly, the data we would need to trace pesticides found in the tissues of marine and coastal animals back to particular terrestrial applications are not available. Without some data or other information, we can only acknowledge the probability of this kind of transport in our opinion; we do not extend the *Action Area* more than three miles from the coast of the coastal states, territories, and possessions included in the proposed PGP.

While EPA has permitting authority on Federal and Indian lands in certain states, some of these areas were excluded from designated critical habitat designations for reasons of national defense or in support of U.S.-tribal relationships. Effects within these areas are included in the *Action Area* for this opinion with respect to jeopardy determinations (i.e., effects to the species), but

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and, including rights-of-way running through the reservation; (b) all dependent Indian communities within the borders of the United States whether within the original or subsequently acquired territory thereof, and whether within or without the limits of a state; and(c) all Indian allotments, the Indian titles to which have not been extinguished, including rights-of-way running through the same (18 USC 1151).

cannot be considered in adverse modification determinations for designated critical habitat. However, the effects of discharges originating from excluded areas on *adjacent* designated critical habitat are considered in adverse modification determinations. For example, EPA has NPDES permitting authority for Indian country lands in California. Designated critical habitat for the southern Distinct Population Segment (DPS) of Pacific eulachon occurs on the Klamath River in California (76 FR 65323, October 20, 2011). The portion of the Klamath River which flows through the Yurok Reservation is excluded from the designated critical habitat. Accordingly, jeopardy determinations would consider effects of PGP discharges to the species over the extent of the Klamath River while adverse modification determinations would only consider effects to designated critical habitat elements essential to the conservation of the species on that portion of the Klamath River designated as critical habitat (i.e., not within the Yurok Reservation).

The action area for this opinion encompasses 3,935 sub-watersheds within 363 thousand square kilometers (approximately 140 thousand square miles) dispersed over 18 states and territories. Among these, 161 sub-watersheds discharge directly to bays or the ocean where ESA-listed species and designated critical habitat under NMFS' jurisdiction may occur.

The distribution of sub-watersheds subject to PGP discharges are illustrated in Figure 1 and Figure 2 for the East and West Coasts, respectively. Waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction may occur in Puerto Rico are shown in Figure 3. The entire extent of waters in Puerto Rico and the Pacific Territories are subject to PGP-authorized discharges. According to the National Atlas, Indian Country Lands in Alaska include only the Annette Islands off South East Alaska

Table 2. Extent of the action area EPA has permitting authority for the PGP and where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur.

	All Watersheds			Coastal Watersheds		
State or Territory	**# Sub-watersheds**	**Acres**	**km^2**	**# Sub-watersheds**	**Acres**	**km^2**
East Coast						
Connecticut Indian Country Lands	2	35,363	143	1	17,182	70
Rhode Island Indian Country Lands	3	68,090	276	1	18,017	73
District of Columbia	6	161,124	652			
Delaware Federally Operated Facilities	9	185,536	751	5	119,489	484
Massachusetts	226	5,205,997	21,079	26	544,455	2,204
Maine Indian Country Lands	13	329,342	1,333	2	87,435	354
New Hampshire	334	7,390,815	29,910	9	241,779	978
Caribbean						
Puerto Rico	219	2,206,073	8,928	52	433,246	1,753
West Coast						
Alaska Indian Country Lands	7	317,423	1,285	6	307,605	1,245
California Indian Country Lands	201	6,349,845	25,697	11	478,323	1,936
Idaho	2,573	56,696,234	229,446	5	89,294	361
Oregon Indian Country Lands	77	1,695,526	6,862	1	6,816	28
Washington Indian Country Lands and Federally Operated Facilities	246	8,162,553	33,033	27	576,597	2,331
Pacific Territories						
American Samoa	4	317,694	1,286	2	48,393	196
Guam	10	364,856	1,477	9	134,470	544
Northern Marianas	5	152,049	615	4	29,451	119

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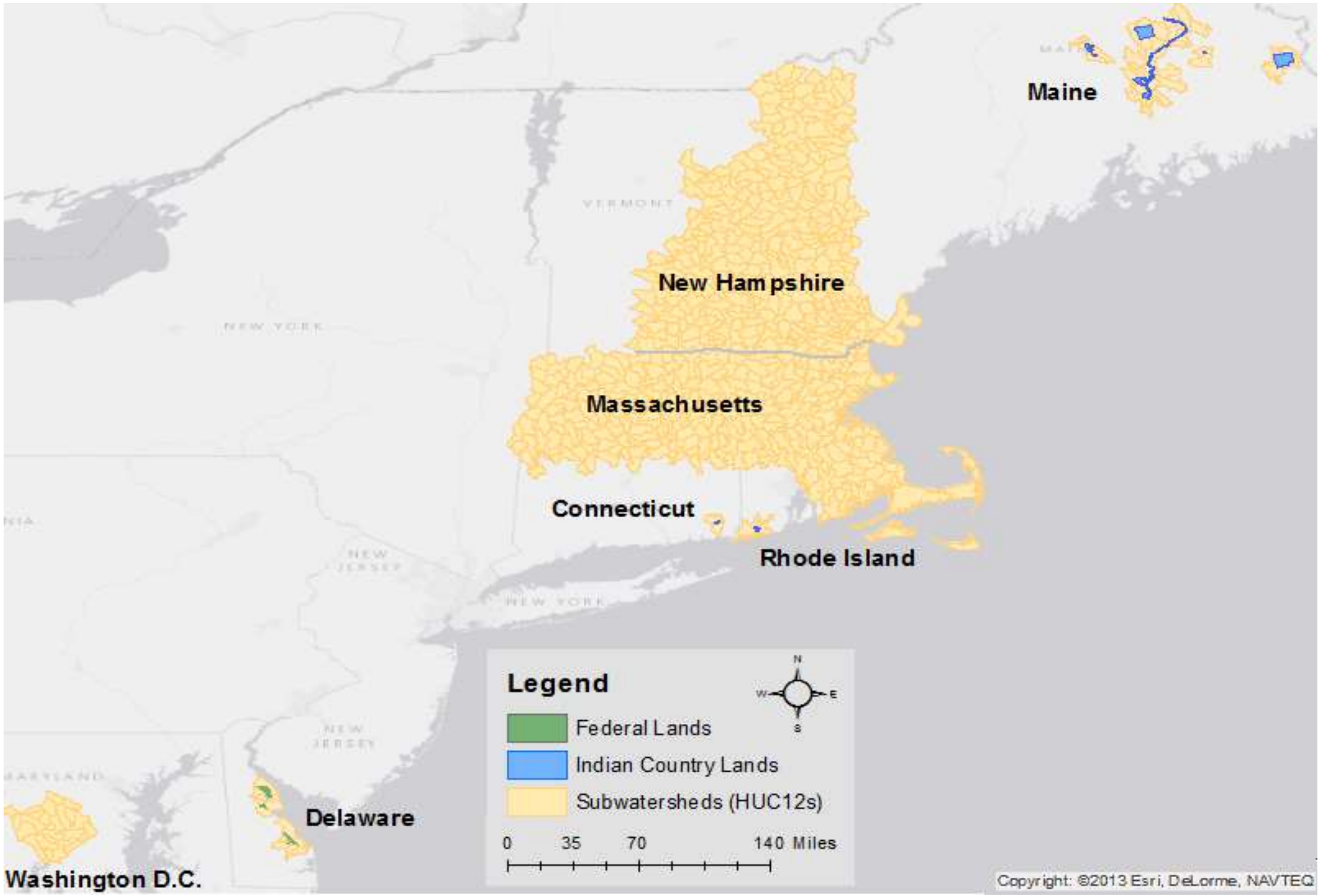

Figure 1. Map of east coast lands and sub-watersheds subject to PGP-authorized discharges in the states of Massachusetts and New Hampshire, the District of Columbia, Federal Facilities in Delaware, and Indian Country Lands in Maine, Connecticut and Rhode Island.

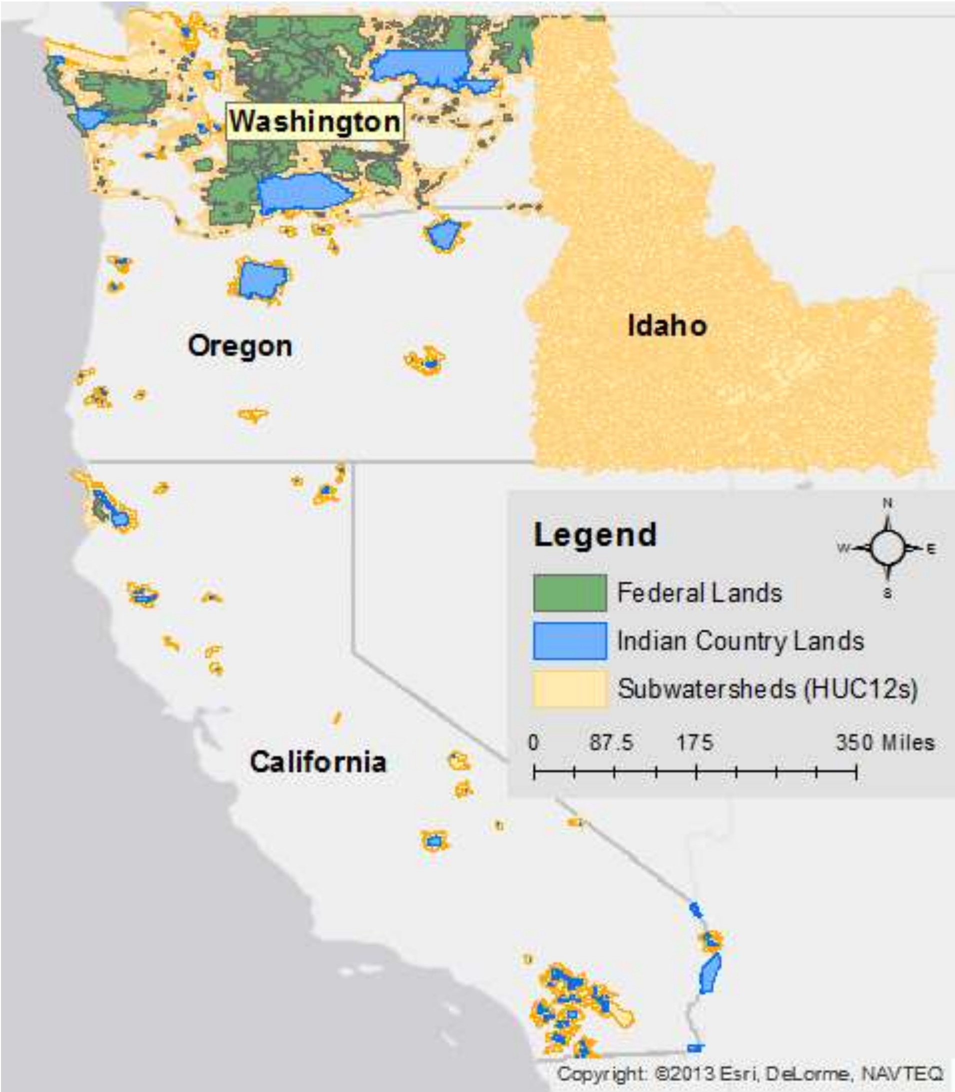

Figure 2. Map of West Coast lands and sub-watersheds subject to PGP-authorized discharges within the State of Idaho and Indian Country Lands in California, Oregon, or Washington, or Located on Federal lands in Washington.

Figure 3. Map of coastal waters of Puerto Rico subject to PGP-authorized discharges where ESA-listed species under NMFS' jurisdiction may occur.

6 STATUS OF THE SPECIES AND DESIGNATED CRITICAL HABITAT

As described in Section 2, during the consultation we identify those endangered or threatened species or designated critical habitat that may be affected by the proposed action. In order for a proposed action to be determined to not likely adversely affect species or designated critical habitat, all of the effects of that action must be expected to be discountable, insignificant, or completely beneficial. Discountable effects are those that are extremely unlikely to occur. Insignificant effects relate to the size of the impact and should never reach the scale where take occurs. Beneficial effects are contemporaneous positive effects without any adverse effects to the species or designated critical habitat.

6.1 Species and Critical Habitat Not Likely to be Adversely Affected

For this opinion, we determined that exposures to pesticide discharges authorized under EPA's PGP would be extremely unlikely for those species that do not frequent coastal waters where EPA has permitting authority (i.e., effects would be discountable). Therefore, EPA's PGP is not likely to adversely affect the following species:

blue whale (*Balaenoptera musculus*, endangered)

false killer whale (*Pseudorca crassidens*, endangered)

fin whale (*Balaenoptera physalus*, endangered)

sei whale (*Balaenoptera borealis*, endangered)

sperm whale (*Physeter macrocephalus*, endangered)

- Humpback Whale (*Megaptera novaeangliae*, endangered)
- North Atlantic Right Whale (*Eubalaena glacialis*) and designated critical habitat (endangered)
- Scalloped Hammerhead (*Sphyrna lewini*) Eastern Pacific DPS (endangered)
- Scalloped Hammerhead (*Sphyrna lewini*) Central and Southwest Atlantic DPS (endangered)

The EPA is the permitting authority on Indian Country lands within range of Gulf sturgeon (threatened) and smalltooth sawfish (endangered), but these lands are inland. While these species may be exposed to PGP-authorized discharges, such exposures are expected to be insignificant given the dissipation and degradation that would occur before reaching the waters they occupy. EPA does not have permitting authority in waters where white and black abalone (both endangered) occur or where the Carolina DPS and south Atlantic DPS of Atlantic sturgeon (both endangered) occur. For these species, exposures to pesticide discharges authorized under the PGP are extremely unlikely (i.e., effects would be discountable), therefore EPA's PGP is not likely to adversely affect these species.

6.2 Species and Designated Critical Habitat Considered in this Opinion

The ESA-listed species and designated critical habitats which occur within the action area that fall under NMFS' jurisdiction and may be exposed to the pesticide discharges and experience direct or indirect effects of those exposures are identified in Table 3 and Table 4.

Table 3. NMFS endangered and threatened species and designated critical habitat considered in this opinion.

Species	ESA Status	Designated Critical Habitat	Recovery Plan
Marine Mammals – Cetaceans			
Southern Resident Killer Whale *(Orcinus orca)*	E – 70 FR 69903	71 FR 69054	73 FR 4176
Salmonids			
salmon, Chinook (*Oncorhynchus tshawytscha*)			
- California coastal	T – 64 FR 50393	70 FR 52488	– –
- Central Valley spring-run	T – 64 FR 50393	70 FR 52488	79 FR 42504
- Lower Columbia River	T – 64 FR 14308	70 FR 52630	78 FR 41911
- Upper Columbia River spring-run	E – 64 FR 14308	70 FR 52630	72 FR 57303
- Puget Sound	T – 64 FR 14308	70 FR 52630	72 FR 2493
- Sacramento River winter-run	E – 59 FR 440	58 FR 33212	79 FR 42504
- Snake River fall-run	T – 59 FR 42529	58 FR 68543	– –
- Snake River spring/summer-run	T – 59 FR 42529	64 FR 57399	– –
- Upper Willamette River	T – 64 FR 14308	70 FR 52630	76 FR 52317b
salmon, chum (*Oncorhynchus keta*)			
- Columbia River	T – 64 FR 14507	70 FR 52630	78 FR 41911
- Hood Canal summer-run	T – 64 FR 14507	70 FR 52630	72 FR 29121
salmon, coho (*Oncorhynchus kisutch*)			
- Central California coast	E – 61 FR 56138	65 FR 7764	– –
- Oregon coast	T – 63 FR 42587	73 FR 7816	78 FR 41911
- Southern Oregon & Northern California coasts	T – 62 FR 24588	64 FR 24049	– –
- Lower Columbia River	T – 70 FR 37160	81 FR 9251	78 FR 41911
salmon, sockeye (*Oncorhynchus nerka*)			
- Ozette Lake	T – 64 FR 14528	70 FR 52630	74 FR 24706
- Snake River	E – 56 FR 58619	58 FR 68543	– –
trout, steelhead (*Oncorhynchus mykiss*)			
- California Central Valley	T – 71 FR 834	70 FR 52488	79 FR 42504
- Central California coast	T – 71 FR 834	70 FR 52488	– –
- South-Central California coast	T – 71 FR 834	70 FR 52488	– –
- Southern California	E – 71 FR 834	70 FR 52488	– –
- Northern California	T – 71 FR 834	70 FR 52488	– –
- Lower Columbia River	T – 71 FR 834	70 FR 52630	74 FR 50165
- Middle Columbia River	T – 71 FR 834	70 FR 52630	– –
- Upper Columbia River	T – 74 FR 42605	70 FR 52630	72 FR 57303
- Upper Willamette River	T – 71 FR 834	70 FR 52630	76 FR 52317b
- Snake River Basin	T – 71 FR 834	70 FR 52630	– –
- Puget Sound	T – 72 FR 26722	81 FR 9251	– –
Atlantic Salmon (*Salmo salar*)	E – 74 FR 29344	74 FR 29300	70 R 75473

Species	ESA Status	Designated Critical Habitat	Recovery Plan
- Gulf of Maine DPS			
Non-Salmonid Anadromous Species			
Eulachon (*Thaleichthys pacificus*)	T – 75 FR 13012	76 FR 65323	– –
Shortnose sturgeon (*Acipenser brevirostrum*)	E – 32 FR 4001	– –	63 FR 69613
Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) - Gulf of Maine DPS	T – 77 FR 5880	81 FR 35701 (Proposed)	– –
- New York Bight DPS - Chesapeake Bay DPS	E - 77 FR 5880		
Green sturgeon, (*Acipenser medirostris*) - Southern DPS	T – 71 FR 17757	74 FR 52300	– –
Marine Fish			
Bocaccio *(Sebastes paucispinis)*	E – 75 FR 22276	79 FR 68041	– –
Yellow Eye Rockfish (*Sebastes ruberrimus*)	T – 75 FR 22276	79 FR 68041	– –
Nassau Grouper	T – 79 FR 51929		
Sea Turtles			
Green Turtle *(Chelonia mydas) – North Atlantic DPS*	E – 43 FR 32800	63 FR 46693	63 FR 28359
Hawksbill Turtle *(Eretmochelys imbricata*)	E – 35 FR 8491	63 FR 46693	57 FR 38818
Kemp's Ridley Turtle *(Lepidochelys kempii)*	E – 35 FR 18319	– –	75 FR 2496
Olive Ridley Turtle (*Lepidochelys olivacea*) Pacific Coast of Mexico breeding populations all other populations	E – 43 FR 32800 T – 43 FR 32800	– –	63 FR 28359
Leatherback Turtle (*Dermochelys coriacea*)	E – 35 FR 8491	44 FR 17710	63 FR 28359
Loggerhead Turtle (*Caretta carettaCaretta caretta*) - Northwest Atlantic and North Pacific DPS	E – 76 FR 58868	79 FR 39856	63 FR 28359
Corals			
Elkhorn Coral (*Acropora palmata*) Staghorn Coral (*Acropora cervicornis*)	T – 71 FR 26852	73 FR 72210	80 FR 12146

Species	ESA Status	Designated Critical Habitat	Recovery Plan
Coral Species - *Mycetophyllia ferox* - The *Orbicella:* *O.faveolata* O. franksi *O. annularis* - Pillar *(Dendrogyra cylindrus)* *- The Acropora* *A. globiceps A. jacquelineae* *A. lokani A. pharaonis* *A. retusa A. rudis* *A. speciosa A. tenella* *- Anacropora spinosa* *- Euphyllia paradivisa* *- Isopora crateriformis* *- Montipora australiensis* *- Pavona diffluens* *- Porites napopora* *- Seriatopora aculeata*	T – 79 FR 54122	– –	– –

Table 4. Physical and biological features of designated critical habitat that are essential to the conservation of the species. Water quality and biological features which may be affected by toxicants are in boldface.

Species DPS or Evolutionarily Significant Unit (ESU)	**Physical or Biological Features Essential for the Conservation of the Species**
Invertebrates	
Elkhorn Coral & Staghorn Coral	Substrate of suitable quality and availability to support successful larval settlement and recruitment, and reattachment and recruitment of fragments
Reptiles	
Green Turtle Florida & Mexico Pacific coast breeding colonies; all other areas Hawksbill Turtle	Activities requiring special management considerations include: • Vessel traffic • Coastal construction • Point and non-point source pollution • Fishing activities • Dredge and fill activities • Habitat restoration
Leatherback Turtle	• Activities identified as modifying CH include: recreational boating ◦ swimming, ◦ sandmining (see 77 FR 32909 for the 6/4/2012 determination on Sierra Club's petition to revise the CH) • **Prey species, primarily Scyphomedusae (*Chrysaora*, *Aurelia*, *Phacellophora*, and *Cyanea*) of sufficient condition, distribution, diversity, and abundance to support individual as well as population growth, reproduction, and development** • Migratory pathway conditions to allow for safe and timely passage and access to/from/within high use foraging areas
Marine Mammals	
Killer Whale - Southern Resident	• **Water quality to support growth and development**; • **Prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth**; and • Passage conditions to allow for migration, resting, and foraging.
Marine and anadromous fish other than Pacific salmonids	
Green Sturgeon - Southern	**Freshwater areas:** • **Abundant prey items for larval, juvenile, subadult, and adult life stages.** • Substrate type or size (i.e., structural features of substrates) • A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival of all life stages. • **Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages.** • A migratory pathway necessary for the safe and timely passage of Southern DPS fish within riverine habitats and between riverine and estuarine habitats (e.g., an unobstructed river or dammed river that still allows for safe and timely passage). • Deep (≥5 m) holding pools for both upstream and downstream holding of adult or subadult fish, with adequate water quality and flow to maintain the physiological needs of the holding adult or subadult fish. • **Sediment quality (i.e., chemical characteristics) necessary for normal behavior, growth, and viability of all life stages**. **Estuarine areas:** • **Abundant prey items within estuarine habitats and substrates for juvenile,**

Species DPS or Evolutionarily Significant Unit (ESU)	Physical or Biological Features Essential for the Conservation of the Species
	subadult, and adult life stages. • Within bays and estuaries adjacent to the Sacramento River (i.e., the Sacramento-San Joaquin Delta and the Suisun, San Pablo, and San Francisco bays), sufficient flow into the bay and estuary to allow adults to successfully orient to the incoming flow and migrate upstream to spawning grounds. • **Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages.** • A migratory pathway necessary for the safe and timely passage of Southern DPS fish within estuarine habitats and between estuarine and riverine or marine habitats. • A diversity of water depths necessary for shelter, foraging, and migration of juvenile, subadult, and adult life stages. • **Sediment quality (i.e., chemical characteristics) necessary for normal behavior, growth, and viability of all life stages. This includes sediments free of elevated levels of contaminants** **Coastal Marine Areas:** • A migratory pathway necessary for the safe and timely passage of Southern DPS fish within marine and between estuarine and marine habitats. • Coastal marine waters with adequate dissolved oxygen levels and **acceptably low levels of contaminants (e.g., pesticides, PAHs, heavy metals that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon).** • **Abundant prey items for subadults and adults, which may include benthic invertebrates and fish.**
Atlantic sturgeon - Gulf of Maine - New York Bight - Chesapeake Bay	• Hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0 to 0.5 parts per thousand range) for settlement of fertilized eggs, refuge, growth, and development of early life stages • Aquatic habitat with a gradual downstream salinity gradient of 0.5 to 30 parts per thousand and soft substrate (e.g., sand, mud) downstream of spawning sites for juvenile foraging and physiological development • Water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support: (1) Unimpeded movement of adults to and from spawning sites; (2) seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and (3) staging, resting, or holding of subadults or spawning condition adults. Water depths in main river channels must also be deep enough (e.g., ≥1.2 m) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river • Water, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support: (1) Spawning; (2) annual and interannual adult, subadult, larval, and juvenile survival; and (3) larval, juvenile, and subadult growth, development, and recruitment (e.g., 13 °C to 26 °C for spawning habitat and no more than 30° C for juvenile rearing habitat, and 6 mg/L dissolved oxygen for juvenile rearing habitat)
Eulachon - Southern	• Freshwater spawning and incubation sites with water flow, quality and temperature conditions and substrate supporting spawning and incubation, and with migratory access for adults and juveniles. • A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of freshwater discharge over time) that supports spawning, and survival of all life stages. • **Water quality suitable for spawning and viability of all eulachon life stages. Sublethal concentrations of contaminants affect the survival of aquatic species**

Species DPS or Evolutionarily Significant Unit (ESU)	Physical or Biological Features Essential for the Conservation of the Species
	by increasing stress, predisposing organisms to disease, delaying development, and disrupting physiological processes, including reproduction. • Suitable water temperatures, within natural ranges, in eulachon spawning reaches. • Spawning substrates for eulachon egg deposition and development. • Freshwater and estuarine migration corridors associated with spawning and incubation sites that are free of obstruction and with water flow, quality and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted. • Safe and unobstructed migratory pathways for eulachon adults to pass from the ocean through estuarine areas to riverine habitats in order to spawn, and for larval eulachon to access rearing habitats within the estuaries and juvenile and adults to access habitats in the ocean. • A flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of freshwater discharge over time) that supports spawning migration and outmigration of larval eulachon from spawning sites. • **Water quality suitable for survival and migration of spawning adults and larval eulachon.** • Water temperature suitable for survival and migration. • **Prey resources to support larval eulachon survival.** • **Nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival.** • **Prey items, in a concentration that supports foraging leading to adequate growth and reproductive development for juveniles and adults in the marine environment.** • **Water quality suitable for adequate growth and reproductive development.**
Puget Sound / Georgia Basin Rockfish species Yelloweye Boccacio	**Adults** • **Quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities,** • **water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities**, and • the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance. **Juvenile boccacio** • **Quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities; and** • **water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities**.
Pacific Salmonids	
Chum Salmon - Columbia River - Hood Canal summer run Sockeye - Lake Ozette Chinook Salmon - Puget Sound - Lower Columbia River - Upper Willamette River Steelhead - Upper Columbia River - Snake River	• Freshwater spawning sites with **water quantity and quality conditions and substrate supporting spawning, incubation and larval development**; • Freshwater rearing sites with: ◦ Water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; ◦ **Water quality and forage supporting juvenile development**; ◦ Natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, **aquatic vegetation**, large rocks and boulders, side channels, and undercut banks. ◦ Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic **vegetation**, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival;

Species **DPS or Evolutionarily Significant Unit (ESU)**	**Physical or Biological Features Essential for the Conservation of the Species**
- Middle Columbia River - Upper Willamette River - Lower Columbia River - Puget Sound Coho Salmon - Lower Columbia River	• Estuarine areas free of obstruction and excessive predation with: ◦ **Water quality**, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh & saltwater; ◦ Natural cover such as **submerged and overhanging large wood, aquatic vegetation**, large rocks and boulders, side channels; ◦ **Juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.** • Nearshore marine areas free of obstruction and excessive predation with: ◦ **Water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation**; and ◦ Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels. • **Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation**.
Coho Salmon - Central California Coast - Southern Oregon/Northern California Coast	Within the range of both ESUs, the species' life cycle can be separated into 5 essential habitat types: • juvenile summer and winter rearing areas; • juvenile migration corridors; • areas for growth and development to adulthood; • adult migration corridors; and • spawning areas. Essential features of coho designated critical habitat include adequate: Substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, **food, riparian vegetation,** space, safe passage
Steelhead - Puget Sound Coho Salmon - Lower Columbia River	• Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development. • Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; **water quality and forage supporting juvenile development**; and natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, **aquatic vegetation,** large rocks and boulders, side channels, and undercut banks. • Freshwater migration corridors free of obstruction with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, **aquatic vegetation,** large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival. • Estuarine areas free of obstruction with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, **aquatic vegetation,** large rocks and boulders, and side channels; and **juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.** • Nearshore marine areas free of obstruction with **water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation**; and natural cover such as submerged and overhanging large wood, **aquatic vegetation**, large rocks and boulders, and side channels. • Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.
Coho Salmon - Oregon Coast	• Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation, and larval development. • Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; **water quality and forage supporting juvenile development;** and natural cover such as

Species DPS or Evolutionarily Significant Unit (ESU)	Physical or Biological Features Essential for the Conservation of the Species
	shade, submerged and overhanging large wood, log jams and beaver dams, **aquatic vegetation,** large rocks and boulders, side channels, and undercut banks. • Freshwater migration corridors free of obstruction with **water quantity and quality conditions** and natural cover such as submerged and overhanging large wood, **aquatic vegetation**, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival. • Estuarine areas free of obstruction with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, **aquatic vegetation**, large rocks and boulders, and side channels; and **juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation**. • Nearshore marine areas free of obstruction with **water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation**; and natural cover such as submerged and overhanging large wood, **aquatic vegetation**, large rocks and boulders, and side channels. • Offshore marine areas with **water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation**.
Chinook Salmon - Snake River fall-run - Snake River spring/summer run	juvenile rearing areas include adequate: spawning gravel, **water quality**, water quantity , water temperature, cover/shelter, **food**, **riparian vegetation,** space **juvenile and adult migration corridors are the same as for Snake River sockeye salmon
Sockeye Salmon - Snake River	spawning and juvenile rearing areas: spawning gravel, **water quality,** water quantity, water temperature, **food, riparian vegetation**, access, juvenile migration corridors:substrate, **water quality**, water quantity, water temperature, water velocity, cover/shelter, **food, riparian vegetation**, space, safe passage conditions **adult migration corridor has the same essential features, excluding “food”**

The following sections describe the status of species that occur in the action area and the threats to those species and where applicable, their designated critical habitat. A comprehensive description of these species, their life history, population dynamics and threats including climate change is available in Appendix A of this opinion.

6.3 Southern Resident Killer Whale

Status. We used information available in the final rule, the 2012 Status Review (NMFS 2013) (NMFS 2012) and the 2011 Stock Assessment Report (NMFS 2014) to summarize the status of this species. The Southern Resident killer whale DPS was listed as endangered in 2005 in response to the population decline from 1996 to 2001, small population size, and reproductive limitations (i.e., few reproductive males and delayed calving). This species occurs in the inland waterways of Puget Sound, Strait of Juan de Fuca, and Southern Georgia Strait during the spring, summer and fall. During the winter, they move to coastal waters primarily off Oregon, Washington, California, and British Columbia.

The most recent abundance estimate for the Southern Resident DPS is 87 whales in 2012. This represents an average increase of 0.4 percent annually since 1982 when there were 78 whales. Population abundance has fluctuated during this time with a maximum of approximately 100 whales in 1995 (NMFS 2013). As compared to stable or growing populations, the DPS reflects a smaller percentage of juveniles and lower fecundity (NMFS 2014) and has demonstrated weak growth in recent decades.

Threats. Current threats to its survival and recovery include: contaminants, vessel traffic, and reduction in prey availability. Chinook salmon populations have declined due to degradation of habitat, hydrology issues, harvest, and hatchery introgression; such reductions may require an increase in foraging effort. In addition, these prey contain environmental pollutants (e.g., flame retardants; PCBs and DDT). These contaminants become concentrated at higher trophic levels and may lead to immune suppression or reproductive impairment (70 FR 69903).

The inland waters of Washington and British Columbia support a large whale watch industry, commercial shipping, and recreational boating; these activities generate underwater noise, which may mask whales' communication or interrupt foraging. The factors that originally endangered the species persist throughout its habitat: contaminants, vessel traffic, and reduced prey. The DPS's resilience to future perturbation is reduced as a result of its small population size (N = 86); however, it has demonstrated the ability to recover from smaller population sizes in the past and has shown an increasing trend over the last several years. NMFS is currently conducting a status review prompted by a petition to delist the DPS based on new information, which indicates that there may be more paternal gene flow among populations than originally detected (Pilot et al. 2010).

Designated critical habitat. The designated critical habitat consists of approximately 6,630 km^2 in three areas: the Summer Core Area in Haro Strait and waters around the San Juan Islands; Puget Sound; and the Strait of Juan de Fuca. It provides the following physical and biological features: water quality to support growth and development; prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth; and inter-area passage conditions to allow for migration, resting, and foraging.

6.4 Pacific Salmonids

6.4.1 The 2016 Five-Year Status Reviews

The Pacific salmonid species have similar life histories, habitat needs, and threats. In May 2016, NOAA Fisheries' West Coast Region completed a five-year status review of all 28 West Coast salmon and steelhead species listed under the ESA (Table 5). Some species, such Oregon Coast coho salmon, mid-Columbia steelhead and Hood Canal chum, rebounded from the lows of past decades. Highly endangered Snake River sockeye have benefitted from a captive broodstock program while Snake River steelhead populations are steady. The California drought and unusually high ocean and stream temperatures over the 5-year period hit many populations hard. In the case of Sacramento River winter-run Chinook salmon, for example, drought conditions and high stream temperatures reduced the 2015 survival of juvenile fish in the first stretch of river to just three percent.

Table 5. Summary of current ESA listing status, recent trends and summary of conclusions for the most recent five-year review for Pacific salmonids (Northwest Fisheries Science Center 2015, Williams et al. 2016).

Species	ESU/DPS	Five-Year Review Risk Trend	ESA Listing Status
Chinook	Upper Columbia spring	Stable	Endangered
	Snake River spring/summer	Stable	Threatened
	Snake River fall	Improving	Threatened
	Upper Willamette spring	Declining	Threatened
	Lower Columbia	Stable/Improving	Threatened
	Puget Sound	Stable/Declining	Threatened
	California Coastal	Mixed	Threatened
	Central Valley Spring	Decreased risk of extinction	Threatened
	Sacramento River winter	Increased risk of extinction	Endangered
Coho	Lower Columbia	Stable/Improving	Threatened
	Oregon Coast	Improving	Threatened
	Southern Oregon/Northern California	Mixed	Threatened
	Central California Coast	Mixed	Endangered
Sockeye	Snake River	Improving	Endangered
	Lake Ozette	Stable	Threatened
Chum	Hood Canal summer	Improving	Threatened
	Columbia River	Stable	Threatened
Steelhead	Upper Columbia	Improving	Threatened
	Snake River	Stable/Improving	Threatened
	Middle Columbia	Stable/Improving	Threatened
	Upper Willamette	Declining	Threatened
	Lower Columbia	Stable	Threatened
	Puget Sound	Stable	Threatened
	Northern California	Mixed	Threatened
	Central California Coast	Uncertain	Threatened
	South Central California	Declining	Threatened
	Southern California	Uncertain	Endangered

Threats. During all freshwater life stages, salmonids require cool water that is free of contaminants. Water free of contaminants supports survival, growth, and maturation of salmon and the abundance of their prey. In addition to affecting survival, growth, and fecundity, contaminants can disrupt normal behavior necessary for successful migration, spawning, and juvenile rearing. Sufficient forage is necessary for juveniles to maintain growth that reduces freshwater predation mortality, increases overwintering success, initiates smoltification, and increases ocean survival. Natural riparian cover such as submerged and overhanging large wood and aquatic vegetation provides shelter from predators, shades freshwater to prevent increase in water temperature, provides nutrients from leaf litter, supports production of insect prey, and creates important side channels. Riparian vegetation stabilizes bank soils and captures fine sediment in runoff, which maintains functional channel bottom substrate for development of eggs and alevins.

The process of smoltification enables salmon to adapt to the ocean environment. Environmental factors such as exposure to chemicals including heavy metals and elevated water temperatures can affect the smoltification process, not only at the interface between fresh water and saltwater, but higher in the watershed as the process of transformation begins long before fish enter saltwater (Wedemeyer et al. 1980).

The three major threats to Atlantic salmon identified in the listing rule also threaten Pacific salmonids: dams, regulatory mechanisms related to dams, and low marine survival. In addition, a number of secondary threats were identified, including threats to habitat quality and accessibility, commercial and recreational fisheries, disease and predation, inadequacy of regulatory mechanisms related to water withdrawal and water quality, aquaculture, artificial propagation, climate change, competition, and depleted fish communities.

The action area for this consultation overlaps with designated critical habitat for all Pacific salmonids. NMFS has identified features of designated critical habitat that are essential to the conservation of the species. Many of these features specific to each life stage (e.g., migration, spawning, rearing, and estuary, see Table 5). The following sections describe the designated critical habitat for Pacific salmonids.

6.4.2 Chinook Salmon Designated Critical Habitat (Nine ESUs)

Designated critical habitat for the Puget Sound, Lower Columbia River, and Upper Willamette River ESUs for Chinook salmon identify features essential to the conservation of the species and sites necessary to support one or more Chinook salmon life stage(s). These features essential to the conservation of the species are detailed in Table 5 and include biological elements that are vulnerable to the stressors of the action. These include water quality conditions that support spawning and incubation, larval and juvenile development, and physiological transitions between fresh and saltwater. The features essential to the conservation of the species also include aquatic invertebrate and fish prey species and water quality to support juvenile and adult development, growth, and maturation, and natural cover of riparian and nearshore vegetation and aquatic vegetation. Designated critical habitat for the Snake River fall-run and Snake River spring/summer run Chinook salmon generically designates water quality, food, and riparian vegetation Features essential to the conservation of the species.

6.4.3 Chum Salmon Designated Critical Habitat (Two ESUs).

Areas designated as critical habitat are important for the species’ overall conservation by protecting quality growth, reproduction, and feeding. features essential to the conservation of the species for both chum salmon ESUs include freshwater spawning, rearing, and migration areas; estuarine and nearshore marine areas free of obstructions; and offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

6.4.4 Coho Salmon Designated Critical Habitat (Four ESUs)

The essential features of designated critical habitat for the Central California Coast and Southern Oregon/Northern California Coast coho salmon ESUs that are vulnerable to the stressors of the action are generically identified as water quality, food, and riparian vegetation. The essential features of designated critical habitat for the Lower Columbia River and Oregon Coast ESUs are more detailed. They include water quality conditions supporting spawning, incubation and larval development, water quality and forage supporting juvenile development; and natural cover of riparian and aquatic vegetation, water quality conditions supporting juvenile and adult physiological transitions between fresh- and saltwater, and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation (Table 5).

6.4.5 Sockeye Salmon Designated Critical Habitat (Two ESUs)

The essential features of designated critical habitat for Lake Ozette sockeye ESU that are potentially affected by the stressors of the action include water quality conditions and forage species supporting spawning, incubation, development, growth, maturation, physiological transitions between fresh and saltwater, and natural cover of riparian and nearshore vegetation and aquatic vegetation. The essential features of designated critical habitat for Snake River sockeye potentially affected by the stressors of the action are identified generically as water quality, food, and riparian vegetation (Table 5).

6.4.6 Steelhead Trout Designated Critical Habitat (Eleven ESUs)

Designated critical habitat. The essential features of designated critical habitat for all steelhead DPSs that are potentially affected by the stressors of the action include water quality conditions and/or forage species supporting spawning, incubation, development, growth, maturation, physiological transitions between fresh and saltwater, and natural cover of riparian and nearshore vegetation and aquatic vegetation (Table 5).

6.5 Atlantic Salmon, Gulf of Maine DPS

Status. The Gulf of Maine DPS of Atlantic salmon was first listed as endangered in response to population decline caused by many factors, including overexploitation, degradation of water quality, and damming of rivers, all of which remain persistent threats. The species' listing currently include all anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River, and wherever these fish occur in the estuarine and marine environment. The USFWS has jurisdiction over this species in freshwater, so the NMFS' jurisdiction is limited to potential PGP-authorized discharges from the coastal lands belonging to the Passamoquoddy Tribe at Pleasant Point. The most recent status review for Atlantic salmon was published in 2006 (Fay et al. 2006). This review stated that fewer than 1,500 adults have returned to spawn each year since 1998. The Population Viability Analysis estimates of the probability of extinction for the Gulf of Mexico DPS of Atlantic Salmon ranges from 19 percent to 75 percent within the next 100 years, even with the continuation of current levels of hatchery supplementation. The abundance was estimated at 1,014 individuals in 2007, the most recent year for which abundance records are available.

In 2015, NMFS announced a new program to focus and redouble its efforts to protect some of the species that are currently among the most at risk of extinction in the near future with the goal of reversing their declining trend so that the species will become a candidate for recovery in the future. Atlantic salmon is one of the eight species identified for this initiative (NMFS 2015c). These species were identified as among the most at-risk of extinction based on three criteria (1) endangered listing, (2) declining populations, and (3) are considered a recovery priority #1. A priority #1 species is one whose extinction is almost certain in the immediate future because of a rapid population decline or habitat destruction, whose limiting factors and threats are well understood and the needed management actions are known and have a high probability of success, and is a species that is in conflict with construction or other developmental projects or other forms of economic activity (55 FR 24296, June 15, 1990).

Designated critical habitat. The designated critical habitat includes all anadromous Atlantic salmon streams whose freshwater range occurs in watersheds from the Androscoggin River northward along the Maine coast northeastward to the Dennys River, and wherever these fish occur in the estuarine and marine environment. The features essential to the conservation of the species identified within freshwater and estuarine habitats of the occupied range of the Gulf of Maine DPS include sites for spawning and incubation, juvenile rearing, and migration. Designated critical habitat and features essential to the conservation of the species were not designated within marine environments because of the limited of the physical and biological features that the species uses during the marine phase of its life.

6.6 Southern Pacific Eulachon

Status. Eulachon are small smelt native to eastern North Pacific waters from the Bering Sea to Monterey Bay, California, or from 61º N to 31º N (Hart and McHugh 1944, Eschmeyer et al. 1983, Minckley et al. 1986, Hay and McCarter 2000). Eulachon that spawn in rivers south of the Nass River of British Columbia to the Mad River of California comprise the southern population of Pacific eulachon. This species status is classified as “at moderate risk of extinction throughout all of its range” (Gustafson 2010) based upon timing of runs and genetic distinctions (Hart and McHugh 1944, McLean et al. 1999, Hay and McCarter 2000, McLean and Taylor 2001, Beacham et al. 2005). Based on a number of data sources, the 2016 Status Review Update for eulachon reports that the spawning population has increased between 2011 and 2015 and that of the size of some sub-populations is larger than originally estimated in 2010 (Gustafson et al. 2016). The status update does not recommend a change in status because it is too early to tell whether recent improvements in the southern DPS of eulachon will persist. Recent poor ocean conditions taken with given variability inherent in wild populations suggest that population declines may again become widespread in the upcoming return years.

Threats. The Biological Review Team 2010 assessment of the status of the southern DPS of eulachon ranked climate change impacts on ocean conditions as the most serious threat to the persistence of eulachon in all four subareas of the DPS: Klamath River, Columbia River, Fraser River, and British Columbia coastal rivers south of the Nass River. Climate change impacts on freshwater habitat and eulachon bycatch in offshore shrimp fisheries were also ranked in the top four threats in all subareas of the DPS. Dams and water diversions in the Klamath and Columbia rivers and predation in the Fraser and British Columbia coastal rivers filled out the last of the top four threats (Gustafson 2010).

Designated critical habitat. The designated critical habitat for the southern population of Pacific eulachon includes freshwater creeks and rivers and their associated estuaries, comprising approximately 539 km (335 mi) of habitat. The physical or biological features potentially affected by the stressors of the action include water quality conditions supporting spawning and incubation, larval and adult mobility, and abundant prey items supporting larval feeding after the yolk sac is depleted, and nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival. Eulachon prey on a wide variety of species including crustaceans such as copepods and euphausiids (Hay and McCarter 2000, WDFW and ODFW 2001), unidentified malacostracans (Sturdevant et al. 1999), cumaceans (Smith and Saalfeld 1955) mysids, barnacle larvae, and worm larvae (WDFW and ODFW 2001).

6.7 Shortnose Sturgeon

Status. We used information available in the Shortnose Sturgeon Recovery Plan (NMFS 1998), the 2010 NMFS Biological Assessment (SNS BA 2010), and the listing document (32 FR 4001) to summarize the status of the species. Shortnose sturgeon were listed as endangered throughout its range on March 11, 1967 pursuant to the Endangered Species Preservation Act of 1966. Shortnose sturgeon remained on the list as endangered with enactment of the ESA in 1973. Shortnose sturgeon occur along the Atlantic Coast of North America, from the Saint John River in Canada to the Saint Johns River in Florida. The Shortnose Sturgeon Recovery Plan describes 19 shortnose sturgeon populations that are managed separately in the wild. Two additional geographically separated populations occur behind dams in the Connecticut River (above the Holyoke Dam) and in Lake Marion on the Santee-Cooper River system in South Carolina (above the Wilson and Pinopolis Dams). While shortnose sturgeon spawning has been documented in several rivers across its range (including but not limited to: Kennebec River, ME, Connecticut River, Hudson River, Delaware River, Pee Dee River, SC, Savannah, Ogeechee, and Altamaha rivers, GA), status for many other rivers remain unknown.

Threats. The viability of sturgeon populations is highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population. The 1998 recovery plan for shortnose sturgeon (NMFS 1998) identify Habitat degradation or loss (resulting, for example, from dams, bridge construction, channel dredging, and pollutant discharges), and mortality (for example, from impingement on cooling water intake screens, dredging, and incidental capture in other fisheries) as principal threats to the species' survival. Introductions and transfers of indigenous and nonindigenous sturgeon, intentional or accidental, may threaten wild shortnose sturgeon populations by imposing genetic threats, increasing competition for food or habitat, or spreading diseases. Sturgeon species are susceptible to viruses enzootic to the west coast and fish introductions could further spread these diseases. Shortnose sturgeon populations are at risk from incidental bycatch, loss of habitat, dams, dredging and pollution. These threats are likely to continue into the future. We conclude that the shortnose sturgeon’s resilience to further perturbation is low.

Designated critical habitat. No critical habitat has been designated for shortnose sturgeon.

6.8 Atlantic Sturgeon (Five DPSs)

Status. The range of Atlantic sturgeon includes the St. John River in Canada, to St. Johns River in Florida. EPA has NPDES permitting authority throughout New Hampshire, Massachusetts, the District of Columbia, Federally operated facilities in Delaware and Tribal lands in Connecticut, Rhode Island, New York, North Carolina, and Florida. Five DPSs of Atlantic sturgeon were designated and listed under the ESA on February 6, 2012 (Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic). The Gulf of Maine, New York Bight, and Chesapeake Bay DPSs are those potentially affected by the 2016 PGP.

Threats. Of the stressors evaluated in the 2007 status review (ASSRT 2007), bycatch mortality, water quality, lack of adequate state and/or Federal regulatory mechanisms, and dredging activities were most often identified as the most significant threats to the viability of Atlantic sturgeon populations. Additionally, some populations were affected by unique stressors, such as habitat impediments (e.g., Cape Fear and Santee-Cooper rivers) and apparent ship strikes (e.g., Delaware and James rivers).

Designated critical habitat. The proposed designated critical habitat for Atlantic sturgeon includes tidally-affected accessible waters of coastal estuaries where the species occurs (81 FR 35701, 81 FR 36077). The essential features of the proposed designated critical habitat for the Atlantic sturgeon DPSs within these rivers do not include plant or animal life that may be affected by the stressors of the action. From north to south, the rivers and waterways that make up the spatial extent of designated critical habitat are detailed in Table 6.

Table 6. River systems in the action area that are included in proposed designated critical habitat for Atlantic sturgeon.

DPS	River/Waterway		
Gulf of Maine	Penobscot Piscataqua	Kennebec Merrimack	Androscoggin
New York Bight	Connecticut Housatonic Delaware	Housatonic	Hudson
Chesapeake Bay	Susquehanna York James	Potomac Mattaponi	Rappahannock Pamunkey

6.9 Green Sturgeon

Status. We used information available in the 2002 Status Review and Status Review Updates (GSSR 2002, 2005, 2015), and the proposed and final listing rules (70 FR 17836; 71 FR 17757) to summarize the status of the species. The Southern DPS of green sturgeon is listed as threatened (71 FR 17757; April 7, 2006). On June 2, 2010, NMFS issued a 4(d) Rule for the Southern DPS, applying certain take prohibitions (75 FR 30714). The most recent 5-year status review was published in August of 2015. Green sturgeon occur in coastal Pacific waters from San Francisco Bay to Canada. The Southern DPS of green sturgeon includes populations south of (and exclusive of) the Eel River, coastal and Central Valley populations, and the spawning population in the Sacramento River, CA (Adams et al. 2007).

The 2015 status update indicates that DPS structure of the North American green sturgeon has not changed and that many of the principle factors considered when listing Southern DPS green sturgeon as threatened are relatively unchanged. Loss of spawning habitat and bycatch in the white sturgeon commercial fishery are two major causes for the species decline. Spawning in the Feather River is encouraging and the decommissioning of Red Bluff Diversion Dam and breach of Shanghai Bench makes spawning conditions more favorable. The prohibition of retention in commercial and recreational fisheries has eliminated a known threat and likely had a very positive effect on the overall population, although recruitment indices are not presently available.

Threats. The 2015 status review (NMFS 2015b) for the southern DPS of green sturgeon indicates that many of the principle factors considered when listing Southern DPS green sturgeon as threatened are relatively unchanged. Current threats to the Southern DPS include entrainment by water projects, contaminants, incidental bycatch and poaching. Given the small population size, the species' life history traits (e.g., slow to reach sexual maturity), and that the threats to the population are likely to continue into the future, the Southern DPS is not resilient to further perturbations. The spawning area for the species is still small, as the species still encounters impassible barriers in the Sacramento, Feather and other rivers that limit their spawning range. Entrainment threat includes stranding in flood diversions during high water events.

Designated critical habitat. Designated critical habitat for the Southern DPS of green sturgeon was designated includes coastal U.S. marine waters within 60 fathoms deep from Monterey Bay, CA to Cape Flattery, Washington, including the Strait of Juan de Fuca, and numerous coastal rivers and estuaries: see the Final Rule for a complete description (74 FR 52300). Essential features identified in this designation that may be affects by the stressors of the action include acceptably low levels of contaminants (e.g., pesticides, PAHs, heavy metals that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon) and abundant prey items (benthic invertebrates and fish) for subadults and adults.

6.10 Bocaccio Puget Sound/Georgia Basin DPS

The bocaccio that occur in the Georgia Basin are listed as an endangered "species," which, in this case, refers to a distinct segment of a vertebrate population (75 FR 22276). The listing includes bocaccio throughout Puget Sound, which encompasses all waters south of a line connecting Point Wilson on the Olympic Peninsula and Partridge on Whidbey Island; West Point on Whidbey Island, Deception Island, and Rosario Head on Fidalgo Island; and the southern end of Swinomish Channel between Fidalgo Island and McGlinn Island (U.S. Geological Survey 1979), and the Strait of Georgia, which encompasses the waters inland of Vancouver Island, the Gulf Islands, and the mainland coast of British Columbia.

Status. Bocaccio have always been rare in recreational fisheries that occur in North Puget Sound and the Strait of Georgia; however, there have been no confirmed reports of bocaccio in Georgia Basin for several years. Although their abundance cannot be estimated directly, NMFS' BRT estimated that the populations of bocaccio and yelloweye rockfish are small in size, probably numbering fewer than 10,000 individuals in Georgia Basin and fewer than 1,000 total individuals in Puget Sound (74 FR 18532) (Drake et al. 2010). Georgia Basin bocaccio are most common at depths between 50 and 250 meters (160 and 820 feet).

Threats. The 2016 draft recovery plan for rockfish indicates that historical overfishing is recognized as the primary cause of the decline of rockfishes in Puget Sound (Palsson et al. 2008, Drake et al. 2010, Williams et al. 2010), there is some uncertainty about the relative impact of some fisheries today, and of the additional remaining threats, which include degraded water quality and habitat, contaminants, derelict fishing gear, and other threats (Palsson et al. 2008, Drake et al. 2010, WDFW 2013).

Designated critical habitat. NMFS proposed critical habitat designation includes approximately 1,185 mi^2 of marine habitat for bocaccio in Puget Sound, Washington. Physical or biological features essential to adult bocaccio include the benthic habitats or sites deeper than 30m (98ft) that possess or are adjacent to areas of complex bathymetry consisting of rock and or highly rugose habitat are essential to conservation because these features support growth, survival, reproduction, and feeding opportunities by providing the structure for rockfish to avoid predation, seek food and persist for decades. Several attributes of these sites determine the quality of the habitat and are useful in considering the conservation value of the associated feature, and whether the feature may require special management considerations or protection. These attributes are also relevant in the evaluation of the effects of a proposed action in a section 7 consultation if the specific area containing the site is designated as critical habitat. These attributes include: (1) Quantity, quality and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities, (2) water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities, and (3) the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance.

6.11 Yelloweye and Canary Rockfish (Puget Sound/Georgia Basin DPS)

Status. July of 2016 NMFS petitioned to delist the canary rockfish based on newly obtained genetic information that demonstrates that the Puget Sound/Georgia Basin canary rockfish population does not meet the DPS criteria and therefore does not qualify for listing under the ESA. Georgia Basin yelloweye rockfish occur through Puget Sound, which encompasses all waters south of a line connecting Point Wilson on the Olympic Peninsula and Partridge on Whidbey Island; West Point on Whidbey Island, Deception Island, and Rosario Head on Fidalgo Island; and the southern end of Swinomish Channel between Fidalgo Island and McGlinn Island (U.S. Geological Survey 1979), and the Strait of Georgia, which encompasses the waters inland of Vancouver Island, the Gulf Islands, and the mainland coast of British Columbia.

The frequency of yelloweye rockfish in collections from Puget Sound appears to have been highly variable; frequencies were less than 1 percent in the 1960s and 1980s and about 3 percent in the 1970s and 1990s. In North Puget Sound, however, the frequency of yelloweye rockfish has been estimated to have declined from a high of greater than 3 percent in the 1970s to about 0.65 percent in more recent samples. This decline combined with their low intrinsic growth potential, threats from bycatch in commercial and recreational fisheries, loss of nearshore rearing habitat, chemical contamination, and the proportion of coastal areas with low dissolved oxygen levels led to this species' listing as threatened under the ESA.

Although their abundance cannot be estimated directly, NMFS' Biological Review Team estimated that the populations of bocaccio, yelloweye rockfish and canary rockfish are small in size, probably numbering fewer than 10,000 individuals in Georgia Basin and fewer than 1,000 total individuals in Puget Sound (74 FR 18532) (Drake et al. 2010).

Designated critical habitat. Physical or biological features essential to the conservation of both adult and juvenile yelloweye rockfish are the same as for adult bocaccio and adult canary rockfish.

6.12 Nassau Grouper

Status. The Nassau grouper (*Epinephelus striatus*) is primarily a shallow-water, insular fish species found from inshore to about 330 feet (100m) depth. The species is distributed throughout the islands of the western Atlantic including Bermuda, the Bahamas, southern Florida and along the coasts of central and northern South America. It is not known from the Gulf of Mexico except at Campeche Bank off the coast of the Yucatan Peninsula, at Tortugas, and off Key West. Adults are generally found near coral reefs and rocky bottoms while juveniles are found in shallower waters in and around coral clumps covered with macroalgae (Laurencia spp.) and over seagrass beds. Their diet is mostly fishes and crabs, with diet varying by age/size. Juveniles feed mostly on crustaceans, while adults (>30 cm; 11.8 in) forage mainly on fish. The Nassau grouper usually forages alone and is not a specialized forager.

Under the authority of the Magnuson-Stevens Fisheries Act, NMFS classified the Nassau grouper as "overfished" in its October 1998 "Report to Congress on the status of Fisheries and Identification of overfished Stocks."

Designated critical habitat. Designated critical habitat has not been designated for this species.

6.13 Sea Turtles

Sea turtles share the common threats described below.

Bycatch: Fishing is the primary anthropogenic threat to sea turtles in the ocean. Fishing gear entanglement potentially drowns or seriously injures sea turtles. Fishing dredges can crush and entrap turtles, causing death and serious injury. Infection of entanglement wounds can compromise health. The development and operation of marinas and docks in inshore waters can negatively impact nearshore habitats. Turtles swimming or feeding at or just beneath the surface of the water are particularly vulnerable to boat and vessel strikes, which can result in serious propeller injuries and death.

Marine Debris: Ingestion or entanglement in marine debris is a cause of morbidity and mortality for sea turtles in the pelagic (open ocean) environment (Stamper et al. 2009). Consumption of non-nutritive debris also reduces the amount of nutritive food ingested, which then may decrease somatic growth and reproduction (McCauley and Bjorndal 1999). Marine debris is especially problematic for turtles that spend all or significant portions of their life cycle in the pelagic environment (e.g., leatherbacks, juvenile loggerheads, and juvenile green turtles).

Habitat Disturbance: Sea turtle nesting and marine environments are facing increasing impacts through structural modifications, sand nourishment, and sand extraction to support widespread development and tourism (Lutcavage et al. 1997, Bouchard et al. 1998, Hamann et al. 2006, Maison 2006, Hernandez et al. 2007, Santidrián Tomillo et al. 2007, Patino-Martinez 2013). These factors decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings through direct loss of and indirect (e.g., altered temperatures, erosion) mechanisms (Ackerman 1997, Witherington et al. 2003, 2007). Lights from developments alter nesting adult behavior and are often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Witherington and Bjorndal 1991, Witherington 1992, Cowan et al. 2002, Deem et al. 2007, Bourgeois et al. 2009).

Beach nourishment also affects the incubation environment and nest success. Although the placement of sand on beaches may provide a greater quantity of nesting habitat, the quality of that habitat may be less suitable than pre-existing natural beaches. Constructed beaches tend to differ from natural beaches in several important ways. They are typically wider, flatter, more compact, and the sediments are more moist than those on natural beaches (Nelson et al. 1987) (Ackerman 1997, Ernest and Martin 1999). Nesting success typically declines for the first year or two following construction, even when more nesting area is available for turtles ((Trindell et al. 1998) (Ernest and Martin 1999, Herren 1999). Likely causes of reduced nesting success on constructed beaches include increased sand compaction, escarpment formation, and changes in beach profile (Nelson et al. 1987, Grain et al. 1995, Lutcavage et al. 1997, Steinitz et al. 1998, Ernest and Martin 1999, Rumbold et al. 2001). Compaction can inhibit nest construction or increase the amount of time it takes for turtles to construct nests, while escarpments often cause female turtles to return to the ocean without nesting or to deposit their nests seaward of the escarpment where they are more susceptible to frequent and prolonged tidal inundation. In short, sub-optimal nesting habitat may cause decreased nesting success, place an increased energy burden on nesting females, result in abnormal nest construction, and reduce the survivorship of eggs and hatchlings. In addition, sand used to nourish beaches may have a different composition than the original beach; thus introducing lighter or darker sand, consequently affecting the relative nest temperatures (Ackerman 1997, Milton et al. 1997).

In addition to effects on sea turtle nesting habitat, anthropogenic disturbances also threaten coastal foraging habitats, particularly areas rich in seagrass and marine algae. Coastal habitats are degraded by pollutants from coastal runoff, marina and dock construction, dredging, aquaculture, oil and gas exploration and extraction, increased under water noise and boat traffic, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999, Lee Long et al. 2000, Waycott et al. 2005).

Pollutants: Conant et al. (2009) included a review of the impacts of marine pollutants on sea turtles: marine debris, oil spills, and bioaccumulative chemicals. Sea turtles at all life stages appear to be highly sensitive to oil spills, perhaps due to certain aspects of their biology and behavior, including a lack of avoidance behavior, indiscriminate feeding in convergence zones, and large pre-dive inhalations (Milton and Lutz 2003). Milton et al. (2003) state that the oil effects on turtles include increased egg mortality and developmental defects, direct mortality due to oiling in hatchlings, juveniles and adults, and impacts to the skin, blood, salt glands, and digestive and immune systems. Vargo et al. (1986) reported that sea turtles would be at substantial risk if they encountered an oil spill or large amounts of tar in the environment. In a review of available information on debris ingestion, Balazs (1985) reported that tar balls were the second most prevalent type of debris ingested by sea turtles. Physiological experiments showed that sea turtles exposed to petroleum products may suffer inflammatory dermatitis, ventilator disturbance, salt gland dysfunction or failure, red blood cell disturbances, immune response, and digestive disorders (Vargo et al. 1986, Lutcavage et al. 1995).

Natural Threats: A number of threats are common to all sea turtles.[11] Predation is a primary natural threat. While cold stunning is not a major concern for leatherback sea turtles, which can tolerate low water temperatures, it is considered a major natural threat to other sea turtle species. Disease is also a factor in sea turtle survival. Fibropapillomatosis (FP) tumors are a major threat to green turtles in some areas of the world and is particularly associated with degraded coastal habitat. Scientists have also documented FP in populations of loggerhead, olive ridley, and flatback turtles, but reports in green turtles are more common. Large tumors can interfere with feeding and essential behaviors, and tumors on the eyes can cause permanent blindness. FP was first described in green turtles in the Florida Keys in the 1930s. Since then it has been recorded in many green turtle populations around the world. The effects of FP at the population level are not well understood. The sand-borne fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* capable of killing greater than 90 percent of sea turtle embryos they infect, threatening nesting productivity under some conditions. These pathogens can survive on decaying organic matter and embryo mortality rates attributed to fusarium were associated with clay/silt nesting areas compared to sandy areas (Sarmiento-Ramırez et al. 2014).

Climate Change and Sea Turtle Nesting Habitat. While all species are affected by the influence of climate change on habitat distribution and quality, the Conant et al. 2009 review describes unique impact of climate change on sea turtle nesting habitat. Rising sea level is one of the most certain consequences of climate change (Titus and Narayanan 1995), and will result in increased erosion rates along nesting beaches. This could particularly affect areas with low-lying

[11] See http://www.nmfs.noaa.gov/pr/species/turtles/threats.htm, updated June 16, 2014

beaches where sand depth is a limiting factor, as the sea will inundate nesting sites and decrease available nesting habitat (Fish et al. 2005, Baker et al. 2006). The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Baker et al. 2006). On some undeveloped beaches, shoreline migration will have limited effects on the suitability of nesting habitat. The Bruun rule specifies that during a sea level rise, a typical beach profile will maintain its configuration but will be translated landward and upward (Rosati et al. 2013). However, along developed coastlines, and especially in areas where erosion control structures have been constructed to limit shoreline movement, rising sea levels will cause severe effects on nesting females and their eggs. Erosion control structures can result in the permanent loss of dry nesting beach or deter nesting females from reaching suitable nesting sites (Council 1990). Nesting females may deposit eggs seaward of the erosion control structures potentially subjecting them to repeated tidal inundation. Non-native vegetation often out competes native species, is usually less stabilizing, and can lead to increased erosion and degradation of suitable nesting habitat. Exotic vegetation may also form impenetrable root mats that can prevent proper nest cavity excavation, invade and desiccate eggs, or trap hatchlings.

6.13.1 Leatherback Sea Turtle

Status. The leatherback sea turtle is unique among sea turtles for its large size, wide distribution (due to thermoregulatory systems and behavior), and lack of a hard, bony carapace. It ranges from tropical to subpolar latitudes, worldwide.

The global population of adult females has declined over 70 percent in less than one generation, from an estimated 115,000 adult females in 1980 to 34,500 adult females in 1995 (Pritchard 1982, Spotila et al. 1996). There may be as many as 34,000 – 94,000 adult leather backs in the North Atlantic, alone (TEWG 2007), but dramatic reductions (> 80 percent) have occurred in several populations in the Pacific, which was once considered the stronghold of the species (Sarti Martinez 2000). The 2013 five year review (NMFS and USFWS 2013b) reports that the East Pacific and Malaysia leatherback populations have collapsed, yet Atlantic populations generally appear to be stable or increasing. Many explanations have been provided to explain the disparate population trends, including fecundity and foraging differences seen in the Pacific, Atlantic, and Indian Oceans. Since the last 5-year review, studies indicate that high reproductive output and consistent and high quality foraging areas in the Atlantic Ocean have contributed to the stable or recovering populations; whereas prey abundance and distribution may be more patchy in the Pacific Ocean, making it difficult for leatherbacks to meet their energetic demands and lowering their reproductive output. Both natural and anthropogenic threats to nesting and marine habitats continue to affect leatherback populations, including the 2004 tsunami in the Indian Ocean, 2010 oil spill in the U.S. Gulf of Mexico, logging practices, development, and tourism impacts on nesting beaches in several countries.

In 2015, NMFS announced a new program to focus and redouble its efforts to protect some of the species that are currently among the most at risk of extinction in the near future with the goal of reversing their declining trend so that the species will become a candidate for recovery in the future. The leatherback sea turtle is one of the eight species identified for this initiative (NMFS 2015c). These species were identified as among the most at-risk of extinction based on three criteria (1) endangered listing, (2) declining populations, and (3) are considered a recovery priority #1. A priority #1 species is one whose extinction is almost certain in the immediate future because of a rapid population decline or habitat destruction, whose limiting factors and threats are well understood and the needed management actions are known and have a high probability of success, and is a species that is in conflict with construction or other developmental projects or other forms of economic activity (55 FR 24296, June 15, 1990).

Designated critical habitat. On March 23, 1979, leatherback designated critical habitat was identified adjacent to Sandy Point, St. Croix, U.S. Virgin Islands from the 183 m isobath to mean high tide level between 17° 42'12" N and 65°50'00" W (44 FR 17710). This habitat is essential for nesting, which has been increasingly threatened since 1979, when tourism increased significantly, bringing nesting habitat and people into close and frequent proximity; however, studies do not support significant designated critical habitat deterioration. On January 20, 2012, NMFS issued a final rule to designate additional designated critical habitat for the leatherback sea turtle (50 CFR 226). This designation includes approximately 43,798 km2 stretching along the California coast from Point Arena to Point Arguello east of the 3000 m depth contour; and 64,760 km^2 stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 m depth contour. The designated areas comprise approximately 108558 km2 of marine habitat and include waters from the ocean surface down to a maximum depth of 80 m. They were designated specifically because of the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (i.e., jellyfish), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

6.13.2 Hawksbill Sea Turtle

Status. The hawksbill sea turtle has a sharp, curved, beak-like mouth. It has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical oceans. The hawksbill turtle was once abundant in tropical and subtropical regions throughout the world. Over the last century, this species has declined in most areas and stands at only a fraction of its historical abundance. According to the 2013 status review (NMFS and USFWS 2013a), nesting populations in the eastern Pacific, and the Nicaragua nesting population in the western Caribbean appears to have improved. However, the trends and distribution of the species throughout the globe largely is unchanged. Although greatly depleted from historical levels, nesting populations in the Atlantic in general are doing better than in the Indian and Pacific Oceans. In the Atlantic, more population increases have been recorded in the insular Caribbean than along the western Caribbean mainland or the eastern Atlantic. In general, hawksbills are doing better in the Indian Ocean (especially the southwestern and northwestern Indian Ocean) than in the Pacific Ocean. The situation for hawksbills in the Pacific Ocean is particularly dire, despite the fact that it still has more nesting hawksbills than in either the Atlantic or Indian Oceans.

Designated critical habitat. On September 2, 1998, NMFS established designated critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico (63 FR 46693). Aspects of these areas that are important for hawksbill sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for hawksbill sea turtle prey.

6.13.3 Kemp's Ridley Sea Turtle

Status. The Kemp's ridley is the smallest of all sea turtle species and considered to be the most endangered sea turtle, internationally (Groombridge 1982, TEWG 2000). The species was first listed under the Endangered Species Conservation Act (35 FR 8491) and listed as endangered under the ESA since 1973. According to the 2015 status review (NMFS and USFWS 2013a), population growth rate (as measured by numbers of nests) stopped abruptly after 2009. Given the recent lower nest numbers, the population is not projected to grow at former rates. An unprecedented mortality in subadult and adult females post-2009 nesting season may have altered the 2009 age structure and momentum of the population, which had a carryover impact on annual nest numbers in 2011-2014. The results indicate the population is not recovering and cannot meet recovery goals unless survival rates improve. The Deep Water Horizon oil spill that occurred at the onset of the 2010 nesting season and exposed Kemp's ridleys to oil in nearshore and offshore habitats may have been a factor in fewer females nesting in subsequent years, however this is still under evaluation. The long-term impacts from the Deep Water Horizon oil spill and response to the spill (e.g., dispersants) to sea turtles are not yet known. Given the Gulf of Mexico is an area of high-density offshore oil exploration and extraction, future oil spills are highly probable and Kemp's ridleys and their habitat may be exposed and injured. Commercial and recreational fisheries continue to pose a substantial threat to the Kemp's ridley despite measures to reduce bycatch. Kemp's ridleys have the highest rate of interaction with fisheries operating in the Gulf of Mexico and Atlantic Ocean than any other species of turtle.

Designated critical habitat. Designated critical habitat has not been designated for this species.

6.13.4 Olive Ridley Sea Turtle

Status. The olive ridley sea turtle is a small, mainly pelagic, sea turtle with a circumtropical distribution. The species was listed under the ESA on July 28, 1978 (43 FR 32800). The species was separated into two listing designations: endangered for breeding populations on the Pacific coast of Mexico, and threatened wherever found except where listed as endangered (i.e., in all other areas throughout its range). The status review (NMFS and USFWS 2014), indicates that, based on the current number of olive ridleys nesting in Mexico, three populations appear to be stable (Mismaloya, Tlacoyunque, and Moro Ayuta), two increasing (Ixtapilla, La Escobilla) and one decreasing (Chacahua). Elsewhere in the eastern Pacific, the large arribada nesting populations have declined since the 1970s. Nesting at some arribada beaches continues to decline (e.g., Nancite in Costa Rica) and is stable or increasing at others (e.g., Ostional in Costa Rica). There are too few data available from solitary nesting beaches to confirm the declining trend that has been described for numerous countries throughout the region including El Salvador, Guatemala, Costa Rica, and Panama. Recent at-sea estimates of density and abundance of the olive ridley in the Pacific show a yearly estimate of 1.39 million (Confidence Interval: 1.15 to 1.62 million), which is consistent with the increases seen on nesting beaches as a result of protection programs that began in the 1990s.

Western Atlantic arribada nesting populations are currently very small. The Suriname olive ridley population is currently small and has declined by more than 90 percent since the late 1960s. However, nesting is reported to be increasing in French Guiana. The other nesting population in Brazil, for which no long term data are available, is small, but increasing. In the eastern Atlantic, long-term data are not available and thus the abundance and trends of this population cannot be assessed at this time. In the northern Indian Ocean, arribada nesting populations are still large, but trend data are ambiguous and major threats continue. Declines of solitary nesting olive ridleys have been reported in Bangladesh, Myanmar, Malaysia, Pakistan, and southwest India.

Designated critical habitat. Designated critical habitat has not been designated for this species.

6.13.5 Loggerhead Sea Turtle

Status. Based on the 2009 status review (Conant et al. 2009), for three of five DPSs with sufficient data (Northwest Atlantic Ocean, South Pacific Ocean, and North Pacific Ocean), analyses indicate a high likelihood of quasi-extinction. Similarly, threat matrix analysis indicated that all other DPSs have the potential for a severe decline in the future.

Northwest Atlantic Ocean loggerhead sea turtle DPS designated critical habitat. The final designated critical habitat for the Northwest Atlantic Ocean loggerhead DPS within the Atlantic Ocean and the Gulf of Mexico includes 36 occupied marine areas within the range of the Northwest Atlantic Ocean DPS (79 FR 39855). These areas contain one or a combination of nearshore reproductive habitat, winter area, breeding areas, and migratory corridors.

6.13.6 Green Sea Turtle

Status. The green sea turtle was separated into two listing designations: endangered for breeding populations in Florida and the Pacific coast of Mexico, and threatened in all other areas throughout its range. On August 1, 2012, NMFS found that a petition to identify the Hawaiian population of green turtle as a DPS, and to delist the DPS, may be warranted (77 FR 45571). In April 2016, we removed the range-wide and breeding population listings of the green sea turtle, and in their place, listed eight DPSs as threatened and 3 DPSs as endangered (81 FR 20057). Among these, only the North Atlantic DPS occurs in waters where EPA has permitting authority.

Once abundant in tropical and subtropical waters, globally, green sea turtles exist at a fraction of their historical abundance, as a result of over-exploitation. The North Atlantic DPS is characterized by geographically widespread nesting with eight sites having high levels of abundance (i.e., <1,000 nesters). Nesting is reported in 16 countries and/or U.S. Territories at 73 sites. This region is data rich and has some of the longest running studies on nesting and foraging turtles anywhere in the world. All major nesting populations demonstrate long-term increases in abundance. The prevalence of FP has reached epidemic proportions in some parts of the North Atlantic DPS. The extent to which this will affect the long-term outlook for green turtles in the North Atlantic DPS is unknown and remains a concern, although nesting trends across the DPS continue to increase despite the high incidence of the disease. There are still concerns about future risks, including habitat degradation (particularly coastal development), bycatch in fishing gear, continued turtle and egg harvesting, and climate change.

Designated critical habitat. On September 2, 1998, NMFS designated critical habitat for green sea turtles (63 FR 46694), which include coastal waters surrounding Culebra Island, Puerto Rico. Seagrass beds surrounding Culebra provide important foraging resources for juvenile, subadult, and adult green sea turtles. Additionally, coral reefs surrounding the island provide resting shelter and protection from predators. This area provides important developmental habitat for the species.

6.14 Corals

Status. There are currently 22 coral species listed as threatened under the ESA, 16 of which occur in the action area (Table 7). Information from the proposed listings (77 FR 73219 and 79 FR 53852) and status reports (ABRT 2005) were used to summarize the status of these species.

Table 7: Threatened coral species occurring in the PGP action area.

Threatened Corals	Currently Known in These U.S. Geographic Areas			
Caribbean Waters: Puerto Rico				
Acropora cervicornis (Staghorn)*and designated critical habitat*		X		
Acropora palmata (Elkhorn) *and designated critical habitat*		X		
Mycetophyllia ferox		X		
Dendrogyra cylindrus		X		
Orbicella annularis		X		
Orbicella faveolata		X		
Orbicella franksi		X		
Pacific Waters				
	Guam	**Northern Mariana Islands**	**Pacific Remote Island Areas**	**American Samoa**
Acropora globiceps	X	X	X	X
Acropora jacquelineae				X
Acropora retusa	X		X	X
Acropora rudis				X
Acropora speciosa			X	X
Euphyllia paradivisa				X
Isopora crateriformis				X
Pavona diffluens	X	X		X
Seriatopora aculeata	X			

Threats. Massive mortality events from disease conditions of corals and the keystone grazing urchin *Diadema antillarum* have precipitated widespread and dramatic changes in reef community structure. Large-scale coral bleaching reduces population viability. In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and bleaching events from ocean warming have added to the poor state of coral populations and yielded a remnant coral community with increased dominance by weedy brooding species, decreased overall coral cover, and increased macroalgal cover. Additionally, iron enrichment may predispose the basin to algal growth. Further, coral growth rates in many areas have been declining over decades. Such reductions prevent successful recruitment as a result of reduced density. Finally, climate change is likely to result in the endangerment of many species as a result of temperature increases (and resultant bleaching), sea level rises, and ocean acidification (van Dam et al. 2012a, Gittings et al. 2013).

Designated critical habitat. On November 26, 2008, NMFS designated critical habitat for elkhorn and staghorn coral. They designated marine habitat in four specific areas: Florida (1,329 square miles), Puerto Rico (1,383 square miles), St. John/St. Thomas (121 square miles), and St. Croix (126 square miles). These areas support the following physical or biological features that are essential to the conservation of the species: substrate of suitable quality and availability to support successful larval settlement and recruitment and reattachment and recruitment of fragments.

7 ENVIRONMENTAL BASELINE

The *Environmental Baseline* is defined as: "past and present impacts of all Federal, State, or private actions and other human activities in an action area, the anticipated impacts of all proposed Federal projects in an action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process" (50 CFR 402.02). The key purpose of the environmental baseline is to describe the natural and anthropogenic factors influencing the status and condition of ESA-listed species and designated critical habitat in the action area. Since this is a programmatic consultation on what is primarily a continuing action with a large geographic scope, this environmental baseline focuses more generally on the status and trends of the aquatic ecosystems in the U.S. and the consequences of that status for listed resources. The action considered in this opinion is the Clean Water Act PGP authorization of discharge of pesticide pollutants to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur, and the interrelated actions of discharges of pesticides that are not included in the definition of pesticide pollutants. For this reason the discussion of the baseline conditions for this opinion focuses on water quality and pesticides. A more comprehensive discussion of the baseline condition of these species is provided in Appendix B, which includes consideration of impacts to the environmental baseline of factors such as climate change, by-catch, vessel-strikes, etc.

Activities that negatively impact water quality also threaten aquatic species. The deterioration of water quality is a contributing factor that has led to the endangerment of some aquatic species under NMFS' jurisdiction. Declines in populations of ESA-listed species leave them vulnerable to a multitude of threats. Due to the cumulative effects of reduced abundance, low or highly variable growth capacity, and the loss of essential habitat, these species are less resilient to additional disturbances. In larger populations, stressors that affect only a limited number of individuals could once be tolerated by the species without resulting in population level impacts; in smaller populations, the same stressors are more likely to reduce the likelihood of survival. It is with this understanding of the *Environmental Baseline* that we consider the effects of the proposed action, including the likely effect that the PGP will have on endangered and threatened species and their designated critical habitat. There may be direct and indirect effects of activities associated with the proposed PGP in streams, wetlands, rivers, lakes, estuaries, irrigation canals, and drainage systems into, over, and in close proximity to which pesticides are applied. Areas adjacent to or downstream from these jurisdictional areas may be indirectly affected by activities authorized under the PGP. As noted in Section 4, we also analyze effects from the interrelated discharges of pesticides that do not fall into the category of pesticide pollutants. Based on the *Action Area*, as defined in Section 5 above, we identified the following regions and states for inclusion in the *Environmental Baseline* section of this opinion: Pacific Coast (Washington, Idaho, Oregon, and California); New England (Maine, New Hampshire, Vermont, and Massachusetts); Mid-Atlantic (District of Columbia, Delaware, and Virginia); U.S. Caribbean (Puerto Rico) and U.S. Pacific Islands (excluding Hawaii). These regions/states cover the vast majority of the proposed action area. At the regional level, our baseline assessment focused on the natural and anthropogenic threats affecting the ESA-listed species (and their habitats) within the action area for each particular region: Pacific Coast – all listed ESUs and DPSs of Pacific salmon and steelhead, eulachon, Southern DPS green sturgeon, and Southern Resident killer whale; New England – Atlantic salmon, Atlantic sturgeon (5 listed DPSs); Mid-Atlantic - Atlantic sturgeon (5 listed DPSs); Caribbean – Nassau grouper, elkhorn coral, staghorn coral, lobed star coral, boulder star coral, mountainous star coral, pillar coral, and rough cactus coral; Pacific Islands – all listed Pacific Islands coral species.

While there are some Tribal lands and federal facilities in regions or states not mentioned above, in general these areas are either very small, far removed from ESA-listed species or habitat, or not affected by the proposed action. For example, any discharges of pesticide on Tribal lands in Florida would have to be transported through Everglades or Big Cypress National Parks, where they would be degraded by exposure to sunlight, microbial action and chemical processes. While all areas of overlap between ESA-listed species (and their designated critical habitat) and the PGP coverage area are evaluated in this opinion, the environmental baseline will focus specifically on the aquatic ecosystems in the regions/states (listed above) where the anticipated effects of the proposed action are considered more likely to adversely affect ESA-listed species.

The action area for this consultation covers a very large number of individual watersheds and an even larger number of specific water bodies (e.g., lakes, rivers, streams, estuaries). It is, therefore, not practicable to describe the environmental baseline and assess risk for each particular area where the PGP may authorize discharges and activities. Accordingly, this opinion approaches the environmental baseline more generally by describing the activities, conditions and stressors which adversely affect ESA-listed species and designated critical habitat. These include natural threats (e.g., parasites and disease, predation and competition, wildland fires), water quality, hydromodification projects, land use changes, dredging, mining, artificial propagation, non-native species, fisheries, vessel traffic, and climate changes. For each of these threats we start with a general overview of the problem, followed by a more focused analysis at the regional and state level for the species listed above, as appropriate and where such data are available.

Our summary of the environmental baseline complements the information provided in the Status of Listed Resources section of this opinion, and provides the background necessary to evaluate and interpret information presented in the Effects of the Proposed Action and Cumulative Effects sections to follow. We then evaluate the consequences of EPA's proposed action in combination with the status of the species, environmental baseline and the cumulative effects to determine whether EPA can insure that the likelihood of jeopardy or adverse modification of designated critical habitat will be avoided.

The quality of the biophysical components within aquatic ecosystems is affected by human activities conducted within and around coastal waters, estuarine and riparian zones, as well as those conducted more remotely in the upland portion of the watershed. Industrial activities can result in discharge of pollutants, changes in water temperature and levels of dissolved oxygen, and the addition of nutrients. In addition, forestry and agricultural practices can result in erosion, run-off of fertilizers, herbicides, insecticides or other chemicals, nutrient enrichment and alteration of water flow. Chemicals such as chlordane, DDE, DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the river bottom and are later consumed by benthic feeders, such as macroinvertebrates, and then work their way higher into the food web (e.g., to sturgeon and sea turtles). Some of these compounds may affect physiological processes and impede a fish's ability to withstand stress, while simultaneously increasing the stress of the surrounding environment by reducing dissolved oxygen, altering pH, and altering other physical properties of the water body. Coastal and riparian areas are also heavily impaired by development and urbanization resulting in storm water discharges, non-point source pollution and erosion.

Climate change will extend growing seasons and spatial extent of arable land in temperate and northern biomes. This would be accompanied by changes land use and pesticide application patterns to control pests (Kattwinkel et al. 2011). However modeling results indicate that predictions of mean trends in pesticide fate and transport is complicated by case specific and location specific conditions (Gagnon et al. 2016). Hellmann et al. (2008) described the consequences for climate change on the effectiveness of management strategies for invasive species. Such species are expected become more vigorous in areas where they had previously been limited by cold or ice cover. Increased vigor would make making mechanical control less effective and pesticide use likely. Some plant species may become more tolerant of herbicides due to elevated CO2. Pesticide fate and transport, toxicities, degradation rates, and the effectiveness of biocontrol agents are expected to change with changing temperature and water regimes, driven largely by effects on rates in organism metabolism and abiotic reactions (Bloomfield et al. 2006, Schiedek et al. 2007, Noyes et al. 2009).

7.1.1 Water Quality

This section describes the current status and recent health trends of aquatic ecosystems within the *Action Area*. EPA sampling results (USEPA 2015) are summarized by region for the following biological, chemical, and physical indicators: 1) Biological – benthic macroinvertebrates; 2) Chemical – phosphorous, nitrogen, ecological fish tissue contaminants, sediment contaminants, sediment toxicity, and pesticides; and 3) Physical – dissolved oxygen, salinity, water clarity, pH, and Chlorophyll a. Cumulatively, these biological, chemical, and physical measures provide an overall picture of the ecological condition of aquatic ecosystems. Different thresholds, based on published references and the best professional judgment of regional experts, are used to evaluate each region as “good,” “fair,” or “poor” for each water quality indicator. EPA rates overall water quality from results of the five key indicators using the following guidelines: “poor” – two or more component indicators are rated poor; “fair” - one indicator is rated poor, or two or more are rated fair; “good” - no indicators are rated poor, and a maximum of one is rated fair.

Benthic macroinvertebrates (e.g., worms, mollusks, and crustaceans) inhabiting the bottom substrates of aquatic ecosystems are an important food source for a wide variety of fish, mammals, and birds. Benthic communities serve as reliable biological indicators of environmental quality because they are sensitive to chemical contamination, dissolved oxygen stresses, salinity fluctuations, and sediment disturbances. A good benthic index rating means that benthic habitats contain a wide variety of species, including low proportions of pollution-tolerant species and high proportions of pollution-sensitive species. A poor benthic index rating indicates that benthic communities are less diverse than expected and are populated by more pollution-tolerant species and fewer pollution-sensitive species than expected.

Chemical and physical components are measured as indicators of key stressors that have the potential to degrade biological integrity. Some of these are naturally occurring and others result only from human activities, but most come from both sources. EPA evaluates overall water quality based on the following primary indicators: surface nutrient enrichment—dissolved inorganic nitrogen and dissolved inorganic phosphorus concentrations; algae biomass—surface chlorophyll a concentration; and potential adverse effects of eutrophication—water clarity and bottom dissolved oxygen levels (USEPA 2015). Contaminants, including some pesticides, PCBs and mercury, also contribute to ecological degradation. Many contaminants adsorb onto suspended particles and accumulate in areas where sediments are deposited and may adversely affect sediment-dwelling organisms. As other organisms eat contaminated sediment-dwellers the contaminants can accumulate in organisms and potentially become concentrated throughout the food web.

Northeast Region (Maine to Virginia)

A wide variety of coastal environments are found in the Northeast region including rocky coasts, drowned river valleys, estuaries, salt marshes, and city harbors. The Northeast is the most populous coastal region in the U.S. In 2010, the region was home to 54.2 million people, representing about a third of the nation's total coastal population (USEPA 2015). The population in this area has increased by ten million residents (~ 23 percent) since 1970. The coast from Cape Cod to the Chesapeake Bay consists of larger watersheds that are drained by major riverine systems that empty into relatively shallow and poorly flushed estuaries. These estuaries are more susceptible to the pressures of a highly populated and industrialized coastal region.

A total of 238 sites were sampled to assess approximately 10,700 square miles of Northeast coastal waters. Figure 4 shows a summary of findings from the EPA's National Coastal Condition Assessment Report for the Northeast Region (USEPA 2015). Biological quality is rated as good in 62 percent of the Northeast coast region based on the benthic index. Poor biological conditions occur in 27 percent of the coastal area. About 11 percent of the region reported missing results, due primarily to difficulties in collecting benthic samples along the rocky coast north of Cape Cod. Based on the water quality index, 44 percent of the Northeast coast is in good condition, 49 percent is rated fair, and 6 percent is rated poor.

Based on the sediment quality index, 60 percent of the Northeast coastal area sampled is in good condition, 20 percent is in fair condition, and 9 percent is in poor condition (11 percent were reported "missing"). Compared to ecological risk-based thresholds for fish tissue contamination, less than 1 percent of the Northeast coast is rated as good, 27 percent is rated fair, and 33 percent is rated poor. Researchers were unable to evaluate fish tissue for 39 percent of the region, including almost the entire Acadian Province, because target species were not caught for analysis. The contaminants that most often exceed the thresholds for a "poor" rating in the assessed areas of the Northeast coast are selenium, mercury, arsenic, and, in a small proportion of the area, total PCBs.

New Hampshire conducted site specific water quality assessments on 42 percent of rivers, 81 percent of aquatic estuarine waters, and 85 percent of ocean waters within the state. Results reported in the New Hampshire 2012 Surface Water Quality Report indicate that approximately 0.8 percent of freshwater rivers and stream mileage is fully supportive of aquatic life, 26.0 percent is not supportive, and 73.2 percent could not be assessed due to insufficient information (NHDES 2012). In estuarine waters, approximately 0.8 percent of the square mileage is fully supportive of aquatic life, 91.9 percent is not supportive and 7.2 percent could not be assessed due to insufficient information. Twenty-six percent of estuarine waters fully met the water quality standards, 54 percent were impaired, and 19 percent could not be assessed due to insufficient information. In ocean waters, approximately 94.1 percent of the square mileage is fully supportive of aquatic life, 0.0 percent is not supportive and 5.9 percent could not be assessed due to insufficient information (NHDES 2012). Fifty-six percent of ocean waters fully met the water quality standards, 29 percent were impaired, and 15 percent could not be assessed due to insufficient information.

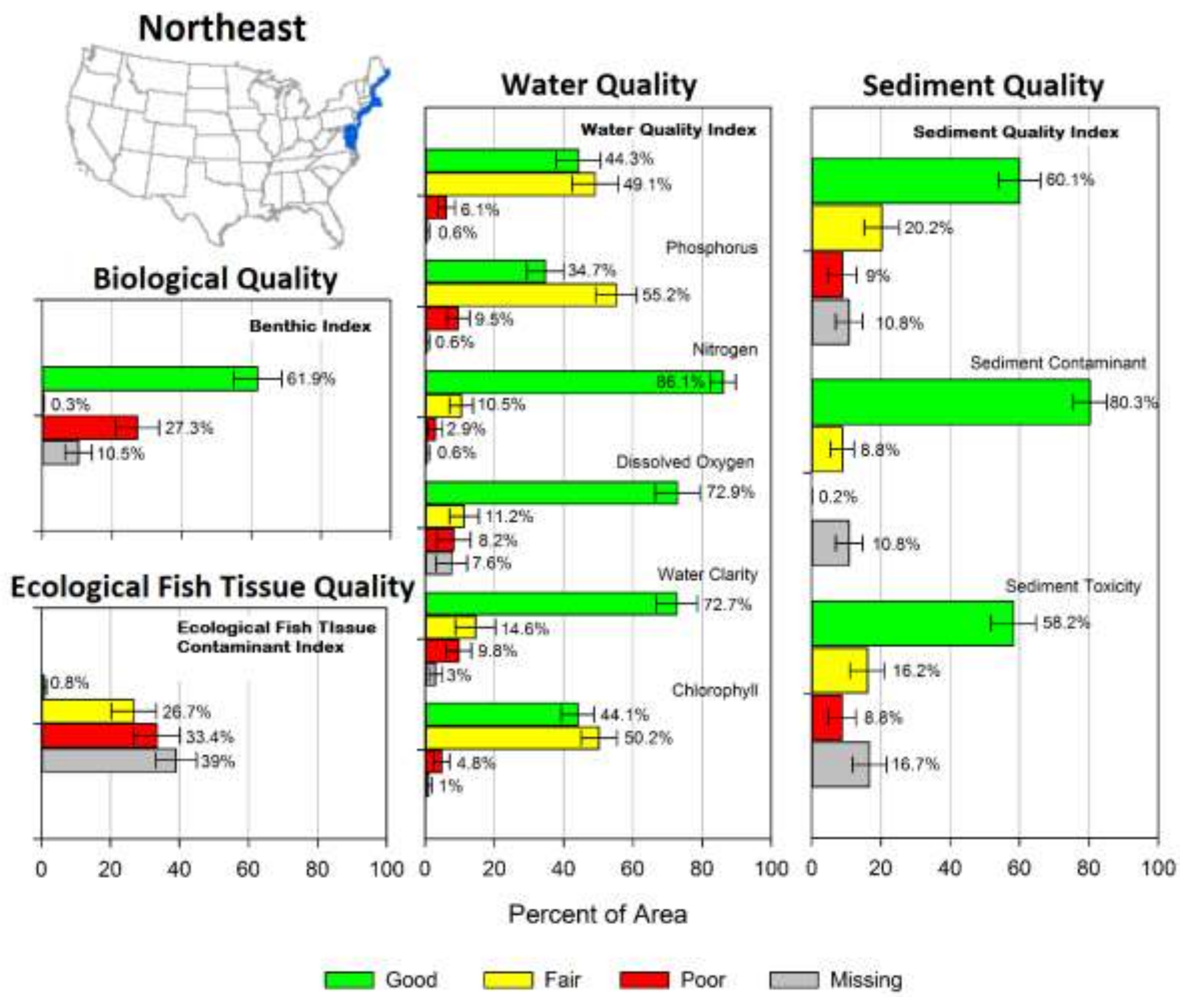

Figure 4. National Coastal Condition Assessment 2010 Report findings for the Northeast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 238 sites sampled). Error bars represent 95 percent confidence levels (USEPA 2015).

All of New Hampshire waters are impaired by mercury contamination in fish tissue, with the source being atmospheric deposition. All of New Hampshire's bays and estuaries are impaired by dioxins and PCBs. The top five reasons for impairment in New Hampshire rivers for 2012 were: mercury (16,962 acres), pH (3,821 acres), E coli (1,306 acres), dissolved oxygen (688 acres), and aluminum (563 acres) (NHDES 2012). The top five reasons for impairment in New Hampshire estuaries for 2012 were: mercury (18 acres), dioxin (18 acres), PCBs (18 acres), estuarine bioassessments (15 acres), and nitrogen (14 acres). The top five reasons for impairment in New Hampshire ocean waters for 2012 were: PCBs (81 acres), mercury (81 acres), dioxin (81 acres), Enterococcus (0.5 acres), and fecal coliform (0.5 acres). Besides atmospheric deposition, sources of impairment in New Hampshire include forced drainage pumping, waterfowl, domestic wastes, combined sewer overflows, animal feeding operations, municipal sources, and other unknown sources (NHDES 2012).

Violation rates among EPA- permitted pollutant sources are relatively low in New Hampshire. A total of 386 (1.7 percent) of 23,192 permitted facilities are in violation of their permits, and only 58 (0.25 percent) of these violations are classified as a significant noncompliance. Of the 254

NPDES permits in New Hampshire, 28 currently have effluent violations and five of these are classified as significant noncompliance.

In 2012, Massachusetts assessed the condition of 2,816 miles (28 percent) of the state's rivers and streams and found 63 percent to be impaired[12]. Four out of the top five impairment causes for rivers and streams in Massachusetts are attributed to pathogens and nutrients. The probable sources for these impaired waters include unknown sources, municipal discharges and unspecified urban stormwater. The distribution of impairment causes and probable sources suggest that eutrophication is a factor in Massachusetts rivers and stream impairments. PCBs in fish tissue from legacy sediment contamination is identified as a contributing factor in 14 percent of assessed river or stream miles. Both invasive species and atmospheric mercury deposition are major contributors to impairments of lakes, reservoirs and ponds. Nearly the entire spatial area of Massachusetts' bays and estuaries were assessed (98 percent of 248 square miles), with 87 percent found to be impaired. Fecal coliform contamination from municipal discharges impair the entire extent of assessed bays and estuaries. PCBs in fish tissue are also a significant factor, occurring in 36 percent of assessed waters. The impairment classification "other cause" is identified in 27 percent of estuaries and bays. This reporting category is used for dissolved gases, floating debris and foam, leachate, stormwater pollutants, and many other uncommon causes lumped together. Among sources for pollutants, stormwater was a major factor for Massachusetts estuaries and bays as three of the top five identified sources of impairments are discharges from municipal separate storm sewer systems (53 percent of impaired area), wet weather discharges (27 percent) and unspecified urban stormwater (25 percent). Among the 29,788 discharge-permitted facilities located in Massachusetts, 956 (3 percent) are in violation, with 115 (0.39 percent) of these violations classified as a significant noncompliance. NPDES permits are held by 833 of these facilities. Effluent violations are identified at 77 of these facilities, with 33 violations classified as in significant noncompliance.

In 2014, the District of Columbia (D.C.) assessed the condition of 98.5 percent of its 39 miles of rivers and streams and 99 percent of its 6 square miles of bays and estuaries[13]. All waters assessed were found to be impaired by PCBs. By impairment group, pesticides accounted for the most causes for impairment for 303(d) listed waters assessed in D.C. The following pesticides were identified as causes for impairment in D.C. rivers/streams and bays/estuaries: heptachlor epoxide (21.9 miles), dieldrin (21.9 miles), chlordane (21.1 miles), DDT (19.4 miles), DDD (16.2 miles), and DDE (16.2 miles). Out of 2,729 facilities with pollutant-source permits in D.C., 48 permits (1.8 percent) are in violation, with three classified as significant noncompliance. Among the twenty-eight NPDES permits in D.C., two had effluent violations (7 percent), but none of the effluent violations were classified as a significant noncompliance.

The remaining East coast portion of the *Action Area* is very small. It includes Tribal and federal lands within 24 subwatersheds distributed among Maine, Vermont, Connecticut, and Delaware. Although 13 of these are in Maine, few river and stream aquatic impairments are reported in this state (8 out of 250 total assessed water bodies are impaired). Impairment causes in Maine are identified as low dissolved oxygen and dioxins. Microbial pollution of rivers and streams are indicated as major impairment causes in Vermont, Connecticut and Delaware, accounting for nearly 60 percent of the impaired river and stream miles among these states (EPA Water Quality

[12] MA 2014 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=MA

[13] DC 2014 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=DC

Assessment and TMDL Information, https://iaspub.epa.gov/waters10/attains_index.home). Mercury, arsenic pollution and “unknown” are also among the top impairment causes for rivers and streams in these states. None of the 35 federally operated permitted facilities in Delaware and Vermont or the six facilities on Tribal land in Connecticut have permit violations (NMFS 2015a). The 9 facilities located in Maine include 5 with violations, 4 of which are classified as a significant noncompliance. There are no NPDES permits for sub-watersheds of Maine or Vermont within the *Action Area*. The single NPDES permitted facility in the Delaware portion of the *Action Area* is currently in compliance with its permit.

West Coast Region

The West Coast region contains 410 estuaries, bays, and sub-estuaries that cover a total area of 2,200 square miles (USEPA 2015). More than 60 percent of this area consists of three large estuarine systems—the San Francisco Estuary, Columbia River Estuary, and Puget Sound (including the Strait of Juan de Fuca). Sub-estuary systems associated with these large systems make up another 27 percent of the West Coast. The remaining West Coast water bodies, combined, compose only 12 percent of the total coastal area of the region.

The majority of the population in the West Coast states of California, Oregon, and Washington lives in coastal counties. In 2010, approximately 40 million people lived in these coastal counties, representing 19 percent of the U.S. population residing in coastal watershed counties and 63 percent of the total population of West Coast states (U.S. Census Bureau, http://www.census.gov/2010census/). Between 1970 and 2010, the population in the coastal watershed counties of the West Coast region almost doubled, growing from 22 million to 39 million people.

A total of 134 sites were sampled to characterize the condition of West Coast waters. Figure 5 shows a summary of findings from the EPA’s National Coastal Condition Assessment Report for the west Coast Region (USEPA 2015).

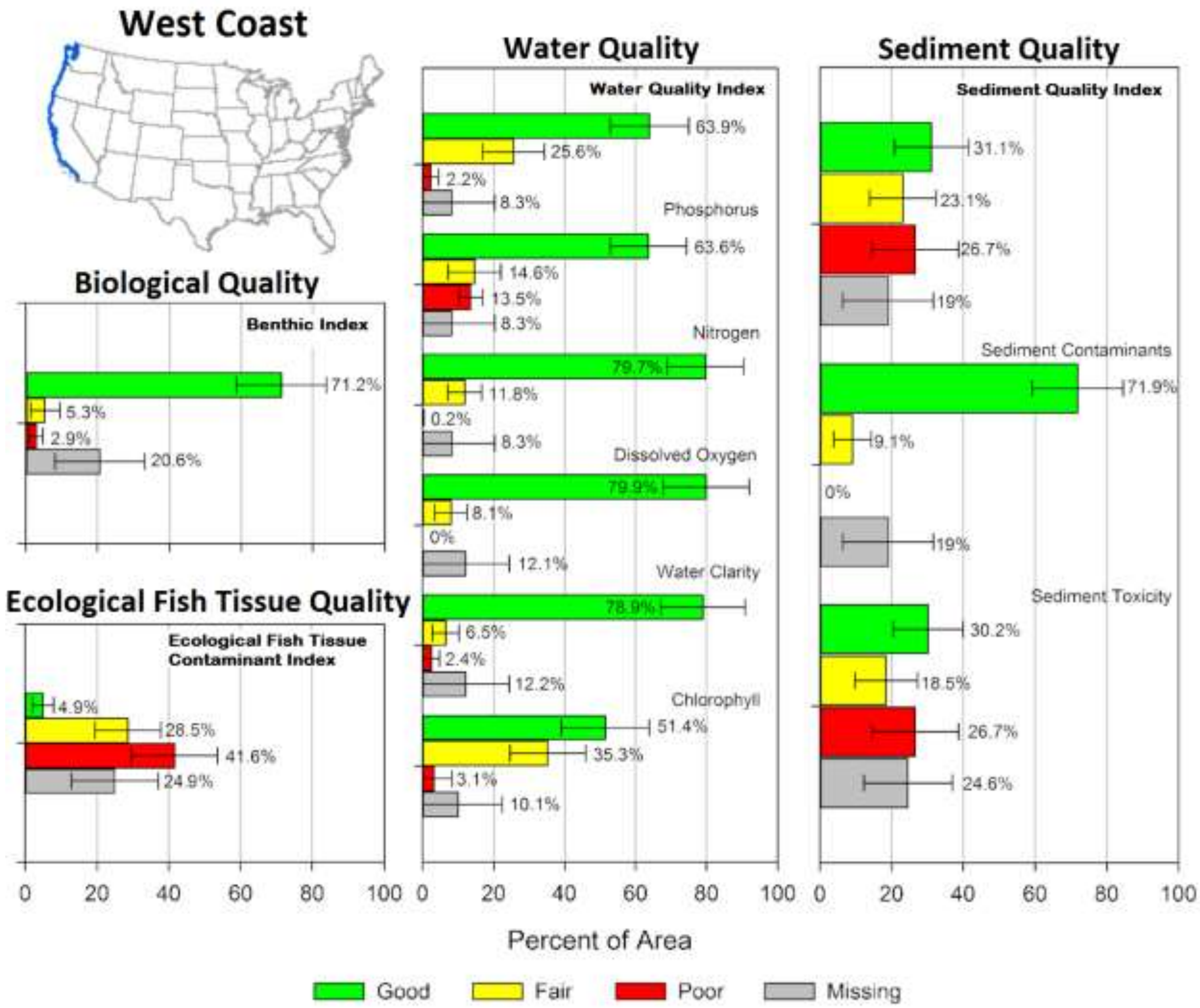

Figure 5. National Coastal Condition Assessment 2010 Report findings for the West Coast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 238 sites sampled). Error bars represent 95 percent confidence levels (USEPA 2015).

Biological quality is rated good in 71 percent of West Coast waters, based on the benthic index. Fair biological quality occurs in 5 percent of these waters, and poor biological quality occurs in 3 percent (data are missing for an additional 21 percent of waters due to difficulty obtaining samples). Based on the water quality index, 64 percent of waters in the West Coast region are in good condition, 26 percent are rated fair, and 2 percent are rated poor (USEPA 2015).

Based on the sediment quality index, 31 percent of West Coast waters sampled are in good condition, 23 percent in fair condition, and 27 percent in poor condition (data missing for 19 percent of waters sampled) (USEPA 2015). Based on the ecological fish tissue contaminant index, 42 percent of West Coast waters are in poor condition, 29 percent in fair condition, and 5 percent in good condition (data missing for 25 percent of waters sampled). The contaminants that most often exceed the thresholds for “poor” condition are selenium, mercury, arsenic, and, in a very small proportion of the area, hexachlorobenzene (USEPA 2015).

Subwatersheds associated with Washington State federal lands where PGP eligible activities may occur (e.g., Department of Defense, Bureau of Land Management, Bureau of Reclamation) or Tribal lands, are distributed throughout the state and along the coast line. Information from the 2008 state water quality assessment report for the entire state was used to infer conditions within the *Action Area*. For the 2008 reporting year, the state of Washington assessed 1,997 miles of rivers and streams, 434,530 acres of lakes, reservoirs, and ponds, and 376 square miles of ocean and near coastal waters (Washington 2008 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=WA). Among assessed waters, 80 percent of rivers and streams, 68 percent of lakes, reservoirs, and ponds, and 53 percent of ocean and near coastal waters were impaired. Temperature (39 percent of assessed waters) and fecal coliform (32 percent of assessed waters) are prominent causes of impairments. These are followed by low dissolved oxygen (19 percent), pH (9 percent), and instream flow impairments (2 percent). Ocean and near coastal impairment causes include fecal coliform in 17 percent of assessed waters, followed by low dissolved oxygen in 12 percent of these waters. The remaining contributors are invasive exotic species, sediment toxicity, and PCBs.

Among the 485 facilities located within Washington's Tribal lands, 67 are in violation of their permits, with 7 of these violations classified as a significant noncompliance (NMFS 2015a). There are 349 NPDES permits within the *Action Area*, but only two of these facilities have effluent violations. There are no violations reported for the 11 EPA permitted facilities within the watersheds associated with federally operated facilities in Washington. Three of these permits are NPDES permits.

The area covered by subwatersheds within Tribal lands in Oregon where EPA has permitting authority account for only 1.5 percent of the *Action Area*. Direct examination of these areas using EPA's geospatial databases from 2006 indicate that 80 percent of the 376 km of rivers and streams assessed are impaired by elevated iron (NMFS 2015a). While the source of the iron is not identified, iron contamination can result from acid mine drainage. Eleven out of the 13 assessed lakes, reservoirs, and ponds in subwatersheds associated with these lands are impaired, with causes listed as temperature and fecal coliform bacteria. This amounts to impairment of 93 percent of the assessed area.

EPA also has permitting authority for Tribal lands in California. The subwatersheds associated with these lands account for about 6 percent of the total *Action Area*, but are dispersed widely and make up a very small fraction of the watersheds within the state. As such, we did not make generalizations about water quality in these areas based on the 2010 statewide water quality assessment report. Rather, information for the relevant watersheds was extracted from EPA Geospatial databases and analyzed separately. Seventy nine percent of the assessed rivers and streams within these Tribal land subwatersheds are impaired by nutrients, aluminum, arsenic, temperature, and chlordane (NMFS 2015a). Stressor sources are attributed to unknown sources, municipal point discharges, agriculture, natural background, and loss of riparian habitat. High impairment rates (93 percent) are also found for assessed lakes, reservoirs and ponds within the *Action Area* in California (NMFS 2015a). The predominant impairment for these waters is arsenic, affecting 45 percent of assessed waters, while mercury is a factor in about 9 percent of assessed waters. Arsenic is also the identified cause of impairment in 97 percent of assessed bays and estuaries (NMFS 2015a). Among the 204 facilities located in the California *Action Area*, a total of 25 facilities are in violation of their NPDES, Clean Air Act, or Resource Conservation and Recovery Act permit, with 2 of these violations classified as a significant noncompliance. The single NPDES permit listed among these permits is in compliance (NMFS 2015a).

Puerto Rico

Since the ESA-listed species and designated critical habitat under NMFS' jurisdiction in Puerto Rico are strictly marine and do not occur in freshwaters or wetlands, this discussion will focus on water quality conditions reported for coastal shoreline and saltwater habitats. In 2014, Puerto Rico assessed the condition of 390 out of 550 miles of coastal shoreline (70.9 percent) and all 8.7 square miles of the surrounding bays and estuaries. The findings indicate that 77 percent of the coastline and 100 percent of the assessed estuaries and bays are impaired (Puerto Rico Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_index.control?p_area=PR#total_assessed_waters). TMDLs are needed in 100 percent of coastal areas sampled but none have been completed. TMDLs are needed in 58.6 percent of bay/estuary areas sampled but are completed for less than 2 percent of assessed areas. Pathogens (e.g., fecal coliform, total coliform, Enterococcus) and pathogen sources dominate the impairment profiles for all three types of assessed waters These include onsite waste water systems, agriculture, concentrated animal feed operations, major municipal point sources, and urban runoff. Coastline impairment causes include pH, turbidity and Enterococcus bacteria. Many of these impairments are attributed to sewage and urban-related stormwater runoff. Rates of noncompliance among EPA-permitted pollution sources are fairly high. Among the 10,077 facilities located in Puerto Rico, 59 percent were in violation of at least one permit in 2012, and nearly all were classified as significant noncompliance. There are 522 facilities with NPDES permits and 84 (16 percent) of these were classified as in significant violation of permit effluent limits as of 2012.

Pacific Islands

EPA has NPDES permitting authority in the Pacific islands of Guam, the Northern Marianas, and American Samoa. Because the ESA-listed species and designated critical habitat under NMFS' jurisdiction in these areas are strictly marine and do not occur in freshwaters or wetlands, this discussion will focus on water quality conditions reported for coastal shoreline and saltwater habitats.

The population of American Samoa was 55,519 in 2010. Factors such as population density, inadequate land-use permitting, and increased production of solid waste and sewage, have impaired water quality in streams and coastal waters of this U.S. territory. The total surface area of American Samoa is very small, only 76.1 sq. miles, which is divided into 41 watersheds with an average size of 1.8 sq. miles. Water quality monitoring, along with coral and fish benthic monitoring, covers 34 of the 41 watersheds, which includes areas populated by more than 95 percent of the total population of American Samoa. For the goal to protect and enhance ecosystems (aquatic life), of the 45.1 shoreline miles (out of 149.5 total) assessed in 2012-2013, 15.5 miles were found to be fully supporting, 12.8 miles were found to be partially supporting, and 16.8 miles were found to be not supporting (Tuitele et al. 2014). For the goal to Protect and Enhance Public Health, all 7.9 shoreline miles assessed in 2012-2013 for fish consumption were found to be not supporting. Eighty four percent of American Samoa's coastline was assessed in 2010 and 60 percent of the assessed waters were found to be impaired. Enterococcus is identified as causing impairments along 50 percent of the coastline evaluated, while 26 percent of assessed coastline had nonpoint source pollutants contributing to impairments. Of the 5.7 km^2 of reef flats assessed in 2010, 76 percent were fully supporting and 24 percent were not supporting the goal of Protect and Enhance Ecosystems (Tuitele et al. 2014). The major stressors identified were PCBs, metals (mercury), pathogen indicators, and other undetermined stressors (Tuitele et al. 2014). The major sources of impairment included sanitary sewer overflows and animal feed operations, each implicated for 50 percent of the waters assessed. Multiple nonpoint sources were identified as a stressor source for 26 percent of assessed waters, while contaminated sediments contributed to impairments in 6 percent of assessed waters. Among the 204 facilities with pollutant permits, a total of 21 (10.3 percent) facilities were in violation, with 17 of these violations classified as a significant noncompliance. Of the six facilities with NDPES permits, two have violated effluent limits, one of which is considered to be in significant noncompliance.

Guam assessed 3 percent of its 915 acres of bays/estuaries and 14 percent of its 117 miles of coastline in 2010 (Guam 2010 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=GU). Impairments are identified in 42 percent of assessed bays and estuaries and the entire extent of assessed coastline. PCBs levels in fish tissue was the cause of impairment in 33 percent of assessed bays and estuaries, followed by antimony, dieldrin, tetrachloroethylene, and trichloroethylene, each listed as causing impairments to 6 percent of assessed waters. Enterococcus bacteria is the cause of impairment in nearly all of Guam's coastal shoreline waters (96 percent), while PCB contamination is a minor contributor to impairment of the coastal shoreline (4 percent). Sources of impairment causes have not been identified for Guam. Among the 403 NPDES, Clean Air Act, or Resource Conservation and Recovery Act EPA-permitted facilities located in Guam, a total of 23 (5.7 percent) facilities are in violation, with 13 of these violations classified as a significant noncompliance. NPDES permits are held by 19 facilities, six of which have effluent violations classified as significant noncompliance.

In the Northern Marianas, 36 percent of the 235.5 miles of assessed shoreline were found to be impaired in 2014 (N. Mariana Islands Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=CN). Phosphate is listed as a cause for all impaired areas. Other causes identified among the impaired stretches of shoreline include microbiological contamination from Enterococcus bacteria (22 percent), dissolved oxygen saturation levels (16 percent), and mercury in fish tissue (1 percent). The presence of Enterococci bacteria was implicated for the impairment of 32.2 miles of Saipan's, 17.8 miles of Rota's, and 24.3 miles of Tinian's shoreline for recreational uses. In addition 15 percent of the assessed waters had impaired biological assemblages. Sources of impairments included sediments (15 percent), unknown sources (13 percent), on-site septic treatment systems (12 percent), urban runoff (12 percent), and livestock operations (7 percent). We did not find any NPDES permitted facilities in the Northern Marianas.

7.1.2 Baseline Pesticide Detections in Aquatic Environments

Pesticide detections for the *Environmental Baseline* are addressed as reported in the U.S. Geological Survey (USGS) National Water-Quality Assessment Program's (NAWQA) national assessment (Gilliom 2006). This approach was chosen because the NAWQA reports provide the same level of analysis for each geographic area. In addition, given the lack of uniform reporting standards and large action area for this opinion, it is not feasible to present a comprehensive basin-specific analysis of pesticide detections.

Over half a billion pounds of herbicides, insecticides, and fungicides were used annually from 1992 to 2011 to increase crop production and reduce insect-borne disease (Stone et al. 2014)

During any given year, more than 400 different types of pesticides are used in agricultural and urban settings. The distributions of the most prevalent pesticides in streams and groundwater correlate with land use patterns and associated present or past pesticide use (Gilliom 2006). When pesticides are released into the environment they frequently end up as contaminants in aquatic environments. Depending on their physical properties, some are rapidly transformed via chemical, photochemical, and biologically mediated reactions into other compounds known as degradates. These degradates may become as prevalent as the parent pesticides depending on their rate of formation and their relative persistence. Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (Gilliom 2006). Mixtures result from the use of different pesticides for multiple purposes within a watershed or groundwater recharge area. Pesticides generally occur more often in natural water bodies as mixtures than as individual compounds. Fish exposed to multiple pesticides at once may also experience additive and synergistic effects. If the effects on a biological endpoint from concurrent exposure to multiple pesticides can be predicted by adding the potency of the pesticides involved, the effects are said to be additive. If, however, the response to a mixture leads to a greater than expected effect on the endpoint, and the pesticides within the mixture enhance the toxicity of one another, the effects are characterized as synergistic. These effects are of particular concern when the pesticides share a mode of action.

From 1992 to 2001, the USGS sampled water from 186 stream sites, bed sediment samples from 1,052 stream sites, and fish from 700 stream sites across the continental U.S. Pesticide concentrations were detected in streams and groundwater within most areas sampled with substantial agricultural or urban land uses. NAWQA results detected at least one pesticide or degradate in more than 90 percent of water samples, more than 80 percent of fish samples, and more than 50 percent of bed sediment samples from streams in watersheds with agricultural, urban, and mixed land use (Gilliom 2006). Compounds commonly detected included 11 agriculture-use herbicides and the atrazine degradate deethylatrazine; 7 urban-use herbicides; and 6 insecticides used in both agricultural and urban areas. Mixtures of pesticides were detected more often in streams than in ground water and at relatively similar frequencies in streams draining areas of agricultural, urban, and mixed land use. Water from streams in these developed land use settings had detections of two or more pesticides or degradates more than 90 percent of the time, five or more pesticides or degradates about 70 percent of the time, and 10 or more pesticides or degradates about 20 percent of the time (Gilliom 2006). NAWQA analysis of all detections indicates that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams (Gilliom 2006). The number of unique mixtures varied with land use. More than half of all agricultural streams and more than three-quarters of all urban streams sampled had concentrations of pesticides in water that exceeded one or more benchmarks for aquatic life. Exceedance of an aquatic life benchmark level indicates a strong probability that aquatic species are being adversely affected. However, aquatic species may also be affected at levels below benchmark criteria. In agricultural streams, most concentrations that exceeded an aquatic life benchmark involved chlorpyrifos (21 percent), azinphos methyl (19 percent), atrazine (18 percent), DDE (16 percent), and alachlor (15 percent) (Gilliom 2006). Organochlorine pesticides that were discontinued 15 to 30 years ago still exceeded benchmarks for aquatic life and fish-eating wildlife in bed sediment or fish tissue samples from many streams.

Stone et al. (2014) compared pesticide levels for streams and rivers across the conterminous U.S. for the decade 2002−2011 with previously reported findings from the decade of 1992−2001. Overall, the proportions of assessed streams with one or more pesticides that exceeded an aquatic life benchmark were very similar between the two decades for agricultural (69 percent during 1992−2001 compared to 61 percent during 2002−2011) and mixed-land-use streams (45 percent compared to 46 percent). Urban streams, in contrast, increased from 53 percent during 1992−2011 to 90 percent during 2002−2011, largely because of fipronil and dichlorvos. Agricultural use of synthetic organic herbicides, insecticides, and fungicides in the continental U.S. had a peak in the mid-1990s, followed by a decline to a low in the mid-2000s (Stone et al. 2014). During the late-2000s, overall pesticide use steadily increased, largely because of the rapid adoption of genetically modified crops and the increased use of glyphosate. The herbicides that were assessed by USGS represent a decreasing proportion of total use from 1992 to 2011 because glyphosate was not previously included in the national monitoring network.

EPA has consulted with NMFS under Section 7(a)(2) of the ESA on the registration of several pesticides with respect to their effects on ESA-listed Pacific salmonids and designated critical habitat under NMFS' jurisdiction[14]. These consultations evaluated pesticides registered for use under one of the four use patterns covered under the PGP. In many cases, these consultations concluded that EPA's re-registration and subsequent use of these pesticides according to the registered labels jeopardize the continued existence and/or adversely modify designated critical habitat for these species. The use of these pesticides for non-PGP use patterns are part of the baseline for ESA-listed salmonids, and, as agricultural uses are ongoing and not subject to consultation, agricultural uses of these pesticides are part of the cumulative effects as well. This series of consultations are listed in Table 15 of the *Risk Characterization* of this opinion.

8 EFFECTS OF THE ACTION

Section 7 of the ESA regulations define "effects of the action" as the direct and indirect effects of an action on the species or designated critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 CFR 402.02). Indirect effects are those that are caused by the proposed action and are later in time, but are reasonably certain to occur. This includes effects on prey resources and "legacy effects" of the action, such as the redistribution of pollutants by stormwater or disturbed sediment and maternal or dietary transfer of accumulated toxicants.

To evaluate the effects in this opinion, we conduct a risk assessment (Section 8.1) in which we consider the likelihood of exposure to the stressors of the action of individuals of species and essential features of designated critical habitat and the potential for adverse responses. We then integrate the information to characterize the risk of adverse effects to identified environmental values, referred to as assessment endpoints. In this risk assessment section, we analyze the risks posed by the discharges without consideration of EPA's decision-making process or protective control measures in the PGP to minimize or prevent adverse effects. We evaluate EPA's process to determine the effectiveness of the PGP program in a programmatic analysis (Section 8.2)..

The programmatic analysis evaluates the decision-making process and the protective control measures EPA intends to establish to protect ESA-listed species or designated critical habitat from the adverse direct or indirect effects of the activities authorized by the PGP. As part of this analysis, we analyze the past performance of the PGP and in individual and general permits that

[14] NMFS Pesticide Consultations with EPA, http://www.nmfs.noaa.gov/pr/consultation/pesticides.htm

the EPA has issued and consider the performance of those controls as indicative of how well the controls of the PGP are likely to work. For many programmatic consultations, the action agency has structured the program so that neither species nor designated critical habitat are exposed to the stressors of the action until there is a separate ESA section 7 consultation addressing site specific activities that will result in exposure. However, in this instance, EPA intends to authorize a large number of discharges without subsequent ESA section 7 consultations, except for those discharges that do not qualify for coverage under the general permit and for which the discharger must seek an individual permit. Accordingly, if there is overlap with species, EPA's programmatic action will result in exposure of species and designated critical habitat to the action.

8.1 Risk Assessment

In the risk assessment portion of this consultation we were concerned with the potential adverse effects of discharges of pesticides under the four use patterns eligible for coverage under the PGP on ESA-listed species and designated critical habitat under NMFS' jurisdiction. Due to the scope and complexity of the action and the uncertainty regarding the type and location of discharges that will actually occur, this analysis applies a qualitative strength of evidence assessment of risks. As noted above, this risk assessment portion considers the effects to adverse endpoints resulting from pesticide discharges without consideration of the effectiveness of EPA's program in minimizing or preventing risk.

The risk assessment portion integrates elements of EPA's ecological risk assessment framework (ERA-Framework, USEPA 1998) into NMFS' assessment approach. The risk assessment is organized in three phases:

1) Problem formulation examines the stressors of the action, the action area, its environmental baseline, and the status of the species and designated critical habitat in order to formulate risk hypotheses[15] on how species may respond to exposures to the stressors of the action. Risk hypotheses organize the analysis by positing the relationships among exposure to stressors, response to stressors, and environmental values, referred to as assessment endpoints. Once the risk hypotheses are formulated, the analyses proceed through a exposure → response → risk characterization path. A risk hypothesis is disproved when there is little or no likelihood of adverse effects to the assessment endpoints, and no further analysis of that hypothesis is merited in the opinion.
2) The exposure and response analysis evaluates how individuals of species and essential features of designated critical habitat may be affected and determines whether stressor exposures would result in adverse responses representing the assessment endpoints. For example, reduced number of viable eggs would represent an effect to the assessment endpoint reduced fecundity.
3) The risk characterization considers the population-level implications of adverse responses representing the assessment endpoints to determine if these are sufficiently large to affect population parameters (e.g., assessment endpoints such as recruitment or reproductive

[15] NOTE: Moved into text – this is important. Risk hypotheses are statements that organize an analysis by describing the relationships among stressor, exposure, and the environmental values to be protected (also referred to as the assessment endpoints).

rate). Effects to the conservation value of the physical and biological features of designated critical habitat are evaluated at this point in the assessment.

8.1.1 Problem Formulation

The problem formulation integrates what is known about the status of the species and designated critical habitat (Section 6) and baseline conditions (Section 7) with the proposed action (Section 3) and the stressors resulting from that action (discussed below) to identify the types of effects that may occur as a result of the action and formulate risk hypotheses to be evaluated in the *Exposure and Response Analysis* (Section 8.1.2) and *Risk Characterization*.

Stressors of the Action

The EPA focused on the active ingredients of pesticide formulations when it registered pesticides pursuant to FIFRA. The PGP authorizes discharges for the use patterns eligible for CWA coverage under the PGP. Many of these pesticide active ingredients persist in the aquatic environment long after their intended uses (see Table 8). In addition, these active ingredients also include adjuvants, surfactants and other additives that were not evaluated in the FIFRA registration process.

The EPA permits more than 4,000 potentially hazardous additives for use in pesticide formulations. For example, nonylphenols are ingredients that may be included in the formulations of pesticide and are common wastewater contaminants from industrial and municipal sources. A national survey of streams found that nonylphenol was among the most common organic wastewater contaminants in the U.S. and was detected in more than fifty percent of the samples tested (Kolpin et al. 2002). The common pesticide additive xylene is a neurotoxin and the additive coal tar is a known carcinogen. To complicate matters, several permitted additives are also registered pesticide active ingredients.

Because other components of pesticide formulations in addition to the active ingredients may be toxic, we considered the effects of adjuvants, surfactants and other additives in the formulations of those pesticides as well as the effects of the active ingredients.

Table 8. Persistence of some commonly used pesticides in surface water and aquatic sediments (Barbash 2007).

Use Class	Chemical Class	Example	Half Life in Surface Water	Half Life in Aquatic Sediment
Herbicides	Amino acid Derivatives	Glyphosate	~2 months	~8 months
	Chlorphenoxy acid	2,4-D	~2 days	~2 months
	Triazines	Atrazine Simazine	~2 years ~3 weeks	~2 months ~8 months
	Urea	Diuron	~3 weeks	~8 months
Insecticides	Carbamates	Carbaryl	~1 week	~2 months
	Organophosphates	Chlorpyrifos Diazinon Malathion	~1 week ~2 months ~2 days	~2 months ~8 months ~3 weeks
	Pyrethroids	Permethrin	~14-21 days	~30-40 days

The 2012 through 2014 annual reports for PGP authorizations identify 260 individual pesticide products containing one or more of 101 individual active ingredients. These represent only those permit holders that were required to submit annual reports under the 2011 PGP. In areas where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur, annual reports were required for the 114 out of the 284 PGP applicants submitting an NOI under the 2011 PGP. Twenty-eight of the PGP applicants submitting an annual report had certified that ESA-listed species were present in at least a portion of their application area and twenty-one of those not required to submit an annual report certified that ESA-listed species were present in the at least a portion of their application area.

Below we described the four use patterns that would be authorized under the PGP: 1) mosquito and other flying insect pest control, 2) weed and algae pest control, 3) animal pest control, and 4) forest canopy pest control.

Mosquito and Other Flying Insect Pest Control. This use pattern includes any application of pesticides in, over or near waterbodies where these pests spend at least part of their life cycle. Applications may occur to prevent disease outbreaks or other health reasons or to support recreational activities. The variety of pesticides and formulations that are used will commonly depend on the life stage of mosquito that is being controlled.

To control larval stages, formulations of *Bacillus thuringiensis israelensis* and *B. sphaericus* are common while formulations of carbaryl, chlorpyrifos, deltamethrin, malathion and sumithrin are common to control flying adults. The Idaho Mosquito and Vector Control Association, founded by Idaho State Department of Agriculture and the State Health Department currently identifies 30 mosquito control districts with 28 active and submitting NOI under the 2011 PGP. Different formulations of *Bacillus thuringiensis* and *B. sphaericus* are applied to mosquito breeding habitats when their larvae are in the first to third instar stages of life, although some districts will also apply methoprene or temephos (in the formulation Abate®). To control flying adult mosquitoes, these districts apply ultra-low volumes of insecticides, which include a malathion-based ultra-low volume concentrate, naled, and pyrethroids[16].

The Massachusetts counties through which the Connecticut River flows, Hampden, Hampshire, and Franklin counties, are not served by a mosquito control district and there are no NOI or annual reports specifically for this area of Massachusetts under this use pattern. However, a statewide NOI was filed by the State Reclamation and Mosquito Control Board. The only annual report provided under the NOI was for 2012, reporting use of 2281 gallons of a phenothrin-piperonyl butoxide formulation. As an indicator of what may be used over the permitting period for the 2016 PGP, the Central Massachusetts Mosquito Control Project (www.cmmcp.org) reports that it applies different formulations of *Bacillus thuringiensis* and *B. sphaericus* to control mosquito larvae, also supplemented with formulations of methoprene. To control flying adult mosquitoes, ultra-low volumes of formulations of sumithrin, d-phenothrin, and etofenprox are used. Annual reports filed under 2011 PGP NOI for Bristol and Essex counties, which include waters where sturgeon occur, indicated that formulations of *Bacillus* and methoprene were applied.

The State of New Hampshire Department of Health and Human Services may use permethrin, sumithrin, and resmethrin commonly with piperonyl butoxide to enhance the insecticidal activity

[16] Idaho State Department of Agriculture: http://www.kellysolutions.com/ID/searchbyproductname.asp

of the pyrethroid by decreasing insect ability to detoxify the pesticide (NH DHHS 2008). There were 565 active ingredients for this use pattern identified in annual reports filed under the 2011 PGP. The top three pesticide classes identified were microbials (e.g., *Bacillus* spp.), the juvenile hormone mimic methoprene, and pyrethroids plus piperonyl butoxide. The only annual reports from New Hampshire for this use pattern were from the Town of Hampton, which applied a wide variety of pesticides, including formulations containing *Bacillus*, methoprene, and pyrethroids with piperonyl butoxide. The City of Portsmouth, and the towns of Newfields, Newton, and Stratham, NH all filed NOI for mosquito control under the 2011 PGP, but like many other NOI, did not identify the pesticides that were to be used. Overall, the annual reports identify the use of 24 individual active ingredients with formulations containing *Bacillus*, methoprene, or pyrethroids making up greater than 80 percent of reported applications.

Weed and Algae Pest Control. The aquatic weed and algae control pesticide use pattern includes the application of pesticides in, over or near waterbodies to control algae and other submergent or emergent nuisance aquatic plants to protect sensitive aquatic habitats and to maintain recreational uses. This is a broad use pattern covering many types of aquatic habitats.

There are a variety of formulations and application methods for this use pattern. For example, the pesticides that the EPA currently authorizes for aquatic weed and algae control in Idaho include 2,4-D, copper compounds, diquat, endothall, fluridone, glyphosate and triclopyr. Application methods include boom sprayers, spreaders, backpack sprayers and aerial applications. Applications under this use pattern include spot treatments or large-scale treatments of several acres. These applications are usually made when the target plants are present and not dormant. Because these factors can vary widely between regions and individual waterbodies, these applications may occur at any time of year.

The annual reports filed pursuant to the 2011 PGP identified 30 individual active ingredients under this use pattern. Dominant pesticides accounting for greater than 50 percent of reported applications included pyridine carboxylic acids (e.g., aminopyralid, chlopyralid) and formulations of glyphosate and 2,4-D, followed by sulfonyl ureas (e.g., chlorsulfuron, metsulfuron).

Animal Pest Control. The Animal Pest Control use pattern includes the application of pesticides in, over or near waterbodies to control a wide variety of aquatic animals. These uses include fisheries management, invasive species eradication and equipment maintenance.

Aquatic nuisance animal pests include a range of taxa including vertebrates and invertebrates such as insects, mollusks or crustaceans in a variety of aquatic habitats. Examples of the types of pesticides authorized for this use pattern include sodium chlorate and rotenone, which are currently authorized by the EPA use in Idaho. In addition, the EPA authorizes other pesticides such as antimycin-A and (3-trifluoromethyl-4-nitrophenol) (commonly known as TFM) for other areas under this use pattern. Applications are usually made over an entire waterbody and applications methods include drip-feed devices, backpack sprayers, boat bailers and aerial applications. Treatments are usually made several years apart and may occur at any time of year.

There were 25 active ingredients for this use pattern identified in annual reports filed under the 2011 PGP. The dominant pesticides, accounting for approximately 50 percent of reported applicatioms include the substituted benzene chlorthalonil, organophosphates (i.e., diazinon, chlorpyrifos, and acephate), spinosyns, indoxacarb, and azoxystrobin.

Forest Canopy Pest Control. The forest canopy pest control use pattern includes pesticide applications in and over forest canopies where these pesticides may enter waters of the U.S.. These applications usually occur over areas in response to specific pest outbreaks. Examples of such pests include gypsy moths, southern pine beetles and locust borers. This is a broad use category and covers a wide range of aquatic habitats with a variety of pesticide formulations and application methods. For example, the EPA authorizes carbaryl, chlorpyrifos and dimethoate for use in Idaho under this use pattern. Other pesticides including diflubenzuron, disparlure, malathion and trichlorfon are authorized by the EPA for forest canopy pest control in other locations. Application methods include hand sprayers, aerial applications and drip or overhead irrigation systems. These applications may occur at any time of year. Annual reports identify 16 active ingredients used for forest canopy pest control under the 2011 PGP. Pyridinecarboxylic acids (i.e., clopyralid and picloram), various formulations of 2,4-D, and the sulfonyl ureas (i.e., metsulfuron and chlorsulfuron) account for greater than 70 percent of reported applications.

Examples of Pesticides and Their Effects

Pesticides are classified according to chemical similarity and these different groups affect species through different modes of action. Some pesticides have been reported to have few, if any, adverse consequences for aquatic organisms, including endangered or threatened species. For example, despite a half-life that is estimated to be about two months in clean river water that is low in sediment, bromacil is not toxic to invertebrates and is only slightly toxic to practically non-toxic to fish. A report for the Bureau of Land Management' use of bromacil indicates that plausible worst case aquatic concentrations resulting from ground application ranged from 0.001 to 0.003 mg/L{ENSR International, 2005 #3249}. Meanwhile the 48-hour median lethal concentration where half of exposed die (LC50) for bromacil in rainbow trout is 36 mg/L, in bluegill sunfish is 127 mg/L and in sheepshead minnows is 162 mg/L (USEPA 1992). The 96-hour LC_{50} in fathead minnow is 182 mg/L (Call et al. 1987). The microbial insecticide *Bacillus thuringiensis* does not adversely affect aquatic vertebrates, including brook trout, white suckers and smallmouth bass even a month after aerial applications (Abbott Laboratories 1982), although it may adversely affect non-target invertebrates, including butterflies (Lepidoptera) (USEPA 1986b). Some chemicals have more severe consequences for organisms that are exposed to them. For example, organophosphates and carbamates inhibit acetylcholinesterase; organotins prevent the formation of adenosine triphosphate; pyrethroids keep sodium channels in neuronal membranes open, which affects the peripheral and central nervous systems and cause a hyper-excitable state; symptoms include tremors, lack of coordination, hyperactivity and paralysis; rotenone which inhibits respiratory enzymes; and limonene which affects the sensory nerves of the peripheral nervous system. The following sections summarizes the toxicity of some of these pesticide classes.

Botanicals

The botanicals include cube resins (other than rotenone) and rotenone. Rotenone is used as a fish toxin (piscicide) and is expected to be highly toxic to fish, including endangered and threatened species of fish. The assessment for rotenone in EPA's BE used modeled concentrations in ponds. Surface waters of a warm water pond are likely to reach peak concentrations of 250 μg/L, and have a predicted 21-day average of 26 μg/L and a 60-day average of 9 μg/L (USEPA, 2008). Coldwater ponds also reach a peak concentration of 250 μg/L, but show increased persistence with a 21-day average of 173 μg/L and a 60-day average of 105 μg/L. Similarly, based on a target treatment rate of 200 μg/L, surface waters of a warm water pond reach peak concentrations of 200 μg/L, and have a predicted 21-day average of 21 μg/L and 60-day average of 7 μg/L.

Meanwhile, Cheng and Farrell (2007) reported that rotenone was not toxic to juvenile rainbow trout when they were exposed at concentrations of 5.0 μg/L during 96-hour tests, but 100 percent of the juveniles died when at concentrations of 6.6 μg/L for 96 hours. Johnson and Finley (1980) reported 96-hour LC_{50} for rotenone was 23 μg/L for rainbow trout, but 2.6 μg/L for channel catfish. Finlayson et al. (2010) exposed rainbow trout for 4 and 8 hours to concentrations of synergized and non-synergized formulations of rotenone. Exposing rainbow trout to a CFT Legumine formulation of rotenone at 5.3 μg/L for an average of eight hours killed half of the rainbow trout. Exposure to a Nusyn-Noxfish formulation of rotenone at 6.2 μg/L for an average of 8 hours also killed half of the rainbow trout.

In addition, populations of aquatic invertebrates have been eliminated in streams that have been treated with rotenone. Binns (1967) reported that aquatic invertebrate populations in the Green River, Wyoming were almost completely eliminated following rotenone treatments. Mangum and Madrigal (1999) reported that the richness of Ephemeroptera in the Strawberry River in north eastern Utah had been reduced by 67-100 percent, Plecoptera by 67-100 percent and Trichoptera by 61-100 percent after two rotenone treatments, of 3 mg/L for 48 hours. In Great Basin National Park, rotenone treatments reduced species in these taxa by 99 percent for one month. More recently, a study of effects to non-target invertebrate taxa at exposures representative of rotenone use in river systems determined that invertebrate species whose breathing structures have membranes specific for gas exchange, and gill-like lamellae were more vulnerable to rotenone than species with different breathing structures (e.g. the "plastron breathers" *A. imperator* and *D. capensis*) (Dalu et al. 2015).

Carbamates

The carbamates whose uses would be authorized by the proposed PGP include carbaryl, asulam and sodium salt. Numerous authors have studied and reported the responses of vertebrate species exposed to carbamates (Zinkl et al. 1977, Shea and Berry 1983, Hanazato 1991, Sharma et al. 1993, Beyers et al. 1994, Beyers and Sikoski 1994, Relyea and Mills 2001, Relyea 2004, Boran et al. 2007, Davidson and Knapp 2007). Carbaryl, which is also known by the trade name Sevin, is an example of the group known as N-methyl carbamates, which includes other pesticides like carbofuran and methomyl. These chemicals act as neurotoxicants by impairing nerve cell transmission in vertebrates and invertebrates; specifically, they interfere with normal nerve transmissions and, as a result, can affect a wide array of physiological systems. Organophosphates have the same mode of action and produce similar physiological responses.

From the BE, based on a target application rate of 1 lb AI/acre, 2 applications with a 7 day interval, EPA's model predicts surface water concentrations of carbaryl of 11.5 μg/l for peak, 8.2 μg/l for the 21-day average, and 4.2 μg/l for the 60-day average (USEPA, 2007). Beyers and Sikoski (1994) studied the toxicity of technical carbaryl (1-napthyl methylcarbamate, 99 percent) and Sevin-4-Oil (a formulation containing 49 percent carbaryl and petroleum distillates) to Federally endangered Colorado squawfish (*Ptychocheilus lucius*) and bonytail (*Gila elegans*). In Colorado squawfish, median lethal concentrations for technical carbaryl were 1.31 mg/L (95 percent confidence interval: 1.23-1.40 mg/L) and were 3.18 mg/L (95 percent confidence interval: 2.87-3.52 mg/L) for Sevin-4-Oil. In bonytail, median lethal concentrations for technical carbaryl were 2.02 mg/L (95 percent confidence interval: 1.78 -2.25 mg/L) and were 3.31 mg/L (3.06,-3.55 mg/L) for Sevin-4-Oil. Because Colorado squawfish and bonytail are about as sensitive to carbaryl as cutthroat trout (*Oncorhynchus clarki*), rainbow trout, Atlantic salmon

(*Salmo salar*) and brook trout (*Salvelinus fontinalis*), these results should also be applicable to ESA-listed Atlantic salmon and listed steelhead (Beyers and Sikoski 1994).
Carlson (1972) exposed fathead minnows to five treatments of carbaryl (8, 17, 62, 210 and 680 µg/L) in a flow through system for nine months; capturing the life cycle of the species. Fathead minnows showed reduced number of eggs per female and reduced number of eggs spawned when exposed to 680 µg/L; none of the eggs that were spawned hatched. Zinkl et al. (1987) reported that carbaryl killed rainbow trout when they were exposed to concentrations at or above 1,000 µg/L for as few as 90 minutes. In this same study, trout exposed to concentrations of 250 – 4,000 µg/L for 24 hours exhibited 61 to 91 percent AChE inhibition.

Exposure to carbaryl appears to make cutthroat trout more susceptible to predation, perhaps by inhibiting AChE activity in brain and muscle. Cutthroat trout experienced higher predation rates when exposed to carbaryl at concentrations of 200 µg/L, 500 µg/L and 1,000 µg/L. At 200 µg/L, an increase in predation was evident (Labenia et al. 2007). Little et al. (1990) reported similar results from their studies of the effects of exposing rainbow trout fry (0.5-1.0 g) to carbaryl at 10, 100 and 1,000 µg/L for 96 hours. At all of these exposure concentrations, significantly more rainbow trout were consumed compared with unexposed fish. At concentrations of 1,000 µg/L, exposed rainbow trout fry experienced significant reductions in swimming capacity, swimming activity, prey strike frequency, daphnids consumed, percent consuming daphnids and percent survival from predation.

Organophosphates

The organophosphates include acephate, chlorpyrifos, diazinon, dichlorvos, dimethoate, malathion, naled, temephos, trichlorfon and triclorfon. Like carbamates, these chemicals act as neurotoxicants by impairing nerve cell transmission in vertebrates and invertebrates; specifically, they interfere with normal nerve transmissions and, as a result, can affect a wide array of physiological systems.

Chlorpyrifos is highly toxic to freshwater fish, aquatic invertebrates and estuarine and marine organisms. According to the BE, the application rate may be as high as 0.025 lb per acre. The modeled Estimated Environmental Concentrations (EEC) assumed that 10 percent of the applied rate may drift to surface water resulting in concentrations of 1.5 – 18.5 µg/L chlorpyrifos in surface water at depths of six inches to six feet. The EPA (1989) reported that application of concentrations as low as 0.01 pounds of active ingredient per acre may cause fish and aquatic invertebrate deaths. The 96-hour LC50 for chlorpyrifos is 0.009 mg/L in mature rainbow trout, 0.098 mg/L in lake trout, 0.806 mg/L in goldfish, 0.01 mg/L in bluegill sunfish and 0.331 mg/L in fathead minnow (USEPA 1986a). Therefore, mature rainbow trout exposed to chlorpyrifos concentrations produced by application rates of 0.025 lbs of chlorpyrifos per acre would be expected to have a 50 percent probability of dying after 96 hours of exposure (alternatively, we would expect about half of an exposed population of rainbow trout to die as a result of their exposure to these concentrations of chlorpyrifos for 96 hours).

When fathead minnows were exposed to Dursban (a formulation of chlorpyrifos) growth was reduced within 30 days at 2.68 micrograms/liter and within 60 days at 1.21 µg/L. The maturation rate of first-generation fish was reduced at all Dursban exposure concentrations and reproduction was significantly reduced at concentrations of at least 0.63 micrograms/liter. Growth rates and estimated biomass of 30-day-old second-generation fish were significantly reduced when they were exposed at concentrations of 0.12 micrograms/liter (Jarvinen et al. 1983). Carp (*Cyprinus carpio*) fingerlings exposed to concentrations of chlorpyrifos ranging from 0.120 to 0.200 mg/L for 96 hours had acute toxicities at concentrations of 0.160 mg/L. When these carp were exposed for 1, 7 and 14 days at concentrations of 0.0224 mg/L and 0.0112 mg/L, they exhibited irregular, erratic and darting swimming movements, hyper-excitability and loss of equilibrium and sinking to the bottom. Caudal bending was also reported during exposures (Halappa and David 2009).

Diazinon exposures have been implicated in five fish kills reported in California since 2002. One of these fish kills occurred in June 2002 and consisted of 2,000 salmon that were found dead in the Tembladera Slough and the Old Salinas River channel in Monterey County, California. Monterey County Agricultural Commissioner staff indicated that a small number of applications of diazinon had been made in the general area when the fish kill occurred. Water samples collected from the sites detected diazinon in four of six samples with concentrations ranging from 0.095 – 0.183 µg/L. Gill samples from all five fish showed recent exposure to chlorpyrifos with concentrations ranging from 5 - 40 µg/kg. Methidathion, another organophosphate, was also detected at low concentrations in the water but was absent in gill tissue. Although concentrations of diazinon in the water column were well below median lethal concentrations for fish that had been observed in the laboratory, peak concentrations probably had not been detected because diazinon concentrations had probably dissipated in the few days between the occurrence of the fish kill and sampling.

The EPA's BE evaluated calculated EECs for diazinon in surface water resulting from the highest application rate on crop types chosen from pesticide usage data from 1992 - 1997. Based on a target application rate of 3.0 pounds AI per acre (almonds), EPA's model predicts surface water concentrations of diazinon of 8.89 ppb for peak, 7.94 ppb for the 21-day average, and 6.39 ppb for the 60-day average. A peak of 72.7 ppb, a 21-day average of 58.9 ppb, and a 60-day average of 45.7 ppb were calculated for potatoes with an application rate of 4.0 pounds AI per acre. An application rate of 1.0 pounds AI per acre was use for blueberries, predicting a peak of 37.7 ppb, a 21-day average of 32.8 ppb, and a 6-day average of 22.4 ppb. Estimates were also calculated for peaches, apples, and cucumbers, with the highest EECs resulting from application to cucumbers at a rate of 4.0 pounds AI per acre.

Diazinon also affects the olfaction of juvenile salmon, which mediates a suite of fish behaviors involved in feeding, predator avoidance, kin recognition, spawning, homing and migration. For example, (Moore and Waring 1996)studied the effects of diazinon exposure on olfaction in Atlantic salmon parr. They first exposed male parr to diazinon concentrations (0, 0.1, 1.0, 2.0, 5.0, 10 and 20 µg/L) for 30 minutes and determined the parrs' ability to detect priming odorant released by female salmon that synchronizes spawning and also has a role as a primer on male plasma steroids and gonadotropin production. At 1.0 µg/L, diazinon significantly reduced the capacity for parr to detect the priming odorant by 22 percent (compared with controls); at 20 µg/L, diazinon inhibited olfaction by 79 percent. Olfaction was affected for up to 4-5 hours following exposure.

Moore and Waring (1996) also studied the effect of longer-term exposure to diazinon on male parrs' plasma reproductive steroid levels after the males were exposed to the urine of ovulating females. Diazinon concentrations of 0.3 – 45 µg/L abolished the induction of male hormones, although levels of testosterone and one ketotestosterone were not significantly affected by the diazinon exposure. Milt production was reduced by about 28 percent at concentrations of diazinon ranging from 0.3 - 45 µg/L. We would expect these outcomes to impair Atlantic salmon's ability to detect and respond to reproductive scents and increase their probability of missing spawning opportunities, which would reduce the lifetime reproductive success of individuals that experience this response.

Scholz et al. (2000) also studied the effects of 24 hour exposures to diazinon on the swimming and feeding behavior of juvenile coho salmon. They reported statistically significant effects on swimming and feeding behaviors in the presence of an alarm cue following exposures at concentrations of diazinon at 1 and 10 µg/L (compared to control fish) and reduced homing at 0.1 µg/L.

EPA's BE also evaluated temephos. Two models were used to calculate temephos concentration, one for tidal waters and one for non-tidal waters. Liquid temephos is applied by air directly to tidal marshes to control heavy infestations of mosquito larvae. For the tidal water scenario, EPA assumed complete mixing of temephos and that 100% of the application reaches the water. Calculations are included for application rates of 0.5, 1.0 and 1.5 ounces of product (43% active ingredient). Using the highest application rate (1.5 oz/A) and shallowest water depth (1 cm), modeling results in a peak concentration of 453 µg/L temephos for tidal waters. A concentration of approximately 1.0 µg/L is needed for 100% mortality of *Aedes* mosquito larvae. For non-tidal waters, temephos is typically applied in one or two treatments per year depending upon need (numbers of breeding mosquitoes). Application rates for the 5%, 2%, and 1% granular products vary from 0.05 – 0.5 pounds of active ingredient per acre (the higher rate is for highly polluted waters). Therefore, using the highest application rate (0.5 lbs AI/A) and double application rate with the shortest interval results in a peak concentration of 25.2 µg/L; a 21-Day average concentration of 2.8 µg/L; and a 90-Day average of 1.0 µg/L.

Temephos shows a wide range of toxicity to aquatic organisms, depending on the formulation. Generally, the technical grade compound is considered moderately toxic while the emulsifiable concentrate and wettable powder formulations are highly to very highly toxic. The most sensitive species of fish is the rainbow trout with a temephos LD_{50} ranging from 0.16 mg/L to 3.49 mg/L (Johnson and Finley 1980). Other 96-hour LD_{50} values are reported as: coho salmon 0.35 mg/L, largemouth bass 1.44 mg/L, channel catfish 3.23 mg/L to >10 mg/L, bluegill sunfish 1.14 mg/L to 21.8 mg/L, and Atlantic salmon 6.7 mg/L to 21 mg/L (Johnson and Finley 1980, Kidd et al. 1991).

Trichlorfon is also highly toxic to several species of fish and aquatic invertebrates, including species like Daphnia and stoneflies that are prey for fish. LC_{50} (96-hour) values for trichlorfon are 0.18 mg/L (48-hour) in Daphnia, 0.01 mg/L in stoneflies, 7.8 mg/L in crayfish, 1.4 mg/L in rainbow trout, 2.5 mg/L in brook trout, 0.88 mg/L in channel catfish and 0.26 mg/L in bluegill (Hudson et al. 1984, Hill and Camardese 1986).

Pyrethroids, Pyrethrins, and Synergists

The pyrethroids, pyrethrins and synergists (substances which enhance the toxicity of a pesticide) whose uses would be authorized by the PGP include permethrin, permethrin, mixed cis, trans, resmethrin, sumithrin, piperonyl butoxide and n-octyl bicycloheptene dicarboximide. The latter substances, piperonyl butoxide and n-octyl bicycloheptene dicarboximide (mgk-264) are synergists that enhancing pesticide toxicity by inhibiting an organism's ability to detoxify the pyrethroid. As we described previously, formulations of these pesticides are used to control adult mosquitoes.

Paul et al. (2005) compared the toxicity of permethrin plus a synergist and technical formulations of permethrin, sumithrin and resmethrin to brook trout (*Salvelinus fontinalis*) and brown trout

(*Salmo trutta*). They reported that the toxicity of the synergized permethrin formulation was significantly increased in 24, 48 and 96-hour tests, compared to tests with the technical formulation. There was little difference in the toxicity of synergized and technical formulations of sumithrin until 48 hours had elapsed. They reported that many test fish were strongly intoxicated by either formulation of permethrin or sumithrin, but the synergized formulations of both chemicals affected fish at lower concentrations. Intoxication was potentially severe enough to reduce the survival of these fish in the wild. Finally, they tested the ability of exposed fish to swim against a current and concluded that fish exposed for 6 hours to synergized permethrin and resmethrin had far less swimming stamina than those exposed to technical formulations. They did not find a difference in the effect on swimming between the synergized and technical formulation of sumithrin. They concluded that the synergized formulations of these pesticides appeared to cause a faster response than the technical formulations and this response increased the lethal and sublethal effect of the insecticides on the trout.

Inert Ingredients

Some of the other ingredients of formulations of these pesticides are also toxic. For example, piperonyl butoxide is a common constituent of insecticide containing formulations (for example, it is a common synergist in formulations of synthetic pyrethroids) and is toxic to aquatic invertebrates and fish. The EPA (2006) reported an LC_{50} for rainbow trout of 1.9 mg/L. In longer term exposures piperonyl butoxide affects fish and aquatic invertebrates at concentrations as low as 0.11 mg/L. Piperonyl butoxide is highly toxic to aquatic invertebrates with a reported EC_{50} of 0.51 mg/L for *Daphnia magna* (USEPA 2006).

As another example, methoxychlor is a co-constituent in formulations with malathion. Formulated products are more toxic than methoxychlor alone. It is also an organo-chlorine insecticide that is toxic to fish and aquatic invertebrates. Johnson and Finley (1980) reported LC_{50}s less than 20 µg/L and one 96-hour LC_{50} of 1.7 µg/L was reported for Atlantic salmon (Howard 1991).

Representative Pesticides Evaluated in the Biological Evaluation

The pesticides evaluated in EPA's BE were selected based on the anticipated risk to ESA-listed species, expected use by Operators not required to submit annual reports, and the frequency at which they were identified in annual reports as agents applied under the 2011 PGP (Table 9). We summarize EPA's analysis in this opinion to describe risk of the discharges to be authorized under the 2016 PGP, as it was identified by EPA. Annual reports included some, but not all of these pesticides. For a number of annual report-pesticides the BE did not assess the use patterns identified in the annual report. In most cases, this is attributable to the identification of more than one use pattern under the annual report.

Table 9. Representative pesticides evaluated by EPA for the PGP (pestides identified in annual reports are in boldface).

MOSQUITOCIDES (Adulticides)	WEED AND ALGAE PEST CONTROL
Naled	Endothall
Permethrin	**2,4-D**
Resmethrin	**Copper (i.e., sulfate and chelate)**
Malathion	Diquat
Sumithrin	**Glyphosate**
Chlorpyrifos	Fluridone
MOSQUITOCIDES (Larvacides)	**Triclopyr**

Bacillus thuringiensis israelensis **Methoprene** **Temephos** **Bacillus sphaericus**	**Imazapyr** **Acrolein**
ANIMAL PEST CONTROL	**FOREST CANOPY PEST CONTROL**
Rotenone (Fish) Antimycin A (Fish) Sodium chlorate (Mollusk) TFM (3-trifluoromethyl-4-nitrophenol) (Lamprey) **Diazinon**	Malathion **Carbaryl** Diflubenzuron **Bacillus thuringiensis kurstaki** Disparlure Chlorothalonil

Pesticides Identified in Annual Reports

Pesticides are grouped among classes based on source (e.g., botanical, *Bacillus*) or chemical properties (e.g., azoles, neonicitinoids). For example, pesticides identified in the annual reports for the 2011 PGP are classified in Table 11. These pesticides do not represent all classes and active ingredients that were applied under the PGP because annual reports are not required of for-hire applicators or Operators who are small entities and do not discharge to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur. Table therefore represents data from all large entities, whether or not they discharge to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur, and from those small entities discharging to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur.

Table 10. Pesticides identified in annual reports. Those reported to be used in areas where ESA-listed species or designated critical habitat under NMFS' jurisdiction occur are in boldface.

Pesticide Class	Mosquito	Aquatic Weed	Animal Pest	Forest Canopy
Aldehyde		**Acrolein**		
Amide		Napropamide		
Anthranilic diamide			Chlorantraniliprole	
Azole	Fenbuconazole	Prothioconazole	Fenbuconazole	
Benzoic acid		**Dicamba**, Mesotrione		Dicamba
Benzoylcyclohexanedione		Diquat dibromide		
Botanical	Abscisic acid	Cytokinin (as kinetin)	Cube Resins other than rotenone **Rotenone**	**Verbenone**
Chlorophenoxy acid or ester		**2,4-D**, MCPA		**2,4-D**
Chloropyridinyl		**Triclopyr**		Triclopyr
Coumarin			Brodifacoum	
Cyclohexenone derivative		Clethodim, Sethoxydim		
Diacylhydrazine			Methoxyfenozide Tebufenozide	
Dithiocarbamate-ETU		Mancozeb, Metam-sodium	Ferbam, Mancozeb	
Imidazolinone		Imazamox, **Imazapic** **Imazapyr**		Imazapic
Inorganic		potassium salts of phosphorous acid Copper ethanolamine Copper ethylene-diamine Copper hydroxide Copper sulfate pentahydrate Copper triethanolamine	Phosphorous acid Copper hydroxide Manganese	
Juvenile hormone mimic	**S-Methoprene**			
Bacillus	**B. sphaericus** **B. thuringiensis** **B. thuringiensis subspecies israelensis**	B. thuringiensis subspecies israelensis	B. thuringiensis subspecies Kurstaki	B. thuringiensis subspecies israelensis
Neonicotinoid	Imidacloprid		Acetamiprid Clothianidin Imidacloprid Thiamethoxam	
N-Methyl Carbamate	Carbaryl		Carbaryl	
N-phenylphthalimide		**Flumioxazin**		
Organophosphonate	Diazinon		Acephate	Fosamine
Organophosphorus	**Naled**, Temephos		Chlorpyrifos, Diazinon	
Oxadiazine	Indoxacarb		Indoxacarb	

Pesticide Class	Mosquito	Aquatic Weed	Animal Pest	Forest Canopy
Petroleum derivative	Aliphatic petroleum solvent **Mineral Oil**	**o-Xylene**		
Pheromone				**3-Methyl-2-cyclohexen-1-one**
Amino Acid Derivative		**Glyphosate**		Glyphosate
Polyalkyloxy Compound	POE isooctadecanol			
Pyrethroid	Bifenthrin **Permethrin w/ Piperonyl butoxide**		Permethrin Pyrethrins	
Pyrethroid Ether	Ethofenprox			
Pyridazinone		Norflurazon		
Pyridinecarboxylic acid		**Aminopyralid, Clopyralid, Fluroxypyr Picloram-potassium**		Clopyralid Picloram-potassium
Quinazoline			Fenazaquin	
Spinosyn	Spinetoram **Spinosad**		Spinetoram	
Strobin	Azoxystrobin		Azoxystrobin	
Substituted Benzene		Chlorothalonil Dichlobenil	Chlorothalonil	
Sulfonylurea		**Chlorsulfuron** **Metsulfuron** **Sulfometuron**		Chlorsulfuron Metsulfuron
Triazine		Indaziflam		
Unclassified		Dazomet Endothall Fluridone Fosetyl-Al Quinclorac		
Uracil		Bromacil		
Urea		**Diuron**		
Xylylalanine			Metalaxyl-M	

Risk Hypotheses for Evaluating Pesticide Discharges under the PGP

Figure 6.1.1-1 in EPA's BE for the 2016 PGP illustrates the pathways by which pesticide discharges (stressor sources) under the different use patterns may cause direct and indirect effects to ESA-listed species (Figure 6). Pesticides act directly to reduce survival and fitness of ESA-listed individuals and indirectly through reducing the survival and fitness of species upon which ESA-listed species rely for forage, shelter, and the maintenance of habitat quality (e.g., riparian vegetation shades water, influencing temperature).

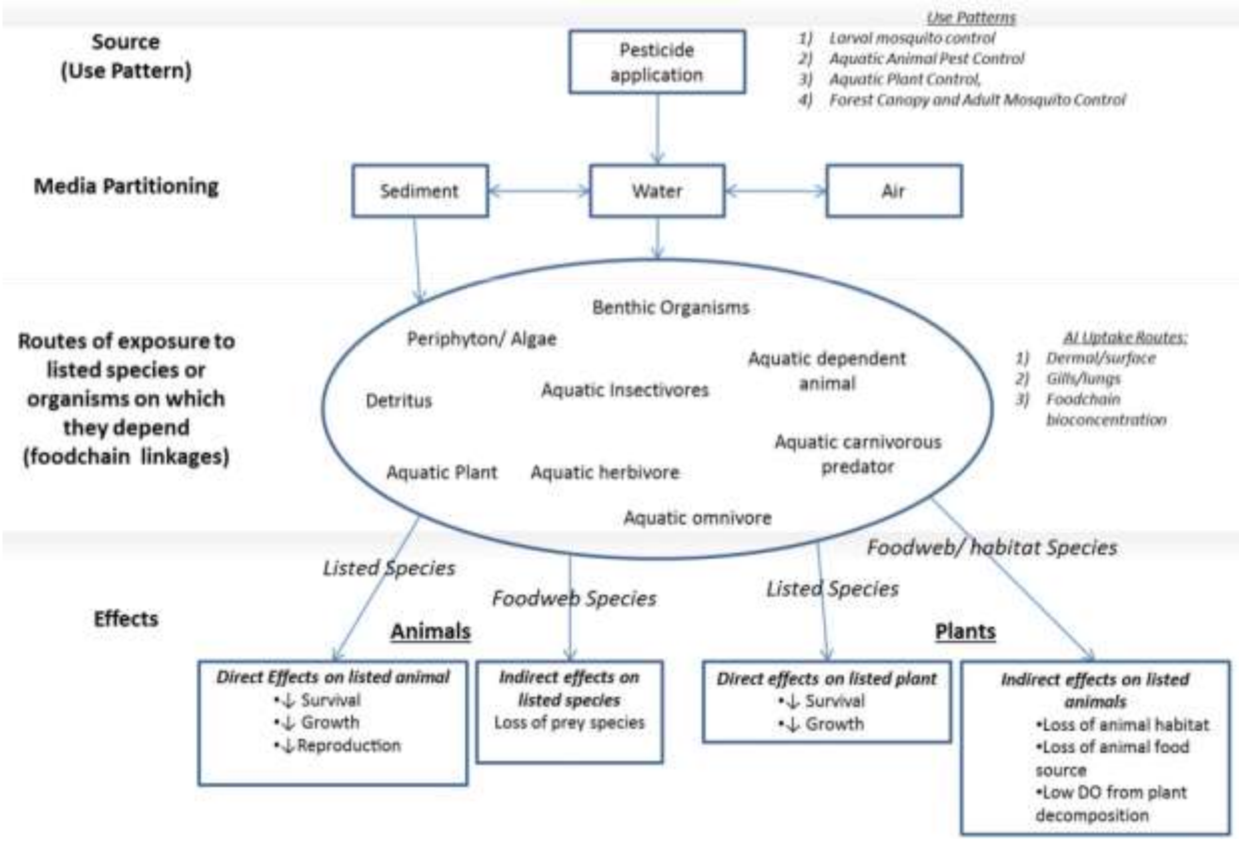

Figure 6. Generalized pathways for pesticides effects to ESA-listed species and designated critical habitat under NMFS' jurisdiction (USEPA 2016a).

The objective of the risk assessment portion of this programmatic opinion is to determine whether pesticides discharge under the use patterns eligible for coverage under the PGP, in the absence of controls and requirements under the PGP, would directly or indirectly adversely affect individual survival or fitness such that the extinction risk of ESA-listed populations or species would be increased or that designated critical habitat necessary for the persistence of ESA-listed species would be destroyed or adversely modified. Generally speaking, the values to be protected are the survival and fitness of individuals and the value of designated critical habitat for conservation of an ESA-listed species. Risk hypotheses are constructed by placing information on the stressors of the action, pesticides, in context of species and essential features of designated critical habitat potentially affected bythese discharges. Pesticide products, including the active ingredients, inert ingredients such as adjuvants and surfactants and metabolites and degradates affect organisms through various toxic mechanisms potentially resulting in effects such as direct lethality, disrupted growth and maturation, reduced offspring survival, or reduced reproductive capacity. Given the scope of the PGP, it is not possible to evaluate all exposures and potential consequences of the authorized discharges.

Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in exposures to toxicants that will affect the survival and fitness of individuals through:

- direct mortality
- reduced growth

    - altered behavior
    - reduced fecundity (i.e., reduced reproductive output or offspring survival)

- Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in exposures to toxicants that will affect the survival and fitness of individuals through:
    - reduction in extent of inhabitable area/avoidance
    - reduction in prey species

Effects to designated critical habitat analysis includes direct and indirect effects on biological elements within the spatial extent of designated critical habitat (e.g., prey, plant cover) affecting the value of the habitat for the conservation of the species. Since the stressors of the action are toxicants, it is the biological features specified in designated critical habitat that may be affected by the action. . The overarching risk hypothesis for evaluating effects to designated critical habitat is:

- Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in adverse effects to designated critical habitat features that are essential to the conservation of the species

8.1.2 Exposure and Response Analysis

The exposure and response analysis evaluates whether individuals of ESA-listed species may be exposed and respond adversely to the stressors of the action, as proposed by the risk hypotheses arrived at in the problem formulation.

Exposures to Pesticide Active Ingredients and Formulations

The *Action Area* where pesticide discharges occur includes large areas over which EPA has permitting authority (see Figure 1, Figure 2, and Figure 3, plus Pacific Islands and Territories). The composition of pesticide products discharge and the timing, frequency, intensity, duration and location, of exposures resulting from individual discharges and aggregate exposures resulting from repeated discharges in one location, or within the home range or migration route of individuals, or within or near essential features of designated critical habitat are unknown. The number of individuals of each species and life stage occurring in affected waters at the time of such discharges are also unknown, especially considering that the numbers of individuals vary with the season, environmental conditions, and changes in population size due to recruitment and mortality over the course of a year. For these reasons, all species and life stages identified in section 6.2, *Species and Designated Critical Habitat in the PGP Action Area* are expected to be exposed to the stressors of the action.

EPA's BE assessment addressed this uncertainty by evaluating representative locations and exposure scenarios. The BE used modeled peak or chronic EECs based on environmental fate characteristics and pesticide use data compiled by EPA-OPP. To estimate EECs for an active ingredient (AI) and use pattern, EPA modeled scenarios intended to represent sites in areas that are highly vulnerable to either runoff, erosion, or spray drift. For ecological risk assessment, EPA relies on a standard water body to receive the edge-of-field runoff estimates. The standard water body is of fixed geometry and includes the processes of degradation and sorption expected to occur in ponds, canals, and low-order streams (e.g., first and second order streams). The water body is assumed to be static (no outflow) as a conservative measure. For pesticides applied directly to water to control aquatic pests (3 of the 4 use patterns covered in the PGP), EPA calculated EECs based on the allowable rates specified by the AI label as well as fate characteristics of the AI. These calculations also assumed static conditions (i.e., little or no dilution or transport) as a conservative measure. In terms of defining exposure of ESA-listed species to AIs. That is to say, AI concentration remaining at a site after its intended use is achieved is assumed to be equal to the modeled AI concentration given its maximum application rate for a given use pattern and its fate properties in air, water, sediment, and soil. EPA stated that this assumption is likely to result in a conservative (i.e., high) estimate of the AI EEC in many cases.

NMFS notes that while EECs provide information, they do not integrate repeated exposures that may be necessary to control a pest species in a given area (e.g., mosquito control), exposure to multiple AIs and adjuvants in product formulations, or multiple exposures that may occur over the spatial extent of individual's home range or migration route. Further, the use of EECs does not address exposures in the shallow backwater pools that are important to salmonid rearing. NMFS also notes that exposures of rockfish, coral and Nassau grouper were not included in these analyses. EPA did not provide modeling data for exposures in marine waters, so any assessment for these species would have to be based on EPA's exposure estimates for other waters, where available. Most monitoring data reporting the detection of pesticides or degradates in environmental media are not realistic indicators of exposure because, unless the data are the result of a structured targeted monitoring program, detected pesticide levels are the result of an unknown prior application or an unknown product formulation, under an unknown use pattern. Ideally targeted monitoring is conducted before pesticide application, at the time of discharge, and after application has ceased to capture information on exposure intensities from background conditions to peak EEC at the time of discharge to the end of exposure period of interest. Further, as "snapshots in time," monitoring data do not capture the peak exposure concentration of a single or multiple AIs, surfactants, and adjuvants in the product formulation(s) used and likely miss exposures to less persistent chemicals.

Responses Considered in EPA's BE

Research conducted over several decades has established that many, but not all, pesticides pose serious risks to survival, development, growth, or reproductive success of aquatic organisms as a direct result of the exposure or because of the chemical's effect on their behavioral patterns. The effects of pesticides on salmonids are well-researched. Meanwhile, information linking exposures to current-use pesticide with such effects has only been collected over the past ten years for corals (Jones and Kerswell 2003, Jones et al. 2003, Raberg et al. 2003, Jones 2004, Negri et al. 2005, Watanabe et al. 2006, Cantin et al. 2007, Markey et al. 2007, Watanabe et al. 2007, Negri et al. 2009, Sheikh et al. 2009, Negri et al. 2011, van Dam et al. 2012a, van Dam et al. 2012b, Bladow et al. 2015, Ross et al. 2015, van Dam et al. 2015) and sturgeon (Cope et al. 2011, Filizadeh and Islami 2011, Frew and Grue 2015). No studies were found for such effects specifically in rockfish or grouper[17]. The available information on pesticides effects on cetaceans and other marine mammals report tissue concentrations and blood chemistry factors which are difficult to link to adverse effects. Further, much of the existing data on pesticides effects on sea turtles evaluates persistent organic chlorines (e.g., DDT, chlordane, dieldrin) which are no longer

[17] ISI Web of Science search 08/26/2016 **TOPIC:** (rockfish or bocaccio or grouper) *AND* **TOPIC:** (pesticide or herbicide or insecticide or fungicide or piscicide)

registered for use in the U.S. and their use is therefore not eligible for coverage under the PGP[18]. The absence of published data for the effects of pesticides on specific species groups does not indicate that such effects do not exist. It is particularly difficult to conduct laboratory research of any kind on very large or long-lived species due to legal restrictions (i.e., the ESA and the Marine Mammal Protection Act) and logistical considerations (e.g., lab space, species water and feed requirements, etc). Given the scope and uncertainty in the exposures, EPA evaluated risk for representative pesticides exposures of standard laboratory species representing (i.e., surrogates for) ESA-listed species present in Massachusetts and Idaho (Table 11).

Table 11. Summary of types of surrogate species EPA used to assess direct effects of active ingredients on ESA-listed species in Idaho and Massachusetts in the biological evaluation effects analyses.

Common Name	Scientific Name	Surrogate organism type
Chinook Salmon	*Oncorhynchus tshawytscha*	Freshwater fish
Sockeye Salmon	*Oncorhynchus nerka*	
Steelhead	*Oncorhynchus mykiss*	
Shortnose Sturgeon	*Acipenser brevirostrum*	
Atlantic Sturgeon	*Acipenser oxyrinchus*	
Green Sea Turtle	*Chelonia mydas*	Saltwater fish
Hawksbill Sea Turtle	*Eretmochelys imbricata*	
Kemp's ridley Sea Turtle	*Lepidochelys kempii*	
Leatherback Sea Turtle	*Dermochelys coriacea*	
Loggerhead Sea Turtle	*Caretta caretta*	

The use of surrogate species cultured for use in standard laboratory tests provides certainty on the expected responses of control organisms. The species used include, but are not limited to, fathead minnow, rainbow trout, flagfish, bluegill, Atlantic silverside, and sheepshead minnow. To assess direct and indirect effects, EPA selected the most sensitive (i.e., lowest) acute and chronic[19] endpoints from available data for each species group (Table 12) and compared those with the EEC to obtain risk quotients (RQ). NMFS has modified this table to include effects on coral photosynthetic symbionts (zooxanthellae).

The RQ was then evaluated against the level of concern (LOC, Table 13). EPA uses LOCs to interpret the risk quotient and to analyze potential risk to non-target organisms and the need to consider regulatory action. When an RQ exceeds the LOC for a particular category, for example, the LOC of 0.05 for ESA-listed threatened and endangered species, EPA presumes a risk of concern to that category. In general, the higher the RQ, the greater the potential risk. If the RQ for a given assessment endpoint was greater than the LOC, EPA would report the exposure as “Likely to Adversely Affect” (LAA) for those ESA-listed species represented by that assessment endpoint. EPA states that this was likely a conservative interpretation in many cases because there may be no actual potential exposure of a given ESA-listed species to an AI use pattern.

[18] ISI Web of Science search terms 08/26/2016 **TOPIC:**(loggerhead or ridley or leatherback or hawksbill) *AND* **TOPIC:** (pesticide or herbicide or insecticide or fungicide or piscicide)

[19] For EPA analyses, acute endpoints reflect mortality of half of exposed organisms after a 96 hour exposure. The chronic endpoints are those that can be directly associated with organism-level apical endpoints such as breeding success and development. PBL NOTE: Undefined term.

NMFS notes that this conservatism also reflects a large degree of uncertainty associated with the subsequent assessments in the BE.

Table 12. Summary of assessment endpoints for use in the risk quotient methodology of assessing risk (USEPA 2004).

Assessment Endpoint	Species Type	Endpoint Type
Acute Direct Toxicity and Indirect Effects (forage species/prey)	Freshwater fish Freshwater Invertebrates Estuarine/Marine Fish Estuarine/Marine Invertebrates	LC_{50}
Acute Direct Toxicity (coral symbionts)	Aquatic plant	EC50 or NOEC
Chronic Direct Toxicity and Indirect Effects (forage species/prey)	Freshwater Fish Freshwater Invertebrates Estuarine/Marine Fish Estuarine/Marine Invertebrates	NOEC

Table 13. Summary of Levels of Concern used in Assessing Estimated Environmental Concentrations.

Risk Presumption	Risk Quotient	Level of Concern	Response threshold used to evaluate EEC
Acute High Risk	EEC/LC50 or EC50	0.5	Acute threshold: Lowest tested EC50 or LC50 for freshwater fish and invertebrates and estuarine/marine fish and invertebrates acute toxicity tests
Acute Restricted Use	EEC/LC50 or EC50	0.1	
Acute ESA-listed Species	EEC/LC50 or EC50	0.05	
Chronic Risk	EEC/NOEC	1.0	Chronic Threshold: Lowest NOEC for freshwater fish and invertebrates and estuarine/marine fish and invertebrates Early life-stage or full life-cycle tests
Plants: Acute Listed Endangered Species (e.g., zooxanthellae)	EEC/EC50 or NOEC	1.0	Lowest EC05 or NOEC for both seedling emergence and vegetative vigor for both monocots and dicots

The outcome of these analyses, summarized in Table 14, indicate frequent, and often large magnitude of RQ exceedences over the LOC. These RQs are based on peak EEC and not actual post application under the four use patterns eligible for coverage under the PGP. However, they also reflect single exposure events to a single A.I., not actual pesticide products as they are applied.

Table 14. Results of risk analyses from EPA's biological evaluation analysis of pesticide uses authorized under the PGP.

Risk Scenario: Specie group and pesticide use pattern	Number of pesticides examined in the BE	Maximum ratio of acute threshold to peak estimated exposure	Percentage of scenarios with elevated acute risk based on LOCs	Maximum ratio of chronic threshold to estimated chronic exposure concentration	Proportion of scenarios with elevated chronic risk
Estuarine Fish (surrogate for sea turtles – see Table 11)[a]					
aquatic animal pest	5	474	67%	636	100%
aquatic weed	8	78	50%	60	100%
forest pest	5	7	67%	355	50%
Mosquito	8	26	83%	66	75%
Estuarine Invertebrates (Indirect effects – forage species)[a]					
aquatic animal pest	5	102	33%	636	100%
aquatic weed	8	417	86%	256	100%
forest pest	5	143	80%	3,186,667	100%
Mosquito	8	529	100%	4,022	100%
Freshwater Fish					
aquatic animal pest	5	308,556	80%	647	100%
aquatic weed	8	119	88%	43	43%
forest pest	5	2	60%	9	25%
Mosquito	8	16	75%	32	71%
Freshwater Invertebrates (Indirect effects – forage species)					
aquatic animal pest	5	390,625	80%	1,113	100%
aquatic weed	8	275	88%	1,634	57%
forest pest	5	80	100%	918,000	100%
Mosquito	7	2,291	100%	2,000	60%
Aquatic nonvascular plants[a]					
aquatic animal pest	1	0.116	100%	-	none
aquatic weed	4	316	67%	-	none
Mosquito	1	0.264	100%	-	none

[a] EPA's BE did not assess pesticide effects for Nassau grouper, rockfish, or coral species. In this opinion, the estuarine fish data are considered surrogate data for Nassau grouper and rockfish, estuarine invertebrate data are considered surrogate data for coral species, and aquatic nonvascular plants data are considered surrogate data for the zooxanthellae of coral species.

Responses to Degradates

Pesticides are transformed into other compounds over time by chemical, photochemical and biologically-mediated reactions; these other compounds are generally called "degradates" or "metabolites" (Boxall et al., 2004; Gilliom et al., 2006). Degradates, like their parent compounds, have the potential to adversely affect water quality, depending on their toxicity. Sinclair and Boxall (2003) reported that 41 percent of degradates were less toxic than their parent compounds, 39 percent had toxicities similar to their parents, 20 percent were more than 3 times more toxic than their parent compound and 9 percent were more than 10 times more toxic.

For example, the major metabolite of carbaryl is 1-naphthol, which is formed by abiotic and microbially mediated processes and has been reported to represent up to 67 percent of the applied carbaryl in degradation studies. This degradate is more toxic than carbaryl itself. Shea and Berry (1983) compared 10-day acute lethalities between carbaryl and 1-naphthol in goldfish (*Carassius auratus*) and killifish (*Fundulus heteroclitus*). They concluded that 1-naphtol was about five times more toxic than carbaryl in goldfish and twice as toxic as carbaryl in killifish. In addition, fish exposed to 1-naphthol showed neurological trauma including erratic swimming behaviors and increased opercula beats following 4-hour exposures at 5 mg/L and 24-hour exposures at 10 mg/L. They did not observe any of these symptoms in the carbaryl treatments.

Responses to Mixtures

Most aquatic species are likely to be exposed to mixtures of pesticides, their degradates and other chemicals that exist in the environment. Once in a mixture, co-occurring pesticides (including their degradates) can either 1) act independently of one another (called an "independent" effect); 2)have additive effects (for example, this might be expected for pesticides with a common mode of action and similar chemical structure); 3) have synergistic effects in which their combined toxicity is greater than their additive toxicity; or 4) have combined toxicity that is less than their additive toxicity (called an "antagonistic" effect).

As an example of synergistic effects, (Relyea and Mills, 2001; 2004) exposed amphibians to a combination of pesticides and chemical cues mimicking natural predators and found that these combinations induced stress and, as a result, increased the mortality rates of the amphibians (see also Sih et al., 2004). For some species, exposing amphibians to combinations of pesticides and natural stressors produced mortality rates that were substantially greater than mortality rates associated with each individual stressor. For example, carbaryl was up to 46 times more lethal to gray treefrog tadpoles (*Hyla versicolor*) when they were exposed to a combination of this pesticide and chemical cues emitted by aquatic predators (Relyea and Mills, 2001). When they were exposed to malathion at concentrations of 5 mg/L, 42 percent of the gray treefrog tadpoles died when predator cues were absent, but 82 percent died when predator cues were present (Rhatigan, 2004).

Mixtures containing malathion resulted in additive effects (when mixed with DDT, toxaphene), synergistic effects (when mixed with Baytex, parathion, carbaryl, perthane) and antagonistic effects (when mixed with copper sulfate) (Macek, 1975). Mixtures of diazinon and parathion killed more bluegill sunfish than predicted. Tierney et al. (2008) exposed juvenile steelhead to environmentally realistic concentrations of a mixture that included chlorpyrifos, diazinon and malathion (the realistic mixture contained chlorpyrifos at 13.4 ng/L; diazinon at 157 ng/L; and malathion at 46.3 ng/L, respectively). Exposures to this mixture for 96 hours compromised the ability of juvenile steelhead to detect changes in odorant concentrations, which would impair behaviors that rely on smell such as homing and migration.

Mixtures that paired two organophosphates produced a greater degree of synergism than mixtures containing one or two carbamates, particularly mixtures containing malathion coupled with either diazinon or chlorpyrifos (Laetz et al. 2009). At the highest exposure treatment, 1.0 EC_{50} (malathion at 37.3 µg/L, chlorpyrifos at 2 µg/L, diazinon at 72.5 µg/L), binary combinations produced synergistic toxicity. Coho salmon exposed to combinations of diazinon and malathion as well as chlorpyrifos and malathion all died (Laetz et al. 2009). Fish exposed to these organophosphate mixtures showed toxic signs of inhibition of AChE, including loss of equilibrium, rapid gilling, altered startle response and increased mucus production before dying.

Organophosphate combinations were also synergistic at the lowest concentrations tested. Diazinon and chlorpyrifos were synergistic when combined at 7.3 μg/L and 0.1 μg/L, respectively. The pairing of diazinon (7.3 μg/L) with malathion (3.7 μg/L) produced severe (> 90 percent) AChE inhibition including classical signs of poisoning as well as death with some combinations. For binary combinations of malathion, diazinon and chlorpyrifos synergism was likely to occur at exposure concentrations that were below the lowest used in this work (i.e., chlorpyrifos concentrations lower than 0.1 µg/L; diazinon concentrations lower than 7.3 µg/L; malathion concentrations lower than 3.7 µg/L.

Responses Not Considered in EPA's Biological Evaluation
Response as a result of impacts to prey base of Southern Resident Killer Whale

We evaluated the potential effects of EPA's issuance of their PGP on designated critical habitat by first reviewing the essential features or primary constituent elements of designated critical habitat for listed designations. Based on our analysis, the primary features that may be affected by pesticide discharges under the four use patterns eligible for coverage under the PGP are those designated as "prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth."Salmon are a significant contributor to the overall ecological food web throughout their range. Two significant indirect effects of the proposed action to Chinook, coho, sockeye and chum salmon and steelhead could result in the further loss of prey species for southern resident killer whales. Such reductions would also likely result in the loss of nutrient transport to freshwater systems that are important to Pacific salmonids themselves (Ford et al. 2010). Bilby et al. (1996) demonstrate that juvenile and older age classes of salmon grow more rapidly with the appearance of spawners because these younger fish will feed on eggs and spawner carcasses. Salmon carcasses in rivers and streambanks are a significant source of food to a wide number of animals and affect the overall productivity of nutrient-poor systems (Bilby et al. 1996, Cederholm et al. 2000). Bilby et al. (1996) showed that up to 45 percent of the carbon in cutthroat trout and 40 percent of the carbon in young coho comes from the decaying carcasses of the previous generation of salmon. Increased body size is directly correlated to increases in over winter survival and marine survival. They suggest that reduced nutrient transport is one important indicator of ecosystem failure and is contributing to the observed reductions in abundance we have seen in many salmon populations, which could further diminish the success of recovery efforts. Given many salmon populations comprise the prey component of killer whale designated critical habitat, any additional reduction in prey attributable to the PGP could adversely modify their designated critical habitat.

Based on killer whale stomach contents from stranded whales and field observations of predation, Ford et al. (1998) determined that 95 percent of the diet of resident killer whales consists of fish, with roughly 66 percent being Chinook salmon. The authors suggested that killer whales might preferentially hunt Chinook salmon because these fish have large body sizes and a high fat content. A reduction in Pacific salmon – Chinook salmon in particular– from effects from the proposed action is likely to have adverse effects on the fitness of southern resident killer whales and their population viability. As noted earlier, a 50 percent reduction in killer whale calving has been correlated with years of low Chinook salmon abundance (Ward et al. 2009).

A reduction in the number of adult Chinook salmon in the Puget Sound would reduce the forage base for southern resident killer whales. Southern resident killer whales are not restricted to Puget Sound, but do spend a large portion of time in Puget Sound, the Strait of Juan de Fuca and

Haro Strait. Prey losses could also be realized throughout their range, including Oregon and California. Such reductions in prey could impede recovery.

Response of Coral Species

The EPA's BE did not asses exposure or response of coral species and Nassau grouper because discharges to marine environments were not expected to result in exposures to these species. The PGP does not cover drift, but in this opinion drift of pesticides resulting from the four use patterns eligible for coverage under the PGP discharges is considered an indirect effect of the action. Land based pesticides do reach and accumulate in reef habitats and enter the food web (Whitall et al. 2015, Salvat et al. 2016). Research indicates that land-sourced herbicides have implications on coral health through effects on the photosynthesizing symbionts, particularly in combination with elevated water temperatures associated with climate change (Negri et al. 2011, van Dam et al. 2012a).

For marine coral and fish species in Puerto Rico and the Pacific Islands, the mosquito control use pattern is a potential source of pesticide exposures that can directly attributable to a use pattern eligible for coverage under the PGP. Early morning aerial applications use ultra low-volume atomizers to maximize contact time with flying mosquitoes. Naled, an organophosphate for the control of mosquitos has a short residual half-life in water (< 1 day), degrading to dichlorvos, both classified as very highly toxic to aquatic invertebrates. Dichlorvos half-life ranges from hours to days in the presence of reduced sulfur species indicative of near coastal marine environments (Gan et al. 2006). Dichlorvos has been reported to persist in seawater for as long as 180 days (Lartiges and Garrigues 1995).

A study by Pierce et al. (2005) investigated the potential for off-shore transport of toxic concentrations of naled and permethrin resulting from routine mosquito control operations. The study confirmed tidal transport of naled and its degradation product, dichlorvos, to the Florida Keys National Marine Sanctuary 14 hours after application at concentrations of 0.1 µg/L for naled and 0.6 µg/L dichlorvos 1 km away from the application site. Permethrin was detected adjacent to application routes at concentrations ranging from 5.1 to 9.4 µg/L 2-4 hours after application.

Since this was a targeted monitoring study, and the only study available of its kind, (i.e., the source and application rate and timing of the pesticide was known) NMFS will use these concentrations as EECs for the pesticides in evaluating toxicity data to assess the hazards posed by discharges eligible for coverage under the PGP for reef-dwelling ESA-listed coral species.

Data provided by EPA in the BE for this consultation indicate an LC50 as low as 0.92 µg/L dichlorvos (95 percent confidence interval of 0.7-1.1 µg/L) for nauplii of the marine copepod species *Tigriopus brevicornis* (Forget et al. 1998). Using the targeted monitoring data from Pierce et al. (2005), this results in an RQ of 0.65 (i.e., 0.6 ug/L EEC/ LC50 0.92 ug/L), far exceeding the acute LOC for ESA-listed species of 0.05 (see Table 13). The data also included a record for responses of the coral species *Acropora tenuis* to dichlorvos, reporting a NOEC for dissociation of soft tissues from the skeleton of a-symbiont juveniles at 0.1 ug/L dichlorvos after 10 days. Using the targeted monitoring EEC, this provides an RQ that is 6-fold EPA's LOC. This study examined the effects of dichlorvos to coral both with and without symbiont colonization and found that the same exposure intensity resulted in significant tissue detachment in symbiont-colonized coral juveniles, with 18 percent of colony fragments affected (Watanabe et al. 2006).

The lowest invertebrate LC50 reported in the data EPA provided for naled was 460 µg/L for oysters exposed for four days (Lowe 1965, USEPA 1992). Chronic values were not found for naled in EPA's dataset. Toxicity data provided by EPA from its Ecotoxicology Knowledgebase (ECOTOX)[20] reported a 4-day LC50 for naled as low as 4.3 ug/L under static exposure (i.e., exposure media was not renewed during the study) of Korean shrimp (Schoettger 1970) providing an RQ of 0.023, which is below the acute listed-species LOC of 0.05 used by EPA. The ECOTOX record for this datum indicates gaps in descriptors for control type and test type and also because the accompanying flow-through test (i.e., exposure media continually pumped through exposure chambers) resulted in LC50 of 15.4 ug/L, suggesting that the results reported by this study were affected by the test conditions affecting response to pesticide exposure. The lowest NOEC reported in ECOTOX was for the growth of opossum shrimp at 0.2 ug/L naled after 31 days exposure (USEPA 1992). Using the targeted monitoring data, this produces an RQ that is twice EPA's LOC of 1.0 for chronic NOECs. The EPA-supplied data may have excluded this NOEC because the ECOTOX record notes that "control data were presented without accompanying methodology," meaning that the data source did not indicate the type of media used for controls (e.g., natural water, reconstituted lab water, inclusion of chemical carriers).

The implications for coral species based on these data suggest adverse effects would occur to coral species as a result of naled use for mosquito control. More recent laboratory work evaluated the implications of exposures resulting from mosquito control with naled on larva of the coral species *Porites astreoides*. Larval survivorship, settlement and post-settlement survival of coral exposed to naled, dichlorvos, and permethrin (Ross et al. 2015). Due to recent pesticide application activity near the source seawater from which the exposure solutions were prepared, the dilution water used in the study contained low background concentrations of pesticides. The controls contained 0.62 ug/L permethrin, 0.7 ug/L naled, and 0.4 ug/L dichlorvos. Larval survival 18 to 20 hours after initiating the study was 80 percent in seawater to which no additional pesticide was added and 60 percent in seawater to which an additional 0.1 ug/L naled was added. At the end of the study, total naled and dichlorvos concentrations were 0.63 and 0.53 ug/L, respectively. Based on the 2006 Watanabe et al. study, adverse effects may occur in coral species due to exposures to the degradate dichlorvos to discharges of naled under the mosquito control use pattern.

Considering the proximity of Puerto Rico coral reefs to areas that may be treated with mosquitocides, exposures to permethrin adjacent to application areas are also pertinent. Pierce et al. (2005) compared the observed permethrin concentrations to acute toxicity thresholds for mysid shrimp, but more recent and more relevant data are available for *Acropora millepora* showing 50 percent inhibition of fertilization at 1 ug/L permethrin and larval settlement reductions of 60-100 percent at this same exposure level (Markey et al. 2007).

[20] Accessed 9/1/2016, https://cfpub.epa.gov/ecotox/advanced_query.htm, Some of these data to not pass EPA's restrictive data screening protocol.

Figure 7. Aerial image example showing the proximity of coral reef crest to shore (Loiza, Puerto Rico).

Response of Nassau Grouper

The EPA-supplied data for marine fish did not include LC50s for naled. Only LC50 data for dichlorvos was provided. These LC50 concentrations were all much higher than the EEC of 0.6 from the Pierce et al. study (2005, Figure 8). EPA's screened ECOTOX LC50 data for saltwater fish species exposed to naled were also much higher than those for invertebrates and higher than the EEC from Pierce et al (2005), ranging from 130 to 2800 µg/L. Adverse effects to Nassau grouper are therefore not expected to occur as a result of discharges of naled under the mosquito control use pattern.

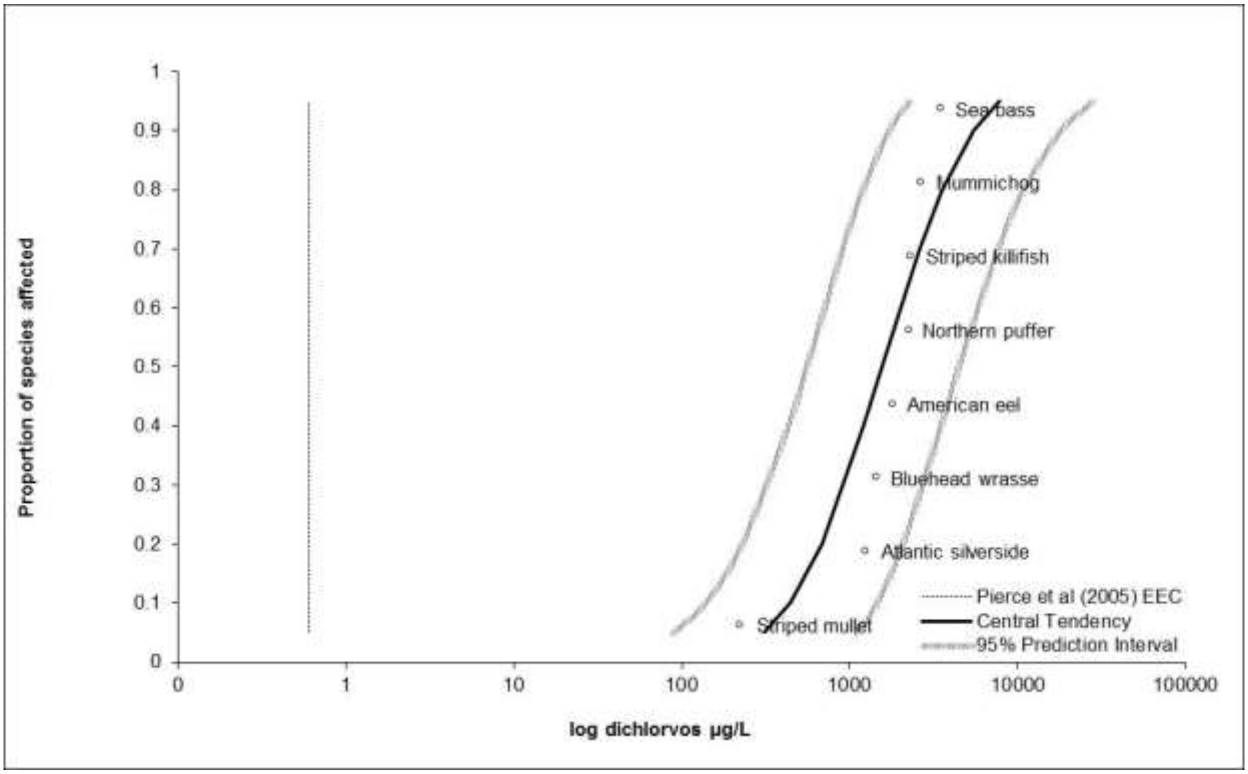

Figure 8. Distribution of LC50s for marine fish species exposed to dichlorvos for four days releative to the concnetration reported in Pierce et al. (2005).[21]

The amount of data for saltwater exposures to permethrin does not allow generation of a sensitivity distribution. The screened ECOTOX data provided by EPA include a 96 hour LC50 of 16 µg/L for sheepshead minnow (Sappington et al. 2001). The data provided by EPA is current up to 2010. More recently an LC50 of 8 µg/L was reported for juvenile red drum (Parent et al. 2011). The available 96 hour LC50 data in ECOTOX for freshwater fish bracket these values, ranging from 1.2 µg/L to greater than 10 mg/L. Considering the proximity of habitats where Nassau grouper may occur to locations where pesticides would be used to control mosquitoes (i.e., mangroves, seagrass, coral reefs), the use of pyrethroid insecticides like permethrin could pose risk to Nassau grouper.

8.1.3 Risk Characterization

The risk characterization evaluates the implications of the exposure and response results, and other available evidence for the assessment endpoints identified in the risk hypotheses to

[21] log probit fit, SSD_Generator_V1.xlt from https://www3.epa.gov/caddis/da_software_ssdmacro.html

determine whether the responses rise to population-level effects. To review, the risk hypotheses evaluated are:

Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in exposures to toxicants that will affect the survival and fitness of individuals through:

- direct mortality
- reduced growth
- altered behavior
- reduced fecundity (i.e., reduced reproductive output or offspring survival)

- Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in exposures to toxicants that will affect the survival and fitness of individuals through:
    - reduction in extent of inhabitable area/avoidance
    - reduction in prey species
- Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in adverse effects to designated critical habitat features that are essential to the conservation of the species

Analyses in NMFS Opinions

EPA has consulted with NMFS under section 7(a)(2) of the ESA on the registration of several pesticides on the West Coast[22]. The outcomes of those consultations identifying risks to ESA-listed salmonids are summarized in Table 15. In a 2008 opinion NMFS concluded that current use of chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 listed salmonid ESUs/DPSs. This opinion was remanded back to NMFS by the U.S. Court of Appeals for the 3rd Circuit and these pesticides are now being reassessed under the interagency effort to develop interim scientific approaches to assess the impact of pesticides on ESA-listed species and designated critical habitat, as required by ESA and as recommended by the April 2013 NAS report. In 2009, NMFS further determined that the current use of carbaryl and carbofuran is likely to jeopardize the continued existence of 22 ESUs/DPSs and the current use of methomyl is likely to jeopardize the continued existence of 18 ESUs/DPSs of listed salmonids. NMFS and EPA plan to revisit this analysis as well.

In 2010 NMFS issued an opinion that concluded pesticide products containing azinphos methyl, disulfoton, fenamiphos, methamidophos, or methyl parathion are not likely to jeopardize the continuing existence of any listed Pacific salmon or destroy or adversely modify designated critical habitat. NMFS also concluded that the effects of products containing bensulide, dimethoate, ethoprop, methidathion, naled, phorate, or phosmet are likely to jeopardize the continued existence of some listed Pacific salmonids and to destroy or adversely modify designated habitat of some listed salmonids. In 2011, NMFS issued an opinion on the effects of four herbicides and two fungicides. NMFS concluded that products containing 2,4-D are likely to jeopardize the existence of all listed salmonids, and adversely modify or destroy the designated critical habitat of some of these ESUs and DPSs. Products containing chlorothalonil or diuron

[22] See: http://www.nmfs.noaa.gov/pr/consultation/pesticides.htm

were also likely to adversely modify or destroy designated critical habitat, but not likely to jeopardize listed salmonids. NMFS also concluded that products containing captan, linuron, or triclopyr BEE do not jeopardize the continued existence of any ESUs/DPSs of listed Pacific salmonids or adversely modify designated critical habitat. In 2012 NMFS issued an opinion on oryzalin, pendimethalin, and trifluralin that concluded each of these chemicals are likely to jeopardize the continued existence of some listed Pacific salmonids, and adversely modify designated critical habitat of some listed salmonids. Also in 2012, NMFS concluded EPA's proposed registration of thiobencarb, an herbicide authorized for use in California only on rice, is not likely to jeopardize the continued existence or adversely modify the designated critical habitat of listed Pacific salmonid species. Finally, in 2015 NMFS concluded that the EPA's proposed registration of the pesticide active ingredient diflubenzuron is likely to jeopardize the continued existence of 23 ESA-listed Pacific salmonid species and is likely to destroy or adversely modify designated critical habitat of 23 listed Pacific salmonids. Also in this opinion, NMFS found that the active ingredients fenbutatin oxide and propargite are each likely to jeopardize the continued existence and likely to destroy or adversely modify designated critical habitat of 21 ESA-listed Pacific salmonid species.

Through these opinions, we learned that exposure to some of the chemicals whose discharges could be authorized by EPA's PGP has been demonstrated to have physical, physiological, or neural effects on individuals that have been exposed and these effects alter the growth, survival, fecundity, and behavior of individuals resulting in increased probability of being captured and killed by predators.

Because the proposed PGP will authorize discharges of formulations of pesticides on, over or near waters of the U.S., NMFS, in this opinion and the opinions summarized above, consider those components of formulations that might be toxic to endangered or threatened species under our jurisdiction as integral to the actions we evaluate. Piperonyl butoxide, nonylphenol and nonylphenol polyethoxylates are examples of "inert" ingredients that may be formulated in pesticide products or added as adjuvant ingredients during pesticide applications. Piperonyl butoxide is a common synergist in formulations of synthetic pyrethroids. Nonylphenol and nonylphenol polyethoxylates are common ingredients in detergents, cosmetics and other industrial products. Toxicity evaluations using the freshwater amphipod, *Hyalella azteca*, demonstrated a piperonyl butoxide-pyrethrin mixture to be "very highly toxic" under EPA's classification system. A national survey of streams found that nonylphenol was among the most common organic wastewater contaminants in the U.S. and was detected in more than 50 percent of the samples tested. The median concentration of nonylphenol in streams was 0.8 µg/L and the maximum concentration detected was 40.0 µg/L. Related compounds were also detected at a relatively high frequency (Kolpin et al. 2002).

Table 15. Conclusions for ESA section 7 consultations identifying risk of pesticide re-registration to ESA-listed salmonids (Pesticides identified in annual reports are in boldface).

Use	Active Ingredient	Jeopardy to species?	Destruction or adverse modification to Designated Critical Habitat?	Date of opinion
Acaricide	Propargite	21 of 28 species	21 of 26 species	1/7/2015
Herbicide	Oryzalin	10 of 28 species	10 of 26 species	5/31/2012
	Pendimethalin	16 of 28 species	14 of 26 species	5/31/2012
	Trifluralin	16 of 28 species	14 of 26 species	5/31/2012
	2, 4-D	**28 of 28 species**	**6 of 26 species**	**6/30/2011**
	Diuron	**no jeopardy**	**9 of 26 species**	**6/30/2011**
	Bensulide	3 of 28 species	3 of 26 species	8/31/2010
Insecticide	Fenbutatin-oxide	21 of 28 species	21 of 26 species	1/7/2015
	Dimethoate	5 of 28 species	5 of 26 species	8/31/2010
	Disulfoton	1 of 28 species	1 of 26 species	8/31/2010
	Ethoprop	3 of 28 species	3 of 26 species	8/31/2010
	Methidathion	12 of 28 species	11 of 26 species	8/31/2010
	Methyl parathion	8 of 28 species	8 of 26 species	8/31/2010
	Naled	**22 of 28 species**	**20 of 26 species**	**8/31/2010**
	Phorate	15 of 28 species	14 of 26 species	8/31/2010
	Phosmet	20 of 28 species	23 of 26 species	8/31/2010
	Carbofuran*	22 of 28 species	20 of 26 species	3/31/2009
	Carbaryl*	**22 of 28 species**	**20 of 26 species**	**3/31/2009**
	Methomyl*	18 of 28 species	16 of 26 species	3/31/2009
	Chlorpyrifos*	**27 of 28 species**	**25 of 26 species**	**11/18/2008**
	Diazinon*	27 of 28 species	25 of 26 species	11/18/2008
	Malathion*	**27 of 28 species**	**25 of 26 species**	**11/18/2008**
Insecticide and fungicide	Diflubenzuron	23 of 28 species	23 of 26 species	1/7/2015

*Malathion, Diazinon and Chlorpyrifos were remanded back to NMFS and carbofuran, carbaryl, and methomyl are scheduled for reanalysis

Analyses in EPA's Biological Evaluation

EPA's BE assessment identified numerous pesticide scenarios that resulted in elevated acute and chronic risk for individuals representing ESA-listed species and the essential biological elements of their designated critical habitat (Table 14). Many RQs were orders of magnitude greater than the LOC, suggesting population level effects are likely to occur. While the BE did not specify the types of sublethal responses represented by the chronic endpoints selected for their analysis, NMFS' review of the source documentation used by EPA in developing its BE confirms that these included measures representing the assessment endpoints of growth (e.g., length, weight) and fecundity (e.g., number of viable eggs), but not behavior. The EPA does not typically evaluate effects on behavior inits assessments because the linkage between individual effects and population-level effects is *"...uncertain and not quantitative given our present state of knowledge."* However, implications for behavior affecting predation vulnerability and habitat use (e.g., avoidance) as indicated for salmonids in NMFS' prior consultations suggest such effects could also occur in other ESA listed species under NMFS' jurisdiction.

NMFS' also notes that while EPA's assessments incorporated conservative measures, they did not integrate the risk of the actual pesticide mixtures (adjuvants, surfactants and synergists) used under the PGP use patterns. The assessment also did not take into account multiple exposures occurring during the course of the season or over the spatial extent of individual home ranges or migration routes.

Risk Characterization Summary

The species jeopardy and designated critical habitat adverse modification determinations in prior NMFS opinions for pesticide re-registrations and the analyses in EPA's BE indicate that pesticide discharges under these use patterns will result in exposures to toxicants that will affect the survival and fitness of individuals through direct mortality, reduced growth, altered behavior, and reduced fecundity of salmonids, sea turtles, rockfish, sturgeon, coral, and Nassau grouper. Further, discharges under these use patterns are expected to result in exposures to toxicants that will affect the survival and fitness of individuals through reduction in extent of inhabitable area/avoidance and reduction in prey species, affecting the prey component of designated critical habitat essential features for the following species: leatherback sea turtle, southern resident killer whale, green sturgeon, eulachon, bocaccio, yelloweye rockfish, steelhead, and chum, sockeye, chinook, and coho salmon.

> ***Taking into consideration that: (1) the composition, timing, frequency and location of discharges for use patterns eligible for coverage under the 2016 PGP are unknown for a majority of the discharges to be authorized, (2) previous NMFS opinions have found jeopardy and adverse modification of designated critical habitat on several of pesticides used under PGP-eligible use patterns, and (3) the BE analyses included RQs that were many orders of magnitude greater than the LOC EPA uses to evaluate exposures, NMFS concludes that:***

- ***Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in exposures to toxicants that will affect the survival and fitness of individuals through:***
  - ***direct mortality***
  - ***reduced growth***
  - ***altered behavior***
  - ***reduced fecundity (i.e., reduced reproductive output or offspring survival)***
- ***Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in exposures to toxicants that will affect the survival and fitness of individuals through:***
  - ***reduction in extent of inhabitable area/avoidance***
  - ***reduction in prey species***

- ***Pesticide discharges under the four use patterns eligible for coverage under the PGP will result in adverse effects to designated critical habitat features that are essential to the conservation of the species***

8.2 Programmatic Analysis

Because the risk characterization concluded that exposures to pesticides and use patterns eligible for coverage under the 2016 PGP potentially cause adverse effects to assessment endpoints, and therefore to population-level effects for *NMFS' Listed Resources of Concern* and adverse effects to the conservation value of designated critical habitat designated for these species. The conclusion presented in EPA's BE is that the additional requirements provided in the proposed 2016 PGP will likely reduce the potential for adverse effects to ESA-listed species from pesticide applications under FIFRA labeling. The following excerpts from EPA's BE describes the mechanisms through which the PGP accomplishes this goal.

> *"...Both the Services can review NOIs and request EPA to put permit coverage on hold and recommend protective measures prior to discharge authorization..."*
>
> *"...Applicators must minimize the discharge of pesticides to waters of the U.S. from the application of pesticides through the use of Pest Management Measures, and, to the extent not determined by the Decision-maker, use only the amount of pesticide and frequency of pesticide application necessary to control the target pest, and use equipment and application procedures appropriate for this task..."*
>
> *"...Applicators must perform regular equipment maintenance (e.g., calibration, cleaning and repair) to ensure correct application as required by pesticide labels and minimize the potential for leaks, spills, and unintended/accidental release of pesticides from pesticide containers into waters of the U.S. ..."*
>
> *"...Decision-makers required to submit an NOI must apply IPM-like practices, which include assessment of alternatives to pesticide use, identification of action thresholds, development of species-specific control strategies, source reduction; pre-application surveillance to determine whether pesticide use is necessary, post-application surveillance, and the minimization of environmental impacts..."*
>
> *"...The requirement that no discharge may cause or contribute to an excursion of any applicable numeric or narrative federal, state, territory, or tribal water quality standard..."*
>
> *"...The requirement of post-application visual surveillance of the application area to determine whether pesticide application was effective and notification to the permitting authority if adverse effects are observed..."*
>
> *"...Pesticide discharge management plans requirements that include problem identification, pest management option evaluation, and spill and adverse incident response procedures ..."*
>
> *"...Corrective action requirements (including documentation and reporting provisions) ..."*
>
> *"...Annual reporting requirements standard conditions that address reporting, including 24-hour reporting..."*

In its BE, EPA states that these requirements are expected to result in more environmental awareness regarding the pesticide use patterns, and increase the use of non-chemical pest controls or pesticides and application methods that are less harmful to non-target species. In addition, the BE states that the record keeping requirements in the PGP enhance the availability of information that could be useful in further reducing the likelihood of impacts on ESA-listed species and designated critical habitat under NMFS' jurisdiction. EPA indicates that information reported as part of NOIs required of certain applicators in the proposed action could help identify future permit refinements that will further reduce potential impacts on non-target species while still having their intended benefit of reducing threats of invasive species, human health diseases, and minimizing pest damage. In the 2016 BE, EPA stated that the reporting requirements under the PGP will provide additional opportunities for adaptive management with respect to minimizing impacts on listed aquatic and aquatic-dependent species.

8.2.1 Programmatic Analysis Questions

The issuance of the proposed PGP is treated as a permitting "program" that would authorize discharges of pesticide pollutants, along with the interrelated actions of discharges of pesticides not included in the definition of pesticide pollutants, within the action area during multiple, independent events conducted by multiple, independent permittees over a five-year period. Below we answer the questions that consider if the PGP can be implemented in a manner that insures they do not jeopardize ESA-listed species or destroy or adversely modify designated critical habitat.

Scope: *Has the PGP been structured to reliably estimate the probable number, location, and timing of the discharges that would be authorized by the program to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur?*

In its 2016 BE, EPA states that estimating past pesticide usage is "not feasible" relative to agricultural pesticide use due to the limited data for these use patterns. In its 2011 BE, EPA expected this information to be gathered through the implementation of the PGP's NOI and annual reporting requirements. EPA's NOI and annual reporting requirements provided insight into the number, location and timing of PGP-authorized activities. However, NOIs are submitted for only a very small fraction of discharges. Most pesticide discharges are automatically covered without filing an NOI. EPA estimates the total number of pesticide dischargers under the PGP to be about 35,000. About 350 NOI were submitted under the PGP, so only about 1 percent of PGP-authorized dischargers can be identified through NOI.

NOI are required from all Decision-makers expecting to discharge to waters of the U.S. where *NMFS' Listed Resources of Concern* occur. The current definition of *NMFS' Listed Resources of Concern* in the 2016 PGP includes only those species that were listed prior to 2011[23] and does not include coral species[24]. The PGP's definition of *NMFS' Listed Resources of Concern* will need to be updated if implementation of the PGP is to produce information that allows EPA to reliably estimate the probable number, location, and timing of the discharges to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur.

Not all PGP-authorized dischargers are required to file an NOI. For this reason, EPA's ability to estimate the scope of the discharges authorized by the program to waters where *NMFS' Listed*

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[23] NMFS makes the distinction between "NMFS ESA-listed species of Concern" and "ESA-listed species and designated critical habitat under NMFS' jurisdiction" to indicate EPA's existing definition of species they intend to protect under the PGP (i.e., listed prior to 2011 and excluding coral) versus the ESAESA-listed species currently under NMFS' jurisdiction (i.e., "ESA-listed species and designated critical habitat under NMFS' jurisdiction")

[24] At that time, NMFS agreed with EPA's conclusion that these species would not be exposed to PGP discharges. NMFS has reconsidered that exposure scenario.

Resources of Concern occur is related to the compliance of Decision-makers with the requirement to file an NOI for planned discharges to such waters (see section 0). Discharges are not covered under the PGP if they fail to file an NOI when required to do so. In such cases, the Decision-maker violates the CWA when their decisions result in unauthorized discharges to waters of the U.S. Because not all discharges are required to file an NOI under the PGP, the availability of the PGP may result in inadvertent violations of the CWA by: 1) Decision-makers who fail to self-identify as a Decision-maker and, therefore make no determination as to whether an NOI is required because they expect automatic coverage, or 2) Decision-makers who do file an NOI, but incorrectly conclude that *NMFS' Listed Resources of Concern* are absent from their pest management area. Discharges made under these circumstances are not covered by the PGP and the consequences of such discharges are indirect effects of EPA's issuance of the PGP.

Current information resources for permit applicants to use to identify where these species occur is not up to date and require examining a large volume of information provided in the form of a series of documents. These include maps of varying detail and lists of applicable receiving waters, including detail to the level of individual streams and creeks. In addition, the current PGP materials do not identify coastal waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur: Plum Island sound at the mouth of the Merrimack river or coastal waters of Cape Cod Bay. Similarly, current PGP resources do not identify coastal waters of Puerto Rico, where Nassau grouper and ESA-listed coral species may be exposed.

Cases where discharges in violation of the CWA were made as a result of failure to file an NOI for the PGP when one was required, or were ineligible for coverage due to the selection of an incorrect ESA eligibility criterion, were not identified by EPA under the 2011 PGP. There is no evidence whether EPA actively tried to identify unintentional violators and bring them into compliance with the CWA through the PGP, and there is no mechanism under the PGP to track dischargers expecting coverage, but not required to file an NOI. NMFS is not confident that pesticide discharges to waters where *NMFS' Listed Resources of Concern* occur are compliant with EPA's requirement that Decision-makers for such discharges file an NOI or select the correct ESA Eligibility Criterion.

EPA relied on outreach to inform the regulated community and state and federal regulators of the conditions under which they are required to file an NOI (P. Chumble, USEPA Office of Water, Water Permits Division, pers. comm. to P. Shaw-Allen, NMFS OPR, July 25, 2016). The following sections describe NFMS expectations for outreach effectiveness as it relates to EPA's ability to estimate the number, locations, and timing of discharges to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur, including the subset represented by EPA's current definition of *NMFS' Listed Resources of Concern*.

Federal Facilities within the States of Delaware and Washington

NMFS expects that, as trustees for public lands, Federal Decision-makers will file an NOI when ESA-listed species occur within their pesticide management areas. Thus, for the states of Delaware and Washington, EPA will be able to estimate the number, location, and timing of the discharges to waters where *NMFS' Listed Resources of Concern* for discharges made by federal Decision-makers in the states of Washington and Delaware.

The District of Columbia

Pesticide discharges affecting the Potomac River in the District of Columbia are regulated by the District's Department of Energy and Environment. The department website includes links to

EPA's PGP website and does not provide any additional information. Only two NOIs were filed from the area, one for Rock Creek Park, where sturgeon would not occur and one, certifying eligibility under Criterion F, for Theodore Roosevelt Island in the Potomac River, where sturgeon do occur. In the District of Columbia, the shores of the Potomac are bordered by the Chesapeake and Ohio Canal National Historical Park north of Georgetown University and to the south by other Federally-owned District Lands (e.g., Bolling Airforce Base, memorials, East Potomac Park) and Washington National Airport. Given the dominance of federal lands along the Potomac River shoreline within the District of Columbia, NMFS expects that NOI will be filed for PGP-eligible discharges to the Potomac. Thus EPA will be able to estimate the number, location, and timing of the discharges to waters where *NMFS' Listed Resources of Concern* occur in this area.

Indian Country Lands

Relatively few NOI were filed for PGP-eligible discharges on Indian Country Lands. Eight were filed from Washington State, three from Idaho, and one from Oregon. These were distributed among pest control districts, federal Operators, and local governments. While some tribes have pesticide codes and their own tribal programs, the EPA works cooperatively with tribes to implement pesticide programs on reservations. EPA's involvement with pesticide use on tribal lands leads NMFS to expect that an NOI will be filed for PGP-eligible discharges that may expose *NMFS' Listed Resources of Concern*, thus EPA will be able to estimate the number, location, and timing of discharges to waters where such species occur in in Indian Country Lands.

Massachusetts

It appears that outreach may not have reached all the Decision-makers in Massachusetts. While the 2011 PGP ESA guidance materials identified the Connecticut River downstream of Turners Falls Dam as waters where the endangered shortnose sturgeon occur, the only NOI for pesticide applications to or near the Connecticut River acknowledging the presence of ESA-listed species is for the city of Holyoke. One other NOI was found to include discharges to the Connecticut River, but did not acknowledge the presence of *NMFS' Listed Resources of Concern*. NMFS also notes that the three counties through which the river flows are not included in a mosquito control district, leaving a gap in the usual regulatory Decision-makers for this use pattern in the state. Taken with EPA's outreach to the regulated community and state and federal regulators, this suggests that local Decision-makers in cities and towns along the Connecticut River may be engaging in PGP-eligible pest control, but are unaware of their need to file an NOI under the PGP.

NMFS explored the Commonwealth of Massachusetts websites to determine if information on the PGP was readily available. While the Massachusetts Department of Agricultural Resources (MDAR) is identified as the state pesticide regulator, its website makes no mention of the PGP or other requirements when discharging to waters of the U.S. NMFS asked MDAR whether and how applicators, particularly small ones, are made aware of the PGP coverage/requirements for coverage. The MDAR indicated that, since the PGP is an NPDES permit, outreach to the local regulated community was the responsibility of the Massachusetts Department of Environmental Protection (MDEP) (S.E. Antunes-Kenyon, Pesticide Operations Coordinator, MDAR, pers. comm. to T. LaScola Director, Division of Crop and Pest Services, MDAR, Forwarded to P. Shaw-Allen, NMFS OPR, July 14, 2016). In turn, MDEP indicated that, for the 2011 PGP, they made presentations to the only two companies licensed to apply pesticides to water in the state and that Decision-makers must use one of these applicators for discharges covered under the 2011 PGP (R. Kubit, MDEP, Division of Watershed Management, pers. comm. to P. Shaw-Allen, NMFS OPR, September 7, 2016). MDEP also indicated that conservation commissions within each municipality are expected to inform Decision-makers when their pest control activities require an NOI. A subsequent conversation with Eugene Benson, Director of the Massachusetts Association of Conservation Commissions revealed that the Association was not aware of the PGP or this expectation of its commissions (Eugene Benson, Director of the Massachusetts Association of Conservation Commissions, pers. comm. to P. Shaw-Allen, NMFS OPR, September 12, 2016).

MDEP and Mr. Benson both noted that a state permit is required for aquatic weed control and coordination with the Massachusetts Natural Heritage and Endangered Species Program occurs when discharges may affect endangered and threatened species in waters of concern flagged by that program. Coordination with this state program does not necessarily mean NMFS ESA concerns would be evident because the waters flagged by the Massachusetts Natural Heritage and Endangered Species Program do not include some of the waters where NMFS requires NOI for EPA General Permits. Further, this program is responsible for advising on compliance with state regulations, not federal regulations. NMFS encountered this issue when consulting on EPA's Multisector General Permit. Eve Schluter, Chief of Regulatory Review, Natural Heritage and Endangered Species Program, recognized this issue and has added the following language to the programs stock language for letters responding to regulatory inquiries (E. Schluter, Chief of Regulatory Review, Natural Heritage and Endangered Species Program, pers. comm. to Pat Shaw-Allen, November 17, 2015):

> *If the purpose of your inquiry is to generate a species list to fulfill Endangered Species Act information requirements for a permit, proposal, or authorization of any kind from a federal agency, it is strongly recommended that you obtain your species lists related to your location data from both the National Marine Fisheries Service at (978)281-9328 and the U.S. Fish and Wildlife Service's Information for Planning and Conservation website (IPaC).*

NMFS anticipates that inclusion of the above statement makes it more likely that NOI will be filed for PGP-authorization of pesticide discharges for the control of aquatic weeds in areas where *NMFS' Listed Resources of Concern* occur. However, there is no process to identify inquiries that were for a federal permit or whether the services were contacted in such cases.

Based on the NOI for the 2011 PGP, the aquatic animal pest use pattern in Massachusetts has been limited to control of aquatic invertebrates in cranberry bogs, where NMFS' species of concern are not expected to occur. There is no guarantee this will not change during the 2016 permit term. NMFS expects that state and federal agencies, who were included in EPA outreach efforts, would be the Decision-makers for forest canopy pest control activities. For this use pattern, NOI are expected to be filed and EPA will be able to estimate the number, location, and timing of the discharges to waters where *NMFS' Listed Resources of Concern* occur. While several mosquito control districts filed NOI in Massachusetts, there appears to be an outreach gap for local government Decision-makers under the mosquito and flying pest control use pattern in areas of Massachusetts where mosquito control districts have not yet been established. These areas are the three counties through which the Connecticut River flows.

New Hampshire

The state of New Hampshire requires a special permit from its Division of Pesticide Control for any pesticide discharge to surface waters under Chapter Pes 600, Aquatic Application of Pesticides under RSA 430:31, Pesticides Controls. The State Pesticides Control Board, which oversees the activity of the Division of Pesticide Control, includes a representative from the State Department of Fish and Game. Given EPA's outreach to state regulators and the State's own regulation of pesticide discharges to surface water, NMFS expects State Pesticides Control Board will ensure that NOI will be filed for pesticide discharges to waters of the U.S. where *NMFS' Listed Resources of Concern* occur. Thus, EPA will be able to estimate the number, location, and timing of the discharges to these waters.

Idaho

The State of Idaho includes PGP information on their Department of Environmental Protection website and a link to the PGP on the main page of the Department of Agriculture Pesticides and Chemigation Program's website. The Pesticides and Chemigation Program annual report also indicates that the group made a concerted effort to stay involved in the development of the 2011 PGP and has a future goal to coordinate with EPA and industry on the PGP and biological options for pesticides to protect ESA-listed salmonids. EPA region 10 worked with NMFS to develop best management practices for applications to waters where *NMFS' Listed Resources of Concern* occur. Idaho includes seven mosquito and vector control districts covering all 44 counties. Forest canopy pest applications would likely be directed by the U.S. Forest Service or a state agency. NOI are expected to be filed for these use patterns. Given the breadth of PGP information provided on state websites, complete coverage for the state under mosquito control districts, the involvement of EPA with NMFS in addressing ESA concerns, NMFS expects that local, state, and federal Decision-makers, regardless of use pattern, will file NOI when ESA concerns require that they do so. However, this is not necessarily the case for all private Decision-makers expecting coverage under the PGP.

Puerto Rico

While elkhorn and staghorn coral were listed as threatened species at the time of the 2011 PGP issuance, they were not included among the *NMFS' Listed Resources of Concern* in the PGP because, at that time, NMFS agreed with EPA that exposures to PGP-authorized discharges were unlikely. Since 2011 NMFS has listed the Nassau grouper and an additional five species of corals that occur in Puerto Rico waters. Four NOI were filed prior to 2016 by Decision-makers in Puerto Rico. Three of these were for control of aquatic weeds by individual irrigation districts and one for control of mosquitoes for all 78 municipalities of the island of Puerto Rico. All four certified under Criterion A, no species present. This contrasts with a 2016 Criterion D NOI, declared pest emergency where NMFS resources of concern occur, filed by the Centers for Disease Control for the application of naled, over two of these municipalities, Ponce and San Juan. While NMFS does not have ESA-listed species that occur in inland waterways that may be affected by these discharges, coral reefs (and Nassau grouper) occur close to shore in some areas (<5 meters, Figure 7). Furthermore, as while the PGP does not cover drift or off-site transport, offsite transport is an indirect effect of PGP authorizations. NMFS expects that pesticide transport from PGP-authorized discharges that were not identified in the NOI from coastal municipalities may reach areas where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur as direct and indirect exposures resulting from the PGP-authorized discharges. Given the potential for exposures to PGP-authorized discharges in Puerto Rico, NMSF expects that under the 2016 PGP some Decision-makers in Puerto Rico will fail to correctly identify the presence of *NMFS' Listed Resources of Concern*.

American Samoa, Guam, Johnston Atoll, Midway Island, Northern Marianas, and Wake Island

Only two terminated NOI were from the Pacific islands where EPA is the permitting authority. The terminated NOI were for the use of an anticoagulant on Wake Island to control animal pest species on Joint Base Elmendorf-Richardson. The U.S. Air Force currently administers Johnston Atoll and Wake island. Midway Island is a National Wildlife Refuge managed by the USFWS. The Northern Marianas is a commonwealth territory. Guam and American Samoa are unicorporated territories. The 2011 PGP includes permit conditions only for the island of Guam. Pesticide applications eligible for coverage under the PGP in the remote Pacific Islands are expected to be infrequent. Given the federal involvement in Johnston Atoll, Wake Island and Midway Island, NOI are expected to be filed when discharges from these areas may expose ESA-listed species and designated critical habitat under NMFS' jurisdiction. The effectiveness of outreach to non-federal Decision-makers on these islands is uncertain.

Summary

While some areas where EPA is the permitting authority are more likely to produce NOI where necessary to address ESA-concerns and allow EPA to estimate the number, location, and timing of the discharges to waters where *NMFS' Listed Resources of Concern* occur (e.g., Federal Operators, Tribes working in concert with EPA), gaps are evident for the Connecticut River Valley of Massachusetts, Puerto Rico and possibly the remote Pacific island territories. Gaps are also expected to occur where private Decision-makers fail to file NOI for discharges to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur. NMFS concludes that these gaps result in an uneven ability to estimate the number, location, and timing of the discharges to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur. Ultimately, because not all dischargers are required to file NOI and decisions not to file an NOI are not tracked, Decision-makers from any state or territory may assume they are automatically covered under the PGP after failing to correctly identify the presence of ESA-listed species and designated critical habitat under NMFS' jurisdiction in their pest management area.

NMFS concludes that the PGP, as currently written, will not enable EPA to reliably estimate the probable number, location, and timing of the discharges that would be authorized by the program to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur because EPA's definition of NMFS' Listed Resources of Concern does not include the endangered Atlantic sturgeon (North Atlantic DPS) or threatened Nassau grouper and grouper, elkhorn coral, staghorn coral, lobed star coral, boulder star coral, mountainous star coral, pillar coral, and rough cactus coral.

> ***NMFS also concludes that discharges to waters of the U.S. will be made in violation of the CWA as an indirect effect of EPA's issuance of the PGP through discharges not covered by the PGP due to:***
>
> ***(1) Apparent gaps in outreach resulting in Decision-makers failing to self-identify as Decision-makers under the PGP and subsequently assuming automatic coverage and orchestrating pesticide discharges without determining whether an NOI must be filed, and***

(2) The ungainliness of the existing PGP information resources for identifying where NMFS' Listed Resources of Concern occur, resulting in failures to identify the presence of such species and failures to file an NOI or correctly certify ESA eligibility.

Stressors: *Has the general permit been structured to reliably estimate the physical, chemical, or biotic stressors that are likely to be produced as a direct or indirect result of the discharges that would be authorized (that is, the stressors produced by the actual discharges to waters of the U.S.)?*

EPA's ability to identify the stressors that are discharged to waters of the U.S. is limited to those discharges where *NMFS' Listed Resources of Concern* occur because only these NOI include information on planned discharges either directly, or indirectly, through supporting documentation (e.g., consultation documents, ESA Section 10 permits). To adequately understand the hazards posed by their multiple authorizations, EPA must also collect that information and evaluate the aggregate stressor impacts that have been authorized under the PGP.

In its 2016 BE, EPA states that estimating past pesticide usage is "not feasible" relative to agricultural pesticide use due to the limited data for these use patterns. In its 2011 BE, EPA expected this information to be gathered through the implementation of the 2011 PGP's NOI and annual reporting requirements. EPA's NOI and annual reporting requirements provided some insight into the discharged to waters where *NMFS' Listed Resources of Concern* occur (see section 0 of this opinion). Only applicants certifying ESA eligibility under Criterion D (for declared pest emergencies) and F (the applicant self-certifies that discharges are NLAA) are required to provide information on the pesticides they intend to use in their NOI prior to discharge. Applicants certifying under Criterion B (i.e., discharges are covered under another consultation) are not required to provide further detail such that a reviewer could confirm that the consultation was valid and up to date or determine whether these discharges, taken with other anticipated PGP discharges in the same area, potentially overlap and present an aggregate risk. NOI for Criterion B-certifying applicants are about as frequent as those certifying under Criterion F, totaling 22 and 27 permitees, respectively (about 8 and 10 percent of NOI). It is left to NMFS to review documentation supporting a criterion B certification and confirm that the certification was valid and that the planned discharges, taken with other PGP-authorized discharges, would not present excess aggregate risk to ESA-listed species and designated critical habitat under NMFS' jurisdiction. Under the current permit, through its NOI EPA collects information for a subset of stressors expected to be produced as result of its PGP authorizations prior to discharge to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur. It is up to NMFS review of documentation supporting NOI certified under Criteria B and E to identify the remaining stressors authorized for discharge under the PGP.

A valid and up to date consultation is a consultation for which none of the criteria for reinitiation of consultation under 50 C.F.R. § 402.16 are met, and for which the opinion or letter of concurrence has not been withdrawn or superseded as the result of a later consultation. Consultations can be either formal or informal, and would have occurred only as a result of a separate federal action. Such consultations address the effects of pesticide discharges and discharge-related activities on federally-listed threatened or endangered species and federally-designated critical habitat, and must have resulted in either: 1) A opinion from NMFS finding no likely jeopardy to ESA-listed species and no destruction/adverse modification of federally-designated critical habitat; or 2) Written concurrence from NMFS with a finding that the pesticide discharges and discharge-related activities are not likely to adversely affect federally-listed species or federally-designated critical habitat. If the consultation resulted in a opinion, the pesticide application activities for which permit coverage is being requested must be carried out in full compliance with any reasonable and prudent alternatives in that opinion, and in full compliance with the reasonable and prudent measures and terms and conditions of any incidental take statement in that opinion.

> ***NMFS concludes that through the NOI process EPA would not be able to reliably estimate the stressors that are likely to be produced as a direct or indirect result of all PGP-authorized discharges because only those NOI identifying discharges to waters where NMFS listed Resources of Concern occur will include information on the planned discharges.***

Overlap: *Has the general permit been structured to reliably estimate whether or to what degree specific endangered or threatened species or designated critical habitat are likely to be exposed to stressors of the action that the proposed permit would authorize for discharge into waters where NMFS' Listed Resources of Concern occur?*

In its 2011 BE EPA proposed to use the NOI process to estimate whether or to what degree specific endangered or threatened species are likely to be exposed to the direct or indirect effects of the activities to be authorized by the proposed permit. The NOI form contained a section where the Decision-maker self-certifies whether the planned pesticide applications will overlap with the distribution of ESA-listed species and how ESA concerns are addressed.

The ability of EPA to reliably estimate whether or to what degree specific endangered or threatened species or designated critical habitat are likely to be exposed to stressors authorized for discharge by the PGP also relies on Decision-makers to accurately identify the presence of such species and to file an NOI when it is required. EPA assumes that all NMFS ESA-listed species could potentially overlap in space and time with any use pattern and pesticide eligible for coverage under the 2016 PGP. Again, the definition of *NMFS' Listed Resources of Concern* in the 2016 draft PGP does not include coral or species listed after 2011. These include Nassau grouper, three DPS's of Atlantic sturgeon, and recently listed coral species. Waters where shortnose sturgeon occur were identified under the 2011 PGP overlap with waters where Atlantic sturgeon are found, but NMFS also includes the Taunton River, coastal waters of Cape Cod Bay, Plum Island Sound, the Piscataqua River, and the Cocheco river (tributary to the Piscataqua) among waters of concern for this species. Given the additional species and waters of concern, as currently written the PGP is not structured to allow EPA to collect reliable information on specific endangered or threatened species or designated critical habitat that are likely to be exposed to PGP-authorized discharges. During the course of this consultation, EPA worked with biologists in NMFS' regions to develop a mapping tool that includes these additional waters and will allow PGP applicants to easily check whether their pesticide management areas overlap with areas where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur. However, the use of this tool is not a component of the current draft of the 2016 PGP.

As discussed previously, review of NOI for the Commonwealth of Puerto Rico revealed differences in criterion selection for the same areas by different Decision-makers, Puerto Rico Department of Health and the U.S. Centers for Disease Control. Terminated NOI for discharges within Massachusetts rights of ways for the control of aquatic vegetation included a map of the treated area indicating one of the rights of way treated passes through the Connecticut River, which had been identified as a water of concern for shortnose sturgeon under the 2011 PGP. Annual reports filed while the NOI was active (2013-2013) also listed two hired pesticide applicators that were not either of the companies identified by MDEP as licensed to apply pesticides to water. This is one example where an NOI was filed with an inaccurate ESA Eligibility Criterion A selection. Because an NOI is not required of all PGP-covered dischargers, EPA cannot know whether or how many Decision-makers will make inaccurate determinations that *NMFS' Listed Resources of Concern* are not present leading them to not file an NOI when one is required due to ESA concerns. Such discharges would not be covered under the PGP and would therefore violate the CWA during their discharge activities. Because such decisions are not tracked, EPA would not be able to identify these dischargers and bring them into compliance with the CWA through the PGP. The probability of such errors will likely increase as additional waters of concern are included under the 2016 PGP.

> ***NMFS concludes that EPA will not be able to reliably estimate whether or to what degree specific endangered or threatened species or designated critical habitat are likely to be exposed to stressors resulting from PGP-authorized discharges due to omission of species listed as threatened or endangered under the ESA since issuance of the 2011 PGP.***

Monitoring/Feedback: *Has the general permit been structured to identify, collect, and analyze information about authorized actions that may have exposed endangered or threatened species or designated critical habitat to stressors at concentrations, intensities, durations, or frequencies that are known or suspected to produce physical, physiological, behavioral, or ecological responses that have potential individual or cumulative adverse consequences for individual organisms or essential elements of designated critical habitat?*

In order to continually identify, collect and analyze information that suggests that the discharges of pesticide on, over or near waters of the U.S. may expose endangered or threatened species or designated critical habitat to pesticide at concentrations, durations or frequencies that are known or suspected to produce physical, physiological, behavioral or ecological responses that have potential individual or cumulative adverse consequences for individual organisms or *essential elements* of designated critical habitat, the EPA proposes to require that Operators self-monitor for adverse effects resulting from these discharges. The PGP requires permittees to monitor and report any adverse incidents resulting from activities authorized by the permit. This places the responsibility for oversight largely on the permittees who would have little incentive to do so given that such observations would be a violation of the PGP and potentially result in enforcement responses by the EPA and/or subsequent loss of a pesticide applicator's license. Under the 2011 PGP, no incidents were reported.

In addition, it is unclear how an Operator will have the ability to visually detect all adverse responses to pesticide exposures to ESA ESA-listed species or their designated critical habitat. For example, while Operators might have the ability to observe the mortality of adult or juvenile listed fish, they likely would not have the resources or ability to visually detect the death of the eggs or alevins of these species. Nor would they likely have the resources or ability to observe reductions in the reproduction or growth rates of these species or other sublethal effects as a result of pesticide exposures. Adequate monitoring by the Operator that would be sufficient to insure that no adverse exposures occurred from authorized discharges would be time and resource intensive. Yet, the EPA states in its BE that:

> *"[Visual monitoring by permittees] … should provide valuable information to EPA and the States about where adverse environmental effects are occurring. This knowledge will help EPA identify where problems may remain and where improvements can be made in the next PGP."*

While we agree that these monitoring efforts may improve the PGP over time, it is unlikely that the self-monitoring and self-reporting conditions of the PGP are sufficient such that the EPA can continually identify, collect and analyze information that suggests that the discharges of pesticide on, over or near waters of the U.S. may expose endangered or threatened species or designated critical habitat under NMFS' jurisdiction to pesticide at concentrations, durations or frequencies that are known or suspected to produce physical, physiological, behavioral or ecological responses that have potential individual or cumulative adverse consequences for individual organisms or *essential elements* of designated critical habitat.

The NOI and annual reports also provide information on discharges that may have exposed endangered or threatened species or designated critical habitat to stressors at concentrations, intensities, durations, or frequencies that may have adverse consequences for ESA-listed species and designated critical habitat under NMFS' jurisdiction. However, while all Decision-makers discharging to waters where *NMFS' Listed Resources of Concern* occur are required to submit and NOI, not all are required to submit annual reports. This will result in gaps in information on the actual discharges that were made, since the NOI only identify the planned discharges and will not include the same level of detail as an annual report.

> ***NMFS concludes that EPA will not be able to collect, and analyze information about authorized actions that may have exposed specific endangered or threatened species or designated critical habitat to stressors at concentrations, intensities, durations, or frequencies that are known or suspected to produce physical, physiological, behavioral, or ecological responses that have potential individual or cumulative adverse consequences for individual organisms or essential elements of designated critical habitat because: 1) dischargers will not always be able to observe adverse responses resulting from their pesticide applications, 2) not all dischargers to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur will provide annual reports identifying the actual discharges that were made, and 3) there is a disincentive for discahrgers to report incidents due the the potential for negative consequences.***

Responses of Listed Resources: *Does the general permit have an analytical methodology that considers: a) the status and trends of endangered or threatened species or designated critical habitat; b) the demographic and ecological status of populations and individuals of those species given their exposure to pre-existing stressors in different drainages and watersheds; c) the direct and indirect pathways by which endangered or threatened species or designated critical habitat might be exposed to the discharges to waters of the United States; and d) the physical, physiological, behavior, and ecological consequences of exposing endangered or threatened species or designated critical habitat to stressors from discharges at concentrations, intensities, durations, or frequencies that could produce physical, physiological, behavioral, or ecological responses, given their pre-existing demographic and ecological condition?*

Section 7(a)(2) of the ESA requires Federal Agencies to use the best scientific and commercial data available to insure that any action authorized, funded or carried out by such Agency is not likely to jeopardize the continued existence of any listed resource. EPA requires permittees to be responsible for complying with this requirement by determining whether the specific actions those permittees carry out, as authorized by the proposed general permit, may affect ESA ESA-listed species or designated critical habitat. However, it is unlikely that the majority of Decision-makers would have access to the best scientific and commercial data available, or the necessary training and experience, to make such determinations. For example, it is NMFS' experience with NOIs filed in the state of Idaho by non-federal decision-makers certifying under Criterion F frequently incorrectly self-certify that their discharge is not likely to adversely affect *NMFS' Listed Resources of Concern* (D. Mabe, Idaho State Director, NMFS Protected Resources, pers. comm. to P. Shaw-Allen, NMFS, June 24, 2015).

The 2016 PGP ESA Eligibility Criteria addresses potential issues by either directly or indirectly incorporating NMFS expertise to supply the necessary analytical methodology to evaluate whether ESA-listed species and designated critical habitat under NMFS' jurisdiction may become exposed to and respond adversely to planned discharges.

An NOI certification under Criterion B requires that prior consultation with NMFS determined that the discharge is not likely to adversely affect ESA-listed species or designated critical habitat. Certification under Criterion C requires that the discharges be authorized under a Habitat Conservation Plan under Section 10 permit under the ESA. In both cases, NMFS has already assessed the implications of planned discharges and concludes that they do not pose ESA concerns.

An NOI certification under Criterion D is required for discharges performed in response to a Declared Pest Emergency and the NOI containing information about the discharge that is occurring is filed within 15 days of initial discharge, making it available for NMFS review. NMFS has 30 days to advise EPA whether the discharge(s) described in the NOI meets the eligibility criterion of not likely to adversely affect *NMFS' Listed Resources of Concern*; whether the eligibility criterion could be met with additional conditions; or whether the eligibility criterion is not met. EPA will advise the Decision-maker within 15 days after receiving notification from NMFS whether the discharge or discharges qualify for coverage beyond the 60-day authorization provided under the permit. If EPA identifies additional conditions to qualify discharges as eligible for coverage beyond 60 days under the permit, those conditions remain in effect for the life of the permit. EPA expects to rely on NMFS' determination in identifying eligibility for continuing authorization, either with or without additional conditions.

Review by NMFS is also indicated for Decision-makers certifying their NOI under ESA Eligibility Criteria E or F. Certification under Criterion E requires confirmation from a NMFS Regional Office prior to NOI submission that discharges are not likely to adversely affect "*NMFS' Listed Resources of Concern*." The NOI must include documentation of NMFS acknowledgment that they have determined the discharges are not likely to adversely affect *NMFS' Listed Resources of Concern* and any additional measures NFMS requires for permit eligibility. To maintain eligibility under the PGP for those discharges, those additional measures must be implemented for the duration of coverage under the PGP.

If a discharger self-certifies that discharges are not likely to adversely affect *NMFS' Listed Resources of Concern* under Criterion F, the NOI is required to include information on the pesticides and application protocols used to facilitate review of the discharge along with the rationale supporting the determination whether the discharge is likely to adversely affect *NMFS' Listed Resources of Concern*. The NMFS will, within 30 days of submission of the NOI, advise EPA whether it believes the planned discharges meet the eligibility criteria of not likely to adversely affect *NMFS' Listed Resources of Concern*, whether the eligibility criterion could be met with additional conditions; or whether the eligibility criterion is not met. EPA will advise the Decision-maker as to whether the intended discharges qualify to proceed under the General Permit or whether an individual permit will be required. EPA expects to rely on NMFS' determination in identifying eligibility for authorization, either with or without additional conditions. While the PGP indicates that if EPA does not contact the discharger within 30 days, they may assume that the discharge is authorized without further conditions. The PGP does not indicate that EPA assumes that ESA concerns have been adequately addressed in cases where NMFS has not responded to the NOI.

> ***NMFS concludes that its review of NOI for EPA incorporates into the PGP an analytical methodology that considers: a) the status and trends of endangered or threatened species or designated critical habitat; b) the demographic and ecological status of populations and individuals of those species given their exposure to pre-existing stressors in different drainages and watersheds; c) the direct and indirect pathways by which endangered or threatened species or designated critical habitat might be exposed to the discharges to waters of the U.S.; and d) the physical, physiological, behavior, and ecological consequences of exposing endangered or threatened species or designated critical habitat to stressors from discharges at concentrations, intensities, durations, or frequencies that could produce physical, physiological, behavioral, or ecological responses, given their pre-existing demographic and ecological condition.***

Compliance: *Does the general permit have a mechanism to reliably determine whether or to what degree Operators have complied with the conditions, restrictions or mitigation measures the proposed permit requires when they discharge to waters of the U.S.?*

The EPA must have an effective means of oversight to know or be able to determine reliably whether or to what degree Operators are complying with the conditions, restrictions or mitigation measures the proposed general permit requires when they discharge pesticide on, over or near waters of the U.S. Under the conditions of the permit, any Operator would be required to allow EPA or an authorized representative to: 1) Enter the premises where a regulated facility or activity is located or conducted; 2) Have access to and copy, at reasonable times, any records that must be kept under the conditions of the permit; 3) Inspect at reasonable times any facilities, equipment, practices, or operations regulated or required under the permit; and 4) Sample or monitor at reasonable times, for the purposes of assuring permit compliance or as otherwise authorized by the Clean Water Act, any substances or parameters at any location. However, the proposed PGP does not provide information regarding the level of oversight EPA plans to carry out. The proposed general permit only states that the Operator must allow EPA to do so.

It is not apparent whether EPA carried out inspections for the 2011 PGP. Since the timing and location of pesticide discharges is often determined by weather conditions and other logistical concerns, EPA is unlikely to be able to schedule an inspection during an actual discharge. The

NMFS opinion for the 2011 PGP included a review of inspection and compliance patterns for general permits recorded in EPA databases indicating that a reduced rate of inspections likely results in a substantial number of undetected permit violations. In the absence of PGP-specific data for the 2011 permit term, EPA-issued individual and general permits from NPDES-permitted sources such as industrial and municipal wastewater, stormwater, and animal feeding operations, were taken as surrogate indicators of compliance performance for EPA-issued PGP? permits. NMFS acknowledges that the PGP-authorized discharges differ from these sources, but in the absence of inspection and compliance data for the 2011 PGP, they are the best available indicator for this aspect of permit performance. NFMS revisited this analysis for the 2016 PGP using data from EPA's Enforcement Compliance History Online database (Accessed September 4th, 2016). Among permits issued by EPA, current data indicate that dischargers with individual permits were more likely to be inspected than dischargers covered under general permits (90 percent versus 17 percent). Noncompliance rates (e.g., effluent violations, reporting violations) were higher among inspected permits, and highest among individual permit holders (Table 16). To make sure that noncompliance rates among inspected permits were not inflated by inspections made in response to reporting violations or reported effluent exceedences (i.e., for-cause inspections), a reanalysis of these data excluded those permits with inspections coded as "case development," "diagnostic," or "non-compliance rates." Very few inspections for cause were identified among the data. The results of this second analysis did not indicate that for-cause inspections inflated noncompliance rates. The occurrence of noncompliance among dischargers that are not inspected is identified through required reporting indicating effluent exceedences, methods violations, or extraordinary discharge incidents and through failures to meet reporting requirements. Overall, noncompliance among inspected dischargers is higher than for uninspected dischargers, but, when inspected, dischargers with EPA-issued general permits are less likely to be found in noncompliance than dischargers with EPA-issued individual permits. This could reflect the dominance of wastewater dischargers among individual permits and a systematic exclusion of dischargers with problematic discharges from coverage under General Permits by EPA.

Table 16. Noncompliance rates among inspected and uninspected dischargers with EPA-issued permits with and without inspections made in response to violations.

Permit Type	Noncompliance Rate Among Inspected Dischargers (inspections for cause excluded)	Noncompliance Rate Among Uninspected Dischargers
EPA-issued General Permit	20 percent (20 percent)	4 percent
EPA-Issued Individual Permit	67 percent (64 percent)	33 percent

Previous investigations of general permits have examined the reliability of self-identification for permit coverage and self-reporting for permit violations. One investigation reported grossly incomplete compliance with State and EPA administered storm water general permits 10 years after implementation (Duke and Augustenborg, 2006). The researchers also determined that general permits administered by EPA attained higher compliance rates than State administered general permits. Another study found a compliance rate of 10 percent under Florida's State wide general permit. Only 14 of the 136 industries examined which should have filed an NOI did so (Cross and Duke, 2008).

Further, inspections and collection of compliance data is only possible for PGP-authorized dischargers who filed an NOI. Cases where discharges in violation of the CWA were made as a result of failure to file an NOI for the PGP when one was required were not identified by EPA under the 2011 PGP. There is no evidence whether EPA actively tried to identify unintentional violators expecting automatic coverage and bring them into compliance with the CWA through the PGP. The implications of the selective requirement to file an NOI were discussed in context of our analysis of EPA's understanding of the scope of its action (Item 1 in this section).

Given the findings the analysis reaffirming the importance of inspections in detecting violations and the expectation that while the work of Duke and Augustenborg (2006) may generally reflect the behavior of a subset of dischargers expecting coverage under the PGP, NMFS expects that the EPA cannot ensure compliance with the protective provisions of NPDES general permits.

> ***NMFS concludes that EPA is not likely to know or be able to reliably determine whether or to what degree Decision-makers comply with the conditions, restrictions or mitigation measures required under the 2016 PGP because the PGP does not specify the level of oversight or inspections that will occur and there does not appear to include a plan to ensure compliance or for identifying cases where an NOI was required but not submitted.***

Adequacy of Controls: *Does the PGP provide EPA a mechanism to prevent or minimize endangered or threatened species or designated critical habitat from being exposed to stressors from discharges at concentrations, durations, or frequencies that a) are potentially harmful to individual listed organisms, populations, or the species, or; b) have ecological consequences that are potentially harmful to individual listed organisms, populations, species or the physical and biological features of their designated critical habitat?*

Controls preventing or minimizing exposure that are specified within the 2016 PGP itself include the requirement that applicators minimize the discharge of pesticides to waters of the U.S.

through the use of Pest Management Measures and use only the amount of pesticide and frequency of pesticide application necessary to control the target pest. Applicators are also required to perform regular equipment maintenance (e.g., calibration, cleaning and repair) to ensure correct application as required by pesticide labels and minimize the potential for leaks, spills, and unintended/accidental release of pesticides from pesticide containers. Decision-makers discharging to waters where *NMFS' Listed Resources of Concern* occur are required to submit an NOI and must apply IPM-like practices, which include assessment of alternatives to pesticide use, source reduction and pre-application surveillance to determine whether pesticide use is necessary.

As stated in Section 0, for those dischargers required to submit an NOI due to ESA concerns, the ESA Eligibility Criteria outlined in the 2016 PGP either directly or indirectly incorporates NMFS expertise. NMFS review of NOI would also identify, or have identified, any additional protective measures necessary to prevent or minimize exposure. The effect of NMFS review was demonstrated under the 2011 PGP. After determining that the controls identified in NOI by Idaho non-federal applicants certifying eligibility under Criterion F were not adequate, NMFS and EPA worked together to develop and require specific best management practices to prevent or minimize exposure. The following paragraph reviews how NMFS expertise is integrated into the ESA Eligibility Criteria and how NMFS review would introduce controls to prevent or minimize exposure.

An NOI certification under Criterion B requires that prior consultation with NMFS determined that the discharge is not likely to adversely affect ESA-listed species or designated critical habitat. Certification under Criterion C requires that the discharges be authorized under a Habitat Conservation Plan under an ESA Section 10 permit. In both cases, NMFS has already assessed the implications of planned discharges, including the need for and measures to prevent or minimize exposures, and concludes that they do not pose ESA concerns.

An NOI certification under Criterion D is required for discharges to waters of the U.S. containing *NMFS' Listed Resources of Concern* that are performed in response to a Declared Pest Emergency Situation. The NOI filed within 15 days of initial discharge is required to include information on the pesticides and application protocols used to facilitate review of the discharge along with the rationale supporting the determination whether the discharge is likely to adversely affect *NMFS' Listed Resources of Concern*, including the description of appropriate measures to be undertaken to avoid or eliminate the likelihood of adverse effects. NMFS will, within 30 days of submission of the NOI, advise EPA whether the past and planned future discharges meet the eligibility criterion of not likely to adversely affect *NMFS' Listed Resources of Concern*; whether the eligibility criterion could be met with additional conditions, including controls that would prevent or minimize exposure; or whether the eligibility criterion is not met. EPA will advise the Decision-maker within 15 days after receiving notification from NMFS whether the discharge or discharges qualify for coverage beyond the 60-day authorization provided under the permit. If EPA identifies additional conditions to qualify discharges as eligible for coverage beyond 60 days under the permit, those conditions remain in effect for the life of the permit. EPA expects to rely on NMFS' determination in identifying eligibility for continuing authorization, either with or without additional conditions.

Review by NMFS is also indicated for Decision-makers certifying their NOI under ESA Eligibility Criteria E or F. Certification under Criterion E requires confirmation from a NMFS Regional Office prior to NOI submission that discharges are not likely to adversely affect *NMFS'*

Listed Resources of Concern. A confirmation indicates that NMFS has evaluated both the potential for effects from the intended discharges and the controls applied to prevent or minimize exposure. The NOI must include documentation of NMFS' acknowledgment that they have determined the discharges are not likely to adversely affect *NMFS' Listed Resources of Concern* and any additional measures, including controls to prevent or minimize exposure, that NFMS requires for permit eligibility. To maintain eligibility under the PGP for those discharges, those additional measures must be implemented for the duration of coverage under the PGP.

If a discharger self-certifies that discharges are not likely to adversely affect *NMFS' Listed Resources of Concern* under Criterion F, the NOI is required to include information on the pesticides and application protocols used to facilitate review of the discharge along with the rationale supporting the determination whether the discharge is likely to adversely affect *NMFS' Listed Resources of Concern*. The NMFS will, within 30 days of submission of the NOI, advise EPA whether it believes the planned discharges meet the eligibility criteria of not likely to adversely affect *NMFS' Listed Resources of Concern*, whether the eligibility criterion could be met with additional conditions, including controls that prevent or minimize exposure; or whether the eligibility criterion is not met. EPA will advise the Decision-maker as to whether the intended discharges qualify to proceed under the General Permit or whether an individual permit will be required. EPA expects to rely on NMFS' determination in identifying eligibility for authorization, either with or without additional conditions. While the PGP indicates that if EPA does not contact the discharger within 30 days, they may assume that the discharge is authorized without further conditions. The PGP does not indicate that EPA assumes that ESA concerns have been adequately addressed in cases where NMFS has not responded to the NOI.

> ***NMFS concludes that components of the PGP intended to reduce discharges or promote the use of less toxic pesticides, in concert with NMFS review of NOI for EPA, serves as the mechanism to prevent or minimize the exposure of endangered or threatened species or designated critical habitat to stressors from discharges: a) at concentrations, durations, or frequencies that are potentially harmful to individual listed organisms, populations, or the species, or; b) to ecological consequences that are potentially harmful to individual listed organisms, populations, the species or essential elements of designated critical habitat.***

9 INTEGRATION AND SYNTHESIS

Here, we integrate information presented in this opinion to summarize the action in its entirety and consequences for ESA-listed species. Through the PGP, EPA would authorize discharges of pesticide pollutants on, over or near waters of the U.S. over the permit period from 2016 to 2021. The EPA estimates the total number of pesticide Decision-makers and Applicators authorized under the 2011 PGP to be about 35,000 and reported that only 357 Operators submitted a NOI. As a result, there is considerable uncertainty regarding the number, location, timing, and composition of discharges to waters of the U.S. authorized that occurred under the 2011 PGP and will occur under the 2016 PGP. Therefore considerable uncertainty remains in this consultation regarding subsequent exposures and responses under the proposed 2016 PGP.

The EPA's BE on the PGP and NMFS opinions on re-registration of several pesticides, establish that pesticides applied according to FIFRA labeling adversely affect ESA-listed species (Table 15). In many cases NMFS opinions conclude that application under FIFRA labeling jeopardizes the continued existence of such species and results in adverse modification of their designated critical habitat (see Table 15 for list of opinions). It is EPA's intention to mitigate this risk through its implementation of the PGP.

The risk analysis of this consultation concludes that, given the uncertainty in the number, location, timing, and composition of discharges, population level effects on salmonid and non-salmonid ESA-listed species in the absence of effective implementation of the protective measures under the PGP that pesticide applications made under FIFRA labelling produces discharges that result in population-level risks to ESA-listed species and designated critical habitat under NMFS' jurisdiction.

The programmatic analysis concluded that, as written, EPA will not be able to reliably estimate the probable number, location, and timing of the discharges that would be authorized by the program to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur for the following reasons:

(1) EPA's definition of *NMFS' Listed Resources of Concern* does not include the endangered Atlantic sturgeon or threatened Nassau grouper and coral species. This definition is used to identify discharges that require that an NOI be submitted and include information on planned discharges, either directly or indirectly. This incomplete definition also prevents EPA from being able to estimate whether or to what degree specific endangered or threatened species or designated critical habitat are likely to be exposed to stressors resulting from PGP-authorized discharges.

(2) EPA would not be able to reliably estimate the stressors that are likely to be produced as a direct or indirect result of all PGP-authorized discharges because only those NOI identifying discharges to waters where NMFS listed Resources of Concern occur will include information on the planned discharges.

(3) EPA is not likely to know or be able to determine whether or to what degree Decision-makers comply with the conditions, restrictions, or mitigation measures require under the 2016 PGP. This is because most PGP-authorized dischargers are automatically covered under the PGP, have no reporting requirement and thus EPA will not be able to identify and inspect a representative number of dischargers to determine compliance.

(4) The self-monitoring and self-reporting conditions of the PGP do not enable EPA to continually identify, collect, and analyze information about authorized actions that may have exposed ESA-listed species or designated critical habitat to stressors at concentrations, intensities, durations, or frequencies that are known or suspected to produce physical, physiological, behavioral, or ecological responses that have potential individual or cumulative adverse consequences for individual organisms or essential elements of designated critical habitat.

(5) Dischargers will not always be able to observe adverse responses resulting from their pesticide applications and not all dischargers will provide annual reports identifying their discharges. Because of this, the EPA would not know if exposures are occurring at concentrations, durations, or frequencies that are known, or suspected to, produce adverse effects to ESA-listed species or essential elements of designated critical habitat.

This consultation focused on those discharges that potentially expose ESA-listed species to stressors at intensities, frequencies, and/or durations that would result in adverse responses such that their loss or impairment may affect the populations to which they belong. The requirement that all Decision-makers making discharges to waters where *NMFS' Listed Resources of Concern* submit an NOI incorporates NMFS expertise either directly or indirectly to assist EPA in identifying discharges that may result in adverse effects and making sure its authorizations prevent or minimize exposures to avoid adverse effects.

The success of this approach requires that:

(1) every discharge authorized under the PGP has a Decision-maker;

(2) that Decision-maker is able to determine whether *NMFS' Listed Resources of Concern* are present in any of their pesticide management areas;

(3) that Decision-maker files an NOI when required to do so due to ESA concerns;

(4) that NMFS reviews the NOI to determine whether the eligibility criterion has been met, could be met with additional conditions, or whether the eligibility criterion is not met;

(5) that EPA relies on NMFS' determination in identifying eligibility for authorization, making any additional condition a requirement for coverage or requiring an individual permit if NMFS determined that eligibility criteria cannot be met; and,

(6) if found eligible for coverage under the PGP, that the Decision-maker proceeds with the discharges identified in the NOI and reviewed by NMFS, implementing any additional controls required for coverage.

These conditions are not necessarily met under the 2016 PGP. Discharges are not covered under the PGP if a Decision-maker fails to file an NOI when required to do so. In such cases, the Decision-maker violates the CWA upon discharge. Because not all discharges are required to file an NOI under the PGP, the availability of the PGP may result in inadvertent violations of the CWA by Decision-makers who fail to self-identify as needing to file an NOI. This may occur when Decision-maker incorrectly conclude that *NMFS' Listed Resources of Concern* are absent from their pest management area. Discharges made under these circumstances are not covered by the PGP and the consequences of such discharges are indirect effects of EPA's issuance of the PGP.

Resources for permit applicants to use to identify where these species require updating and improvement. While shortnose sturgeon are an anadromous species, current PGP resources do not include Plum Island sound at the mouth of the Merrimack river or coastal waters of Cape Cod Bay where sea turtles and sturgeon may become exposed to PGP-authorized discharges. Finally, current PGP resources do not identify coastal waters of Puerto Rico, where Nassau grouper and ESA-listed coral species may be exposed.

Cases where discharges in violation of the CWA were made as a result of failure to file an NOI under the 2011 PGP when an NOI was required were not identified by EPA. There is no evidence whether EPA actively tried to identify unintentional violators and bring them into compliance with the CWA through the PGP. Further, there is no mechanism under the PGP to track dischargers expecting coverage, but not required to file an NOI.

The EPA's issuance of the 2016 PGP without effective support for the regulated community to recognize the requirements of Decision-makers under the PGP, or make correct determinations on the presence or absence of ESA-listed species, exposes the regulated community to an increased risk of unknowingly making discharges in violation of the CWA and the ESA, thus placing ESA-listed species and designated critical habitat at risk.

Based on our evaluation of PGP implementation in the different areas where EPA has permitting authority, species vulnerable to the indirect effects of EPA's issuance of the PGP are those that occur in Idaho, Massachusetts, and Puerto Rico. Because the timing, intensity, frequency, and duration of these exposures cannot be known, NMFS must give benefit of the doubt to these species and designated critical habitat, including the physical and biological features of designated critical habitat that will be affected by toxicants. (designated by an asterisk *):

- Idaho
  - salmon, Chinook (Snake River fall-run ESU)*
  - salmon, Chinook (Snake River spring/summer-run ESU)*
  - salmon, sockeye (Snake River ESU)*
  - steelhead (Snake River Basin DPS)*
- Washington
  - Designated Critical Habitat (Chinook Salmon) for Southern resident killer whale
- Massachusetts
  - Atlantic sturgeon (Gulf of Maine DPS
  - Atlantic sturgeon (New York Bight DPS)
  - shortnose sturgeon
  - green sea turtle (North Atlantic DPS)
  - hawksbill sea turtle
  - Kemp's ridley sea turtle
  - leatherback sea turtle
  - loggerhead sea turtle (Northwest Atlantic DPS)
- Puerto Rico
  - Nassau Grouper
  - elkhorn coral
  - staghorn coral
  - lobed star coral
  - boulder star coral
  - mountainous star coral
  - pillar coral
  - rough cactus coral

10 Cumulative Effects

Cumulative effects include the effects of future state, tribal, local or private actions that are reasonably certain to occur in the action area considered in this opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, NMFS searched for information on future state, tribal, local or private actions that were reasonably certain to occur in the action area. NMFS conducted electronic searches of business journals, trade journals and newspapers using electronic search engines. Those searches produced no evidence of future private action in the action area that would not require Federal authorization or funding and is reasonably certain to occur. As a result, at the spatial and temporal scale of this programmatic action, NMFS is not aware of any specific actions of this kind that are likely to occur in the action area during the near future.

The future intensity of specific non-Federal activities in the action area is molded by difficult-to-predict future economy, funding levels for restoration activities, and individual investment decisions. However, due to their additive and long-lasting nature, the adverse effects of non-Federal activities that are stimulated by general resource demands, and driven by changes in human population density and standards of living, are likely to compound in the future. Specific human activities that may influence water quality and contribute to declines in the abundance, range, and habitats of ESA-listed species or the conservation value of designated critical habitat in the action area include the following: urban and suburban development; shipping; infrastructure development; water withdrawals and diversion; recreation, including off-road vehicles and boating; expansion of agricultural and grazing activities, including alteration or clearing of native habitats for domestic animals or crops; and introduction of non-native species which can alter native habitats or out-compete or prey upon native species.

Activities which degrade water quality will continue into the future. These include conversion of natural lands, land use changes from low impact to high impact activities, water withdrawals, pesticide pollution from agricultural applications and irrigation water return, effluent discharges, the progression of climate change, the introduction of nonnative invasive species, and the introduction of contaminants and pesticides from nonagricultural upland uses other than those covered by the PGP. Nationally, water quality in more than 36,000 miles of rivers and streams are impaired by pesticides (USEPA 2016b). While some of these impairments include persistent organochlorines that are no longer in use (e.g., DDT, chlordane), many of these pesticides are potentially discharges under the PGP (e.g., 2,4-D, carbofuran, chlorpyrifos, cypermethrin, permethrin, malathion, simazine).

Under Section 303(c) of the CWA individual states are required to adopt water quality standards to restore and maintain the chemical, physical, and biological integrity of the nation's waters. EPA must approve of state water quality standards and this approval is subject to ESA section 7 consultation. While some of the stressors associated with non-federal activities which degrade water quality will be directly accounted for in section 7 consultations between NMFS and EPA, some may be accounted for only indirectly, while others may not be accounted for at all. In particular, many non-point sources of pollution, which are not subject to CWA NPDES permit and regulatory requirements, have proven difficult for states to monitor and regulate. Non-point source pollution have been linked to loss of aquatic species diversity and abundance, coral reef degradation, fish kills, seagrass bed declines and toxic algal blooms (Gittings et al. 2013). Non-point sources of pollution are expected to increase as the human population continues to grow. States will need to address increases in non-point source pollution in the future to meet the state's approved water quality standards and designated water body use goals. Given the challenges of monitoring and controlling non-point source pollution and accounting for all the potential stressors and effects on ESA-listed species, chronic stormwater discharges will continue to result in aggregate impacts.

While specific actions were not identified, the collective impact of ongoing activities contribute to climate change and is discussed in the Comprehensive Environmental Baseline provided in Appendix B.

11 CONCLUSION

After considering the current status of ESA-listed species, the environmental baseline, the potential effects of the action, and the cumulative effects of concurrent and future nonfederal actions in context of the controls, monitoring, and feedback loops, and integration of NMFS expertise through the ESA Eligibility Criteria, it is NMFS' opinion that EPA's reissuance of the PGP will likely jeopardize the continued existence of Southern Resident Killer Whale, Chinook salmon (Snake River fall-run, Snake River spring/summer-run), sockeye salmon (Snake River ESU), steelhead (Snake River Basin), Atlantic sturgeon (Gulf of Maine and New York Bight DPSs), shortnose sturgeon, green sea turtle (North Atlantic DPS), hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtle, loggerhead sea turtle (Northwest Atlantic DPS), Nassau grouper, elkhorn coral, staghorn coral, lobed star coral, boulder star coral, mountainous star coral, pillar coral, and rough cactus coral.

After placing the current status of the designated critical habitat, critical habitat proposed for designation listing under the ESA, the environmental baseline, the potential effects of the action, and the cumulative effects of concurrent and future nonfederal actions in context of the controls monitoring and feedback loops, and integration of NMFS expertise through the ESA Eligibility Criteria, it is NMFS' opinion that EPA's reissuance of the PGP will is likely to destroy or adversely modify designated critical habitat for Chinook salmon (Snake River fall-run ESU, Snake River spring/summer-run ESU), sockeye salmon (Snake River ESU), and steelhead (Snake River Basin DPS).

12 REASONABLE AND PRUDENT ALTERNATIVE AND INCIDENTAL TAKE STATEMENT

Because we have concluded that the proposed general permit fails to comply with the requirements of section 7(a)(2) of the ESA, we have provided a Reasonable and Prudent Alternative (RPA) that would allow EPA to comply with those requirements. Regulations implementing Section 7 of the Act (50 CFR 402.02) define RPAs as alternative actions, identified during formal consultation, that: (1) Can be implemented in a manner consistent with the intended purpose of the action; (2) Can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) Are economically and technologically feasible for the action agency to implement; and (4) Would, in NMFS' opinion, avoid the likelihood of jeopardizing the continued existence of endangered or threatened species or resulting in the destruction or adverse modification of critical habitat. Because the general permit, for purposes of endangered or threatened species under NMFS' jurisdiction, authorizes discharges in the District of Columbia, Idaho, Massachusetts and New Hampshire, all Indian lands and Federal lands in Delaware, and Washington State, the RPA described below applies only in those locations. In addition, this RPA is not applicable to discharges to waters of the United States on Federal lands for which an existing consultation covers those activities.

The goal of the RPAs and RPMs below is to ensure that the potential for exposure of ESA-listed species and designated critical habitat ("NMFS' Listed Resources of Concern") to PGP-authorized discharges is accurately identified, that NMFS will receive all NOI and annual reports associated with such discharges, and that these NOI and annual reports will contain the necessary information that will allow NMFS to advise EPA on its authorization of such discharges with respect to EPA's obligations under the ESA. The RPAs will allow EPA to demonstrate that it is able to satisfy the requirements of section 7(a)(2) of the ESA by: (1) reliably estimating the probable number, location and timing of the discharges that would be authorized by the permit when NMFS Listed Resources of Concern may be exposed; (2) reliably estimating whether or to what degree specific endangered or threatened species or designated critical habitat are likely to be exposed to authorized discharges and (3) reliably determining whether or to what degree operators have complied with the conditions of the permit. By extension, effective identification of the potential for ESA concerns and subsequent engagement of NMFS expertise, where necessary, contributes to EPA's ability to prevent or minimize endangered or threatened species or designated critical habitat from being exposed to: a) stressors from discharges at concentrations, durations, or frequencies that are potentially harmful to individual listed organisms, populations, or the species or the essential features of designated critical habitat, or; b) ecological consequences that are potentially harmful to individual listed organisms, populations, the species or the essential features of designated critical habitat.

12.1 RPA

The 2016 PGP Reasonable and Prudent Alternative (RPA) consists of two elements that EPA must implement in their entirety to ensure that PGP-authorized actions are not likely to jeopardize the continued existence of endangered or threatened species under the jurisdiction of NMFS or destroy or adversely modify critical habitat that has been designated for any of these species. This RPA ensures that EPA complies with the requirements of section 7(a)(2) of the ESA.

12.1.1 2016 PGP RPA Element One

Rationale: While the 2011 and 2016 PGPs provide an additional layer of protection over restrictions provided by the FIFRA registrations, the programmatic analysis in the biological opinion for the 2011 PGP concluded that EPA's issuance of the PGP was likely to jeopardize the continued existence of 33 endangered or threatened species under NMFS' jurisdiction and result in the destruction or adverse modification of critical habitat designated for 29 of those species. Since issuance of the 2011 PGP, NMFS has listed additional species that occur within the action area as threatened or endangered under the ESA. This includes Nassau grouper, three DPS of Atlantic sturgeon, and 13 domestic coral species. In addition, NMFS issued updated ESA-listings for two DPS of green sea turtle and the Middle Columbia River steelhead trout.

The 2016 PGP applies protective measures throughout the permit for discharges that may expose "NMFS Listed Resources of Concern as defined in Appendix A of the permit." However, Appendix A of the draft 2016 PGP identifies NMFS Listed Resources of Concern as:

> *"...federally-listed endangered and threatened species and federally-listed critical habitat for which NMFS' 2011 Endangered Species Act Section 7 Consultation Biological Opinion on the U.S. Environmental Protection Agency's Proposed Pesticides General Permit concluded the draft 2011 PGP, absent any additional mitigating measures, would either jeopardize the continued existence of such species or destroy or adversely modify such critical habitat. The Biological Opinion included a Reasonable and Prudent Alternative, implemented through this permit, to avoid likely jeopardy to listed species or adverse modification of critical habitat. Additional information, including maps noting where these resources overlap with PGP areas of coverage is available at www.epa.gov/npdes/pesticides. "*

RPA: In order for the 2016 PGP to provide protection of species listed and critical habitat designated by NMFS since the 2011 PGP, the definition of NMFS Listed Resources of Concern must be amended. In addition, the web address provided with the definition is currently inactive as EPA is updating its websites. The RPA would have EPA change the definition of NMFS Listed Resources of Concern in Appendix A to read:

> *"...federally-listed endangered and threatened species and federally-listed designated critical habitat for which NMFS' 2016 Endangered Species Act Section 7 Consultation Biological Opinion on the U.S. Environmental Protection Agency's Proposed Pesticides General Permit concluded the **2016 PGP**, absent any additional mitigating measures, would either jeopardize the continued existence of such species or destroy or adversely modify such critical habitat. The Biological Opinion included a Reasonable and Prudent Alternative, implemented through this permit, to avoid likely jeopardy to listed species or adverse modification of critical habitat. Additional information, including maps noting where these resources overlap with PGP areas of coverage is available at **[insert a functioning website address that will remain on throughout the permit term]**."*

12.1.2 2016 PGP RPA Element Two

Rationale: EPA must improve the tools available for the 2016 PGP applicants to identify the presence of ESA-listed species under NMFS's jurisdiction to ensure that 2016 PGP applicants are able to easily and accurately identify the presence of NMFS Listed Resources of Concern in their pesticide application area. EPA also needs to make it clear in the NOI form the type of information needed for self-certification of no adverse effects to NMFS Listed Resources of Concern to ensure NMFS receives the correct information to be able to review NOIs. Based on input from our regions and apparent gaps in the locations for which NOI were submitted, NMFS is not confident that PGP applicants accurately identified the presence of ESA-listed species under NMFS' jurisdiction and thus their requirement to submit an NOI.

RPA: As in the 2011 PGP, the 2016 PGP will require applicants to identify whether they must submit an NOI due to discharges made to waters of the U.S. where ESA-listed species under NMFS' jurisdiction occur. The 2016 PGP needs to make it clear in the NOI form the type of information needed for self-certification of no adverse effects to NMFS Listed Resources of Concern to ensure NMFS receives the correct information to review NOIs. The 2016 PGP will provide improved tools and clarifications in the ESA procedures for applicants:

a) EPA, with assistance and data from NMFS, has developed a user friendly webmap for permit applicants to determine whether they are required to submit an NOI due to overlap with NMFS Listed Resources of Concern. EPA will provide a clear link to this webmap on the main web page for the 2016 PGP and will be available for use upon issuance of the 2016 PGP.

b) The 2016 PGP NOI instructions for the required rationale on the NOI form (item 2.g. under section D) must include the following clarifying language (in bold face):

> Your rationale supporting your determination that you meet the criterion for which you are submitting this NOI, **for example, the *specific* BMPs applied, visual monitoring, equipment and/or site inspections, and other** appropriate measures that will be undertaken to avoid or eliminate the likelihood of adverse effects. For certifications pursuant to Criterion D, indicate whether the discharge is likely to

adversely affect NMFS Listed Resources of Concern in response to a pest emergency and, if so, any feasible measures to avoid or eliminate such adverse effects; for example, it is not sufficient to state that "integrated pest management procedures will be applied" without describing the specific measures to be taken (attach additional pages as necessary):

12.2 Incidental Take Statement

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this ITS.

The measures to avoid or minimize take described below are non-discretionary and must be undertaken by the EPA so that they become a binding condition of any applicant, as appropriate, for the exemption in section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this incidental take statement. The protective coverage of section 7(o)(2) may lapse if the EPA: (1) Fails to assume and implement the terms and conditions; or (2) Fails to require any applicant to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the general permit. In order to monitor the impact of incidental take, the EPA must report the progress of the action and its impact on the species to NMFS OPR as specified in the incidental take statement (50 CFR§402.14(i)(3)). The reporting requirements will be established in accordance with 50 CFR220.45 and 228.5.

12.2.1 Amount of Take

ESA Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 CFR § 402.14 (i)(1)(i).). When, as here, the precise location and number of events resulting in incidental take is unknown, NMFS may identify a surrogate rather than an amount or level of incidental take. A "surrogate (e.g., similarly affected species or habitat or ecological conditions) may be used to express the amount or extent of anticipated take provided that the biological opinion or ITS: The surrogate describes the causal link between the surrogate and take of the listed species, explains why it is not practical to express the amount or extent of anticipated take or to monitor take-related impacts in terms of individuals of the listed species, and sets a clear standard for determining when the level of anticipated take has been exceeded." (50 C.F.R. § 402.14).

The proposed action in the 2016 PGP is anticipated to cause incidental of ESA-listed species under NMFS' jurisdiction in the action area. Incidental take due to this action cannot be accurately quantified or monitored as a number of individuals because the action area includes large areas over which EPA has permitting authority and the exact location, composition, time, and frequency of the individual discharges that will be authorized under the 2016 PGP are unknown. We are therefore not able to quantify how many individuals of each species and life stage exist in affected waters, especially considering that the numbers of individuals vary with the season, environmental conditions, and changes in population size due to recruitment and mortality over the course of a year. In addition, currently we have no means to determine which deaths or injuries in populations across the entire range of the ESA-listed species and designated critical habitat covered in this opinion are due to the discharges authorized under the PGP versus other environmental stressors, competition, and predation.

Because we cannot determine the amount of take, NMFS identifies, as a surrogate for the allowable extent of take, the ability of this action to proceed without any adverse incident, defined below, to non-target species, that is attributed to any pesticide discharged in accordance with the general permit in waters where ESA-listed species under NMFS' jurisdiction occur. An adverse incident to fish is considered attributable to a pesticide discharged in accordance with the general permit if that pesticide is known to have been discharged prior to, and near or upstream of the adverse incident and there is evidence that the pesticide caused the adverse incident (e.g. the detection of pesticide, adjuvants, surfactants, or degradates in water samples from the area or in tissue samples of affected fish). An adverse incident means an unusual or unexpected incident that an Operator has observed upon inspection or of which the Operator otherwise become aware, in which**:**

- There is evidence that a person or non-target organism has likely been exposed to a pesticide, <u>and</u>
- The person or non-target organism suffered a *toxic or adverse effect.*

The phrase *toxic or adverse effects* includes effects that occur within waters of the United States on non-target plants, fish or wildlife that are unusual or unexpected (e.g., effects are to organisms not otherwise described on the pesticide product label or otherwise not expected to be present) as a result of exposure to a pesticide and may include:

- Distressed or dead juvenile and small non-target aquatic organisms
- Washed up or floating non-target aquatic organisms
- Non-target aquatic organisms swimming abnormally or erratically
- Non-target aquatic organisms lying lethargically at water surface or in shallow water
- Non-target aquatic organisms that are listless or nonresponsive to disturbance
- Stunting, wilting, or desiccation of non-target submerged or emergent aquatic plants
- Other dead or visibly distressed non-target aquatic organisms (amphibians, turtles, invertebrates, etc.)

The phrase, *toxic or adverse effects*, also includes any adverse effects to humans (e.g., skin rashes) or domesticated animals that occur either from direct contact with or as a secondary effect from a discharge (e.g., sickness from consumption of plants or animals containing pesticides) to Waters of the United States that are temporally and spatially related to exposure to a pesticide (e.g., vomiting, lethargy).

The association of take with adverse pesticide incidents in waters where ESA-listed species and designated critical habitat occur relates to the expectation that individuals of ESA-listed species would be similarly affected during such incidents and take of the ESA-listed individuals may not be detected due to co-occurring events such as scavenging, decay, or submergence. Further, the occurrence of a single incident would indicate an unknown number of future incidents will likely occur. Any incident where non-target organisms appear injured or killed as a result of PGP-authorized discharges to waters of the United States containing NMFS listed species will be considered an exceedance of take.

12.2.2 Reasonable and Prudent Measures

To satisfy its obligations pursuant to section 7(a)(2) of the ESA, the EPA must: (1) Monitor the direct, indirect, and cumulative impacts of the activities authorized by the issuance of the general permit; and (2) Evaluate the direct, indirect, or aggregate impacts of the activities authorized by the issuance of the general permit and the consequences of those effects on ESA-listed species under NMFS' jurisdiction. The purpose of the monitoring is to provide data for the EPA to use to identify necessary modifications to the general permit in order to reduce exposures to ESA-listed species under NMFS' jurisdiction. NMFS believes all measures described as part of the proposed action, together with use of the Reasonable and Prudent Measures and Terms and Conditions described below, are necessary and appropriate to minimize the likelihood of incidental take of ESA-listed species due to implementation of the proposed action.

The EPA shall:

Monitor any incidental take or surrogate measure of take that occurs from the action;

Ensure that permit applicants discharging to waters where ESA-listed species under NMFS' jurisdiction occur are aware of the ESA requirements; and

Report annually to NMFS OPR on the monitoring results from the previous year.

12.2.3 Terms and Conditions

1) To be exempt from the prohibitions of section 9 of the ESA, the EPA must comply with the following condition. This condition implements the reasonable and prudent measures described above. These conditions are non-discretionary.

   The EPA shall include the following instructions requiring reporting of adverse incidents to fish in the general permit:

   "Notwithstanding any of the other adverse incident notification requirements of this section, if an Operator becomes aware of an adverse incident affecting a federally listed threatened or endangered species or its federally designated critical habitat, which may have resulted from a discharge from the Operator's pesticide application, Operator must immediately notify NMFS in the case of an anadromous or marine species, or FWS in the case of a terrestrial or freshwater species. This notification must be made by telephone, to the contacts listed on EPA's website at https://www.epa.gov/npdes/pesticide-permitting, immediately upon the Operator becoming aware of the adverse incident, and must include at least the following information:

      a. The caller's name and telephone number;
      b. Operator name and mailing address;
      c. The name of the affected species;
      d. How and when the Operator became aware of the adverse incident;

e. Description of the location of the adverse incident;

f. Description of the adverse incident and the pesticide product, including the EPA pesticide registration number, for each product applied in the area of the adverse incident; and

g. Description of any steps the Operator has taken or will take to alleviate the adverse impact to the species

Additional information on federally-listed threatened or endangered species and federally-designated critical habitat is available from NMFS (www.nmfs.noaa.gov) for anadromous or marine species or FWS (www.fws.gov) for terrestrial or freshwater species. Note: In an adverse incident affecting federally listed threatened or endangered species or designated critical habitat, the Operator should leave the affected organisms alone, make note of any circumstances likely causing the death or injury, note the location and number or extent of aquatic organisms involved and, if possible, take photographs. In some circumstances, the Operator may be asked to carry out instructions provided by the Services to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved."

2) EPA will develop an outreach strategy specifically targeted towards awareness of ESA species under the 2016 PGP. The outreach strategy will be developed and implemented in coordination with NMFS, and the target audience identified, within 6 months of the implementation date of the 2016 PGP. The need for additional outreach and NMFS review will be revisited as necessary during the annual report reviews described in RPA Element 3.

3) Under the RPA in the 2011 opinion, EPA provided NMFS with summaries of the current registered application rates, the expected environmental concentrations (EECs) of pesticides in water resulting from those applications, and the toxicity information used to assess the risk to endangered and threatened species as presented in the EPA's most recent FIFRA risk assessment documents for all pesticides identified by PGP applicants that apply pesticides to areas with NMFS Listed Resources of Concern under Part 1.1.2.4, criteria D and F of the 2011 PGP. EPA also provided to NMFS the original risk assessment documents from which these summaries were derived for those pesticides under the 2011 PGP. This information was helpful for NMFS to provide guidance to the 2011 PGP applicants on how to prevent or minimize adverse exposures to ESA-listed species and designated critical habitat.

   EPA will continue to provide NMFS with its most recent FIFRA risk assessment documents containing the current registered application rates, the expected environmental concentrations of pesticides in water resulting from those applications, and the toxicity information used to assess the risk to endangered and threatened species for all pesticides identified by PGP applicants that apply pesticides to areas with NMFS Listed Resources of Concern under Part 1.1.2.4, criteria D and F, of the 2016 PGP. This information will be provided as part of the annual reports.

4) To insure implementation of the 2016 PGP, EPA must monitor and evaluate the information obtained through its NOI and annual reports. In the NOI, the operator must identify where and when such discharges would occur, what those discharges would be

and of which use patterns these discharges would consist. NMFS will have the opportunity to review every discharge that might result in exposure to endangered and threatened species or designated critical habitat under NMFS jurisdiction. NMFS will then determine whether the planned discharge or discharge(s) (future discharge or discharges in the case of Declared Pest Emergency Situations) meets the general permit's eligibility criteria of not likely to adversely affect NMFS Listed Resources of Concern, would meet it with additional conditions or would not meet the eligibility criteria. The NOI process is designed to ensure that no individual discharge or combination of discharges is likely to adversely affect listed species or designated critical habitat, with the limited exception of discharges in response to a Declared Pest Emergency Situation, described below. While the general permit does authorize discharges to address Declared Pest Emergency Situations prior to review of discharges by NMFS, this authorization has significant limits. The PGP specifies that a Declared Pest Emergency Situation is an event defined by public declaration by a federal agency, state, or local government, beginning less than ten days after identification of a pest problem posing significant risk to human health and the environment or significant economic loss. Once NMFS has reviewed a past or ongoing discharge pursuant to the NOI process for declared pest emergencies and provided its determination to EPA on whether the discharge(s) meet or could have met the eligibility criteria, any conditions or prohibitions applied by EPA remain in effect for the life of the permit for that discharger. This element of the RPA is designed to prevent repeated declarations of pest emergencies by the same operator, with a recurring 60 day of discharge authorization under the general permit without any conditions or prohibitions in place.

EPA will meet with NMFS within 6 months of the issuance of the 2016 PGP to develop a strategy for analyzing and summarizing the annual reports that will be submitted by PGP discharges. EPA will use this strategy to develop a summary report and continue to provide the report, and its source information, to NMFS for each year of the permit term until the permit expires in 2021. The strategy will include measures to ensure continuity in the process in the event of staffing changes. The 1st report will come April 15, 2018. EPA will meet with NMFS within 3 weeks of the receipt of the report by NMFS to review the information in the annual report.

13 REINITIATION NOTICE

This concludes formal consultation on the U.S. Environmental Protection Agency's issuance of the Pesticides General Permit. As provided in 50 CFR 402.16, Reinitiation of formal consultation is required and shall be requested by the Federal agency or by the Service, where discretionary Federal involvement or control over the action has been retained or is authorized by law and:

> (a) If the amount or extent of taking specified in the incidental take statement is exceeded;
>
> (b) If new information reveals effects of the action that may affect ESA-listed species or designated critical habitat in a manner or to an extent not previously considered;
>
> (c) If the identified action is subsequently modified in a manner that causes an effect to the ESA-listed species or designated critical habitat under NMFS' jurisdiction that was not considered in the biological opinion;
>
> (d) If a new species is listed or critical habitat is designated that may be affected by the identified action in a way not considered in this opinion;

A determination that Decision-makers who should file NOIs for discharges to waters of the United States containing ESA-listed species under NMFS' jurisdiction have failed to do so or that Decision-makers incorrectly identify Criterion A or F as applicable to their proposed discharges shall constitute new information reveals effects of the action that may affect ESA-listed species or designated critical habitat in a manner or to an extent not previously considered and require reinitiation pursuant to (b), above.

For those facilities with endangered species protection certifications in the NOI based on an existing formal consultation, any instance where the amount or extent of take specified in the incidental take statement is exceeded requires that the U.S. Environmental Protection Agency must immediately request reinitiation of Section 7 consultation.

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APPENDIX A

COMPREHENSIVE STATUS OF THE SPECIES AND DESIGNATED CRITICAL HABITAT IN THE ACTION AREA

The ESA-listed species and designated critical habitats which occur within the action area that fall under NMFS' jurisdiction and may be exposed to the pesticide discharges and experience direct or indirect effects of those exposures are identified in Table 1 and Table 2.

Table 1. NMFS endangered and threatened species and designated critical habitat considered in this opinion.

Species	ESA Status	Designated Critical Habitat	Recovery Plan
Marine Mammals – Cetaceans			
Southern Resident Killer Whale *(Orcinus orca)*	E – 70 FR 69903	71 FR 69054	73 FR 4176
Salmonids			
salmon, Chinook (*Oncorhynchus tshawytscha*)			
- California coastal	T – 64 FR 50393	70 FR 52488	– –
- Central Valley spring-run	T – 64 FR 50393	70 FR 52488	79 FR 42504
- Lower Columbia River	T – 64 FR 14308	70 FR 52630	78 FR 41911
- Upper Columbia River spring-run	E – 64 FR 14308	70 FR 52630	72 FR 57303
- Puget Sound	T – 64 FR 14308	70 FR 52630	72 FR 2493
- Sacramento River winter-run	E – 59 FR 440	58 FR 33212	79 FR 42504
- Snake River fall-run	T – 59 FR 42529	58 FR 68543	– –
- Snake River spring/summer-run	T – 59 FR 42529	64 FR 57399	– –
- Upper Willamette River	T – 64 FR 14308	70 FR 52630	76 FR 52317b
salmon, chum (*Oncorhynchus keta*)			
- Columbia River	T – 64 FR 14507	70 FR 52630	78 FR 41911
- Hood Canal summer-run	T – 64 FR 14507	70 FR 52630	72 FR 29121
salmon, coho (*Oncorhynchus kisutch*)			
- Central California coast	E – 61 FR 56138	65 FR 7764	– –
- Oregon coast	T – 63 FR 42587	73 FR 7816	78 FR 41911
- Southern Oregon & Northern California coasts	T – 62 FR 24588	64 FR 24049	– –
- Lower Columbia River	T – 70 FR 37160	81 FR 9251	78 FR 41911
salmon, sockeye (*Oncorhynchus nerka*)			
- Ozette Lake	T – 64 FR 14528	70 FR 52630	74 FR 24706
- Snake River	E – 56 FR 58619	58 FR 68543	– –
trout, steelhead (*Oncorhynchus mykiss*)			
- California Central Valley	T – 71 FR 834	70 FR 52488	79 FR 42504
- Central California coast	T – 71 FR 834	70 FR 52488	– –
- South-Central California coast	T – 71 FR 834	70 FR 52488	– –
- Southern California	E – 71 FR 834	70 FR 52488	– –
- Northern California	T – 71 FR 834	70 FR 52488	– –

Species	ESA Status	Designated Critical Habitat	Recovery Plan
- Lower Columbia River	T – 71 FR 834	70 FR 52630	74 FR 50165
- Middle Columbia River	T – 71 FR 834	70 FR 52630	– –
- Upper Columbia River	T – 74 FR 42605	70 FR 52630	72 FR 57303
- Upper Willamette River	T – 71 FR 834	70 FR 52630	76 FR 52317b
- Snake River Basin	T – 71 FR 834	70 FR 52630	– –
- Puget Sound	T – 72 FR 26722	81 FR 9251	– –
Atlantic Salmon (*Salmo salar*) - Gulf of Maine DPS	E – 74 FR 29344	74 FR 29300	70 R 75473
Non-Salmonid Anadromous Species			
Eulachon (*Thaleichthys pacificus*)	T – 75 FR 13012	76 FR 65323	– –
Shortnose sturgeon (*Acipenser brevirostrum*)	E – 32 FR 4001	– –	63 FR 69613
Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*)			
- Gulf of Maine DPS	T – 77 FR 5880	81 FR 35701 (Proposed)	– –
- New York Bight DPS	E - 77 FR 5880		
- Chesapeake Bay DPS			
Green sturgeon, (*Acipenser medirostris*) - Southern DPS	T – 71 FR 17757	74 FR 52300	– –
Marine Fish			
Bocaccio *(Sebastes paucispinis)*	E – 75 FR 22276	79 FR 68041	– –
Yellow Eye Rockfish (*Sebastes ruberrimus*)	T – 75 FR 22276	79 FR 68041	– –
Nassau Grouper	T – 79 FR 51929		
Sea Turtles			
Green Turtle *(Chelonia mydas) – North Atlantic DPS*	E – 43 FR 32800	63 FR 46693	63 FR 28359
Hawksbill Turtle *(Eretmochelys imbricata*)	E – 35 FR 8491	63 FR 46693	57 FR 38818
Kemp's Ridley Turtle *(Lepidochelys kempii)*	E – 35 FR 18319	– –	75 FR 2496
Olive Ridley Turtle (*Lepidochelys olivacea*)			
Pacific Coast of Mexico breeding populations	E – 43 FR 32800	– –	63 FR 28359
all other populations	T – 43 FR 32800		
Leatherback Turtle (*Dermochelys coriacea*)	E – 35 FR 8491	44 FR 17710	63 FR 28359
Loggerhead Turtle (*Caretta carettaCaretta caretta*)			
- Northwest Atlantic and North Pacific DPS	E – 76 FR 58868	79 FR 39856	63 FR 28359
Corals			
Elkhorn Coral (*Acropora palmata*) Staghorn Coral (*Acropora cervicornis*)	T – 71 FR 26852	73 FR 72210	80 FR 12146

Species	ESA Status	Designated Critical Habitat	Recovery Plan
Coral Species - *Mycetophyllia ferox* - The *Orbicella:* *O.faveolata* O. franksi *O. annularis* - Pillar *(Dendrogyra cylindrus)* - *The Acropora* *A. globiceps* *A. jacquelineae* *A. lokani* *A. pharaonis* *A. retusa* *A. rudis* *A. speciosa* *A. tenella* - *Anacropora spinosa* - *Euphyllia paradivisa* - *Isopora crateriformis* - *Montipora australiensis* - *Pavona diffluens* - *Porites napopora* - *Seriatopora aculeata*	T – 79 FR 54122	– –	– –

The following sections describe the status of species that occur in the action area and the threats to those species and where applicable, their designated critical habitat.

1 *CETACEANS*

1.1 Southern Resident Killer Whale

Status. The Southern Resident killer whale DPS was listed as endangered in 2005 in response to the population decline from 1996 to 2001, small population size, and reproductive limitations (i.e., few reproductive males and delayed calving). This species occurs in the inland waterways of Puget Sound, Strait of Juan de Fuca, and Southern Georgia Strait during the spring, summer and fall. During the winter, they move to coastal waters primarily off Oregon, Washington, California, and British Columbia. We used information available in the final rule, the 2012 Status Review (NMFS 2013) and the 2011 Stock Assessment Report (NMFS 2014) to summarize the status of this species.

The most recent abundance estimate for the Southern Resident DPS is 87 whales in 2012. This represents an average increase of 0.4 percent annually since 1982 when there were 78 whales. Population abundance has fluctuated during this time with a maximum of approximately 100 whales in 1995 (NMFS 2013). As compared to stable or growing populations, the DPS reflects a smaller percentage of juveniles and lower fecundity (NMFS 2014) and has demonstrated weak growth in recent decades.

Life history. Southern Resident killer whales are geographically, matrilineally, and behaviorally distinct from other killer whale populations. The DPS includes three large, stable pods (J, K, and L), which occasionally interact (Parsons et al. 2009). Most mating occurs outside natal pods, during temporary associations of pods, or as a result of the temporary dispersal of males (Pilot et al. 2010). Males become sexually mature at 10 – 17 years of age. Females reach maturity at 12 – 16 years of age and produce an average of 5.4 surviving calves during a reproductive life span of approximately 25 years. Mothers and offspring maintain highly stable, life-long social bonds, and this natal relationship is the basis for a matrilineal social structure. They prey upon salmonids, especially Chinook salmon (Hanson et al. 2010).

Threats. Current threats to its survival and recovery include: contaminants, vessel traffic, and reduction in prey availability. Chinook salmon populations have declined due to degradation of habitat, hydrology issues, harvest, and hatchery introgression; such reductions may require an increase in foraging effort. In addition, these prey contain environmental pollutants (e.g., flame retardants; PCBs and DDT). These contaminants become concentrated at higher trophic levels and may lead to immune suppression or reproductive impairment (70 FR 69903).

The inland waters of Washington and British Columbia support a large whale watch industry, commercial shipping, and recreational boating; these activities generate underwater noise, which may mask whales' communication or interrupt foraging. The factors that originally endangered the species persist throughout its habitat: contaminants, vessel traffic, and reduced prey. The DPS's resilience to future perturbation is reduced as a result of its small population size (N = 86); however, it has demonstrated the ability to recover from smaller population sizes in the past and has shown an increasing trend over the last several years. NMFS is currently conducting a status review prompted by a petition to delist the DPS based on new information, which indicates that there may be more paternal gene flow among populations than originally detected (Pilot et al. 2010).

Designated critical habitat. The designated critical habitat consists of approximately 6,630 km^2 in three areas: the Summer Core Area in Haro Strait and waters around the San Juan Islands; Puget Sound; and the Strait of Juan de Fuca. It provides the following physical and biological features: water quality to support growth and development; prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth; and inter-area passage conditions to allow for migration, resting, and foraging.

2 SALMONIDS

Salmonids have similar life histories, habitat requirements, and threats. These are discussed in the sections below, before proceeding to describing the essential features of critical habitat for each species.

2.1 The 2016 Status Review for Pacific Salmonids

In May 2016, NOAA Fisheries' West Coast Region completed a five-year status review of all 28 West Coast salmon and steelhead species listed under the ESA (Table 3). Some species, such Oregon Coast coho salmon, mid-Columbia steelhead and Hood Canal chum, rebounded from the lows of past decades. Highly endangered Snake River sockeye have benefitted from a captive broodstock program while Snake River steelhead populations are steady. The California drought and unusually high ocean and stream temperatures over the 5-year period hit many populations hard. In the case of Sacramento River winter-run Chinook salmon, for example, drought conditions and high stream temperatures reduced the 2015 survival of juvenile fish in the first stretch of river to just 3 percent.

Since 1997 NMFS promulgated a total of 29 limits to the ESA section 9(a) take prohibitions for 21 threatened Pacific salmon and steelhead ESUs or Distinct Populations Segments (DPSs)(62

FR 38479, July 18, 1997; 65 FR 42422, July 10, 2000; 65 FR 42485, July 10, 2000; 67 FR 1116, January 9, 2002; 73 FR 7816, February 11, 2008). On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160). NMFS took this action to provide appropriate flexibility to ensure that fisheries and artificial propagation programs are managed consistently with the conservation needs of threatened salmon and steelhead. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. Throughout this section discussing listed salmonids, we use the word “species” to apply to DPSs and ESUs.

Table 2. Summary of Current ESA Listing Status, Recent Trends and Summary of Conclusions for the Most Recent Five-year Review for Pacific Salmonids (Northwest Fisheries Science Center 2015, Williams et al. 2016).

Species	ESU/DPS	ESA listing status	Recent risk trend
Chinook	Upper Columbia spring	Endangered	Stable
	Snake River spring/summer	Threatened	Stable
	Snake River fall	Threatened	Improving
	Upper Willamette spring	Threatened	Declining
	Lower Columbia	Threatened	Stable/Improving
	Puget Sound	Threatened	Stable/Declining
	California Coastal	Threatened	Mixed
	Central Valley Spring	Threatened	Decreased risk of extinction
	Sacramento River winter	Endangered	Increased risk of extinction
Coho	Lower Columbia	Threatened	Stable/Improving
	Oregon Coast	Threatened	Improving
	Southern Oregon/Northern California	Threatened	Mixed
	Central California Coast	Endangered	Mixed
Sockeye	Snake River	Endangered	Improving
	Lake Ozette	Threatened	Stable
Chum	Hood Canal summer	Threatened	Improving
	Columbia River	Threatened	Stable
Steelhead	Upper Columbia	Threatened	Improving
	Snake River	Threatened	Stable/Improving
	Middle Columbia	Threatened	Stable/Improving
	Upper Willamette	Threatened	Declining
	Lower Columbia	Threatened	Stable
	Puget Sound	Threatened	Stable
	Northern California	Threatened	Mixed
	Central California Coast	Threatened	Uncertain
	South Central California	Threatened	Declining
	Southern California	Endangered	Uncertain

The most recent status review for Atlantic salmon was published in 2006 (Fay et al. 2006). This review stated that fewer than 1,500 adults have returned to spawn each year since 1998. The Population Viability Analysis estimates of the probability of extinction for the Gulf of Mexico DPS of Atlantic Salmon ranges from 19 percent to 75 percent within the next 100 years, even with the continuation of current levels of hatchery supplementation. The abundance was estimated at 1,014 individuals in 2007, the most recent year for which abundance records are available.

2.2 Salmonid Life Histories

Salmonids exhibit either an ocean-type or stream-type behavior. Ocean-type migrate to the ocean within their first year of life (sub-yearlings). Stream-type salmonids usually migrate to sea at a larger size, after months or years of freshwater rearing. Stream-type salmonids of the genus *Oncorhynchus* include steelhead, coho, and most types of Chinook and sockeye salmon. Stream type salmonids depend more on freshwater conditions than on favorable estuarine conditions. All Pacific salmon species are semelparous (i.e., they die after spawning) and exhibit obligatory anadromy (i.e., there are no recorded landlocked or naturalized freshwater populations; they must spend portions of their lives in both salt and freshwater habitats). Atlantic salmon and some southern populations of steelhead are iteroparous, being capable of returning to the ocean after spawning and returning to freshwaters to spawn again after recovery.

2.3 Threats to Salmonids

Specifically, during all freshwater life stages, salmonids require cool water that is free of contaminants. Water free of contaminants supports survival, growth, and maturation of salmon and the abundance of their prey. In addition to affecting survival, growth, and fecundity, contaminants can disrupt normal behavior necessary for successful migration, spawning, and juvenile rearing. Sufficient forage is necessary for juveniles to maintain growth that reduces freshwater predation mortality, increases overwintering success, initiates smoltification, and increases ocean survival. Natural riparian cover such as submerged and overhanging large wood and aquatic vegetation provides shelter from predators, shades freshwater to prevent increase in water temperature, provides nutrients from leaf litter, supports production of insect prey, and creates important side channels. Riparian vegetation stabilizes bank soils and captures fine sediment in runoff, which maintains functional channel bottom substrate for development of eggs and alevins.

The process of smoltification enables salmon to adapt to the ocean environment. Environmental factors such as exposure to chemicals including heavy metals and elevated water temperatures can affect the smoltification process, not only at the interface between fresh water and saltwater, but higher in the watershed as the process of transformation begins long before fish enter saltwater (Wedemeyer et al. 1980).

The three major threats to Atlantic salmon identified in the listing rule also threaten Pacific salmonids: dams, regulatory mechanisms related to dams, and low marine survival. In addition, a number of secondary threats were identified, including threats to habitat quality and accessibility, commercial and recreational fisheries, disease and predation, inadequacy of regulatory mechanisms related to water withdrawal and water quality, aquaculture, artificial propagation, climate change, competition, and depleted fish communities.

2.4 Salmonid Designated Critical Habitat

The action area for this consultation contains designated critical habitat for anadromous salmonids. NMFS has identified essential features of designated critical habitat for each life stage (e.g., migration, spawning, rearing, and estuary) common for each species. To fully understand the conservation role of these habitats, specific physical and biological habitat features (e.g., water temperature, water quality, forage, natural cover, etc.) were identified for each life stage.

2.4.1 Chinook salmon (9 ESUs)

Life history. There are 9 ESA-listed Chinook salmon ESUs. Chinook are the largest of the Pacific salmon and prefer streams that are deeper and larger than those used by other Pacific salmon species. Chinook salmon ESUs exhibit either “stream-type” or “ocean-type” life histories. Stream-type Chinook salmon reside in freshwater for a year or more following emergence before migrating to salt water. Stream-type ESUs normally return in late winter and early spring (spring-run) as immature adults and reside in deep pools during summer before spawning in fall. Ocean-type Chinook salmon migrate to the ocean within their first year and usually return as full mature adults in fall (fall-run) and spawn soon after river entry. (Healey 1991).

Temperature and stream flow can significantly influence the timing of migrations and spawning, as well as the selection of spawning habitat (Geist et al. 2008, Hatten et al. 2009). All Chinook salmon are semelparous (i.e. they die after spawning). Fall-run Chinook salmon generally spawn in the mainstem of larger rivers and are less dependent on flow, although early autumn rains and a drop in water temperature often provide cues for movements to spawning areas. Spring-run Chinook salmon take advantage of high flows from snowmelt to access the upper reaches of rivers. Chinook salmon primarily feed on small invertebrates and vertebrates, with the diet of adult oceanic Chinook salmon comprised primarily of fish.

Designated critical habitat. Designated critical habitat for the Puget Sound, Lower Columbia River, and Upper Willamette River ESUs for Chinook salmon identify essential features and sites necessary to support one or more Chinook salmon life stage(s). These include biological elements that are vulnerable to the stressors of the action. These include water quality conditions that support spawning and incubation, larval and juvenile development, and physiological transitions between fresh and saltwater. The essential features also include aquatic invertebrate and fish prey species and water quality to support juvenile and adult development, growth, and maturation, and natural cover of riparian and nearshore vegetation and aquatic vegetation. Designated critical habitat for the Snake River fall-run and Snake River spring/summer run Chinook salmon generically designates water quality, food, and riparian vegetation essential features.

2.4.2 Chum salmon (2 ESUs)

Life history. In general, North American chum salmon migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. Chum salmon usually spawn in the lower reaches of rivers during summer and fall. Redds are dug in the mainstem or in side channels of rivers from just above tidal influence to nearly 100 km from the sea. Juveniles use shallow, low flow habitats for rearing that include inundated mudflats, tidal wetlands and their channels, and sloughs. The duration of estuarine residence for chum salmon juveniles are known for only a few estuaries. Observed residence time ranges from 4 to 32 days, with about 24 days as the most common.

Immature chum salmon disperse over the North Pacific Ocean and maturing adults return to the home streams usually at two to five years of age, and in some cases up to seven years (Bigler 1985). This ocean-type life history means that the survival and growth for juvenile chum salmon depends less on freshwater conditions than on favorable estuarine conditions. Chum salmon feed on a variety of prey organisms depending upon life stage and size. In freshwater Chum salmon feed primarily on small invertebrates; in saltwater, their diet consists of copepods, tunicates, mollusks, and fish.

Designated critical habitat. Areas designated as critical habitat are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. essential features for both chum salmon ESUs include freshwater spawning, rearing, and migration areas; estuarine and nearshore marine areas free of obstructions; and offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

2.4.3 Coho salmon (4 ESUs)

Life history. North American coho salmon will migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. During this migration, juvenile coho salmon tend to occur in both coastal and offshore waters. Coho salmon exhibit a stream-type life history. Most coho salmon enter rivers between September and February. In many systems, coho salmon wait to enter until fall rainstorms have provided the river with sufficiently strong flows and depth. Coho salmon spawn from November to January, and occasionally into February and March. Some spawning occurs in third-order streams, but most spawning activity occurs in fourth- and fifth-order streams with gradients of 3 percent or less. After fry emerge in spring they disperse upstream and downstream to establish and defend territories with weak water currents such as backwaters and shallow areas near stream banks. Juveniles rear in these areas during the spring and summer. In early fall juveniles move to river margins, backwater, and pools. During winter juveniles typically reduce feeding activity and growth rates slow down or stop. By March of their second spring, juveniles feed heavily on insects and crustaceans and grow rapidly before smoltification and outmigration (Olegario 2006), spending only a short time (one to three days) in the estuary with little feeding (Thorpe 1994, Miller and Sadro 2003). After entering the ocean, immature coho salmon initially remain in nearshore waters close to the parent stream. Along the Oregon/California coast, coho salmon primarily return to rivers to spawn as three-year olds, having spent approximately 18 months rearing in fresh water and 18 months in salt water. In some streams, a smaller proportion of males may return as two-year olds. The presence of two-year old males can allow for substantial genetic exchange between brood years. The relatively fixed three-year life cycle exhibited by female coho salmon limits demographic interactions between brood years. This makes coho salmon more vulnerable to environmental perturbations than salmonids that exhibit overlapping generations, i.e., the loss of a coho salmon brood year in a stream is less likely to be reestablished by females from other brood years than for other Pacific salmon.

Coho salmon feed on a variety of prey organisms depending upon life stage and size. While at sea, coho salmon tend to eat fish including herring, sand lance, sticklebacks, sardines, shrimp and surf smelt. While in estuaries and in fresh water coho salmon are significant predators of Chinook, pink, and chum salmon, as well as aquatic and terrestrial insects. Smaller fish, such as fry, eat chironomids, plecoptera and other larval insects, and typically use visual cues to find their prey.

Designated critical habitat. The essential features of designated critical habitat for the Central California Coast and Southern Oregon/Northern California Coast coho salmon ESUs that are vulnerable to the stressors of the action are generically identified as water quality, food, and riparian vegetation. The essential features of designated critical habitat for the Lower Columbia

River and Oregon Coast ESUs are more detailed. They include water quality conditions supporting spawning, incubation and larval development, water quality and forage supporting juvenile development; and natural cover of riparian and aquatic vegetation, water quality conditions supporting juvenile and adult physiological transitions between fresh- and saltwater, and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.

2.4.4 Sockeye salmon (2 ESUs)

Life history. Most sockeye salmon exhibit a lake-type life history (i.e., they spawn and rear in or near lakes), though some exhibit a river-type life history. Spawning generally occurs in late summer and fall, but timing can vary greatly among populations. In lakes, salmon commonly spawn along “beaches” where underground seepage provides fresh oxygenated water. Incubation is a function of water temperature, but generally lasts between 100 to 200 days (Burgner 1991). Sockeye salmon fry primarily rear in lakes; river-emerged and stream-emerged fry migrate into lakes to rear. Juvenile sockeye salmon generally rear in lakes from one to three years after emergence, though some river-spawned salmon may migrate to sea in their first year. Juvenile sockeye salmon feeding behaviors change as they transition through life stages after emergence to the time of smoltification. In the early fry stage, from spring to early summer, juveniles forage exclusively in the warmer littoral (i.e., shoreline) zone where they depend mostly on fly larvae and pupae, copepods, and water fleas. In summer, underyearling sockeye salmon move from the littoral habitat to a pelagic (i.e., open water) existence where they feed on larger zooplankton; however, flies may still make up a substantial portion of their diet. Older and larger fish may also prey on fish larvae. Distribution in lakes and prey preference is a dynamic process that changes daily and yearly depending on many factors, including: water temperature; prey abundance; presence of predators and competitors; and size of the juvenile. Peak emigration to the ocean occurs in mid-April to early May in southern sockeye populations (<52ºN latitude) and as late as early July in northern populations (62ºN latitude) (Burgner 1991). Adult sockeye salmon return to their natal lakes to spawn after spending one to four years at sea. The diet of adult salmon consists of amphipods, copepods, squid, and other fish.

Designated Critical Habitat. The essential features of designated critical habitat for Lake Ozette sockeye ESU that are potentially affected by the stressors of the action include water quality conditions and forage species supporting spawning, incubation, development, growth, maturation, physiological transitions between fresh and saltwater, and natural cover of riparian and nearshore vegetation and aquatic vegetation. The essential features of designated critical habitat for Snake River sockeye potentially affected by the stressors of the action are identified generically as water quality, food, and riparian vegetation.

2.4.5 Steelhead trout (11 DPSs)

Life history. Steelhead have a longer run time than other Pacific salmonids and do not tend to travel in large schools. They can be divided into two basic run-types: the stream-maturing type (summer steelhead) and the ocean-maturing type (winter steelhead). Summer steelhead enter fresh water as sexually immature adults between May and October (Nickelson et al. 1992, Busby et al. 1996) and hold in cool, deep pools during summer and fall before moving to spawning sites as mature adults in January and February (Barnhart 1986, Nickelson et al. 1992). Winter steelhead return to fresh water between November and April as sexually mature adults and spawn shortly after river entry (Nickelson et al. 1992, Busby et al. 1996). Steelhead typically spawn in small tributaries rather than large, mainstem rivers and spawning distribution often overlaps with coho salmon, though steelhead tend to prefer higher gradients (generally two to seven percent, but up to 12 percent or more) and their distributions tend to extend further upstream than coho salmon. Summer steelhead commonly spawn higher in a watershed than do winter steelhead, sometimes even using ephemeral streams from which juveniles are forced to emigrate as flows diminish. Fry usually inhabit shallow water along banks and stream margins of streams (Nickelson et al. 1992) and move to faster flowing water such as riffles as they grow. Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelson et al. 1992). In Oregon and California, steelhead may enter estuaries where sand bars create low salinity lagoons. Migration of juvenile steelhead to these lagoons occurs throughout the year, but is concentrated in the late spring/early summer and in the late fall/early winter periods (Shapovalov and Taft 1954, Zedonis 1992). Juveniles rear in fresh water for one to four years, then smolt and migrate to the ocean in March and April (Barnhart 1986). Steelhead typically reside in marine waters for two or three years prior to returning to their natal streams to spawn as four or five-year olds. Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby et al. 1996). Females spawn more than once more commonly than males, but rarely more than twice before dying (Nickelson et al. 1992). Iteroparity is also more common among southern steelhead populations than northern populations (Busby et al. 1996).

Steelhead feed on a variety of prey organisms depending upon life stage, season, and prey availability. In freshwater juveniles feed on common aquatic stream insects such as caddisflies, mayflies, and stoneflies but also other insects (especially chironomid pupae), zooplankton, and benthic organisms (Pert 1993 , Merz 2002). Older juveniles sometimes prey on emerging fry, other fish larvae, crayfish, and even small mammals, though these are not a major food source (Merz 2002). The diet of adult oceanic steelhead is comprised primarily of fish and squid (Light 1985, Burgner et al. 1992).

Designated critical habitat. The essential features of designated critical habitat for all steelhead DPSs that are potentially affected by the stressors of the action include water quality conditions and/or forage species supporting spawning, incubation, development, growth, maturation, physiological transitions between fresh and saltwater, and natural cover of riparian and nearshore vegetation and aquatic vegetation.

2.4.6 Atlantic salmon

Status. The Gulf of Maine DPS of Atlantic salmon was first listed as endangered in response to population decline caused by many factors, including overexploitation, degradation of water quality, and damming of rivers, all of which remain persistent threats. The listing was refined to include all anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River, and wherever these fish occur in the estuarine and marine environment. The USFWS has jurisdiction over this species in freshwater, so the NMFS jurisdiction is limited to potential PGP-authorized discharges from the coastal lands belonging to the Passamoquoddy Tribe at Pleasant Point. We used information available in the 2006 Status Review (Fay et al. 2006) and the Final Rule to List the Expanded Gulf of Maine DPS as Endangered Under the ESA (74 FR 29344) to summarize the status of the species, as follows.

In 2015, NMFS announced a new program to focus and redouble its efforts to protect some of the species that are currently among the most at risk of extinction in the near future with the goal of reversing their declining trend so that the species will become a candidate for recovery in the future. Atlantic salmon is one of the eight species identified for this initiative (NMFS 2015b). These species were identified as among the most at-risk of extinction based on three criteria (1) endangered listing, (2) declining populations, and (3) are considered a recovery priority #1. A priority #1 species is one whose extinction is almost certain in the immediate future because of a rapid population decline or habitat destruction, whose limiting factors and threats are well understood and the needed management actions are known and have a high probability of success, and is a species that is in conflict with construction or other developmental projects or other forms of economic activity (55 FR 24296, June 15, 1990).

Life History. Adult Atlantic salmon in the Gulf of Maine typically spawn in early November and juveniles spend approximately two years feeding on small invertebrates and occasionally small vertebrates in freshwater until they weigh approximately two ounces and are six inches in length. Smoltification (the physiological and behavioral changes required for the transition to salt water) usually occurs at age two for this DPS after which the species migrates more than 4,000 km in the open ocean to reach feeding areas in the Davis Strait between Labrador and Greenland. Adult salmon feed opportunistically and their diet is composed primarily of other fish. The majority (90 percent) spend two winters at sea before reaching maturity and returning to their natal rivers, with the remainder spending one or three winters at sea. At maturity, Gulf of Maine DPS salmon typically weigh between 8 to 15 pounds and average 30 inches in length.

Designated critical habitat. The designated critical habitat includes all anadromous Atlantic salmon streams whose freshwater range occurs in watersheds from the Androscoggin River northward along the Maine coast northeastward to the Dennys River, and wherever these fish occur in the estuarine and marine environment. The essential features identified within freshwater and estuarine habitats of the occupied range of the Gulf of Maine DPS include sites for spawning and incubation, juvenile rearing, and migration. Designated critical habitat and essential features were not designated within marine environments because of the limited of the physical and biological features that the species uses during the marine phase of its life.

3 NON-SALMONID ANADROMOUS FISH

3.1 Southern Pacific eulachon

Status. Eulachon are small smelt native to eastern North Pacific waters from the Bering Sea to Monterey Bay, California, or from 61º N to 31º N (Hart and McHugh 1944, Eschmeyer et al. 1983, Minckley et al. 1986, Hay and McCarter 2000). Eulachon that spawn in rivers south of the Nass River of British Columbia to the Mad River of California comprise the southern population of Pacific eulachon. This species status is classified as “at moderate risk of extinction throughout all of its range” (Gustafson 2010) based upon timing of runs and genetic distinctions (Hart and McHugh 1944, McLean et al. 1999, Hay and McCarter 2000, McLean and Taylor 2001, Beacham et al. 2005). Based on a number of data sources, the 2016 Status Review Update for eulachon reports that the spawning population has increased between 2011 and 2015 and that of the size of some sub-populations is larger than originally estimated in 2010 (Gustafson et al. 2016). The status update does not recommend a change in status because it is too early to tell whether recent improvements in the southern DPS of eulachon will persist. Recent poor ocean conditions taken with given variability inherent in wild populations suggest that population declines may again become widespread in the upcoming return years.

Life Cycle. Adult eulachon are found in coastal and offshore marine habitats (Allen et al. 1988, Hay and McCarter 2000, Willson et al. 2006). Larval and post larval eulachon prey upon phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and other eulachon larvae until they reach adult size (WDFW and ODFW 2001). The primary prey of adult eulachon are copepods and euphausiids, malacos, tracans, and cumaceans (Smith and Saalfeld 1955, Barraclough 1964, Drake and Wilson 1991, Sturdevant et al. 1999, Hay and McCarter 2000).

Although primarily marine, eulachon return to freshwater to spawn. Adult eulachon have been observed in several rivers along the west coast (Odemar 1964, Minckley et al. 1986, Emmett et al. 1991, Jennings 1996, Wright 1999, Hay and McCarter 2000, Larson and Belchik 2000, Musick et al. 2000, WDFW and ODFW 2001, Moyle 2002). For the southern population of Pacific eulachon, most spawning is believed to occur in the Columbia River and its tributaries as well as in other Oregonian and Washingtonian rivers (Emmett et al. 1991, Musick et al. 2000, WDFW and ODFW 2001). Eulachon take less time to mature and generally spawn earlier in southern portions of their range than do eulachon from more northerly rivers (Clarke et al. 2007).

Spawning is strongly influenced by water temperatures, so the timing of spawning depends upon the river system involved (Willson et al. 2006). In the Columbia River and further south, spawning occurs from late January to March, although river entry occurs as early as December (Hay and McCarter 2000). Further north, the peak of eulachon runs in Washington State is from February through March while Alaskan runs occur in May and river entry may extend into June (Hay and McCarter 2000). Females lay eggs over sand, course gravel or detritial substrate. Eggs attach to gravel or sand and incubate for 30 to 40 days after which larvae drift to estuaries and coastal marine waters (Wydoski and Whitney 1979).

Eulachon generally die following spawning (Scott and Crossman 1973). The maximum known lifespan is 9 years of age, but 20 to 30 percent of individuals live to 4 years and most individuals survive to 3 years of age, although spawning has been noted as early as 2 years of age (Wydoski and Whitney 1979, Barrett et al. 1984, Hugg 1996, Hay and McCarter 2000, WDFW and ODFW 2001). The age distribution of spawners varies between river and from year-to-year (Willson et al. 2006).

Threats. The Biological Review Team 2010 assessment of the status of the southern DPS of eulachon ranked climate change impacts on ocean conditions as the most serious threat to the persistence of eulachon in all four subareas of the DPS: Klamath River, Columbia River, Fraser River, and British Columbia coastal rivers south of the Nass River. Climate change impacts on freshwater habitat and eulachon bycatch in offshore shrimp fisheries were also ranked in the top four threats in all subareas of the DPS. Dams and water diversions in the Klamath and Columbia rivers and predation in the Fraser and British Columbia coastal rivers filled out the last of the top four threats (Gustafson 2010).

Designated critical habitat. The designated critical habitat for the southern population of Pacific eulachon includes freshwater creeks and rivers and their associated estuaries, comprising approximately 539 km (335 mi) of habitat. The physical or biological features potentially affected by the stressors of the action include water quality conditions supporting spawning and incubation, larval and adult mobility, and abundant prey items supporting larval feeding after the yolk sac is depleted, and nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival. Eulachon prey on a wide variety of species including crustaceans such as copepods and euphausiids (Hay and McCarter 2000, WDFW and ODFW 2001), unidentified malacostracans (Sturdevant et al. 1999), cumaceans (Smith and Saalfeld 1955) mysids, barnacle larvae, and worm larvae (WDFW and ODFW 2001).

3.2 Shortnose Sturgeon

Status. We used information available in the Shortnose Sturgeon Recovery Plan (NMFS 1998), the 2010 NMFS Biological Assessment (SNS BA 2010), and the listing document (32 FR 4001) to summarize the status of the species. Shortnose sturgeon were listed as endangered throughout its range on March 11, 1967 pursuant to the Endangered Species Preservation Act of 1966. Shortnose sturgeon remained on the list as endangered with enactment of the ESA in 1973. Shortnose sturgeon occur along the Atlantic Coast of North America, from the Saint John River in Canada to the Saint Johns River in Florida. The Shortnose Sturgeon Recovery Plan describes 19 shortnose sturgeon populations that are managed separately in the wild. Two additional geographically separated populations occur behind dams in the Connecticut River (above the Holyoke Dam) and in Lake Marion on the Santee-Cooper River system in South Carolina (above the Wilson and Pinopolis Dams). While shortnose sturgeon spawning has been documented in several rivers across its range (including but not limited to: Kennebec River, ME, Connecticut River, Hudson River, Delaware River, Pee Dee River, SC, Savannah, Ogeechee, and Altamaha rivers, GA), status for many other rivers remain unknown.

Life History. Sturgeon are a long-lived species, taking years to reach sexual maturity. Male shortnose sturgeon tend to sexually mature earlier than females, and sturgeon residing in more northern latitudes reach maturity later than those at southerly latitudes. Sturgeon are broadcast spawners, with females laying adhesive eggs on hard bottom, rocky substrate at upstream, freshwater sites. When the males arrive at the spawning site, they broadcast sperm into the water column to fertilize the eggs. Despite their high fecundity, sturgeon have low recruitment.

Spawning periodicity varies by species and sex, but there can be anywhere from 1 to 5 years between spawning, as individuals need to rebuild gonadal material. There is difficulty in definitively assessing where and how reliably spawning occurs. Presence of eggs, age-1 juveniles and capture of “ripe” adults moving upstream (i.e., likely on a spawning run) serve as strong indicators, but due to their life history and the impacts sturgeon populations have taken, there are additional hurdles to successful spawning. Because sturgeon are iteroparous, and populations in some areas so depleted, eggs deposited at the spawning grounds may not be fertilized if males do not arrive at the spawning grounds that year.

Hatching occurs approximately 94-140 hrs after egg deposition, and larvae assume a bottom-dwelling existence. The yolksac larval stage is completed in about 8-12 days, during which time larvae move downstream to rearing grounds over a 6 – 12 day period. Size of larvae at hatching and at the juvenile stage varies by species. During the daytime, larvae use benthic structure (e.g., gravel matrix) as refugia. Juvenile sturgeon continue to move further downstream into brackish waters, and eventually become residents in estuarine waters for months or years.

Generally, sturgeon are benthic omnivores, feeding on benthic invertebrates that are abundant in the substrate in that area. Shortnose sturgeon forage over sandy bottom, and eat benthic invertebrates like amphipods.

Juvenile shortnose generally move upstream during spring and summer and downstream for fall and winter; however, these movements usually occur above the salt- and freshwater interface. During summer and winter, adult shortnose sturgeon inhabit freshwater reaches of rivers and streams influenced by tides. During summer, at the southern end of its range, shortnose sturgeon congregate in cool, deep, areas of rivers taking refuge from high temperatures. Adult shortnose sturgeon prefer deep, downstream areas with soft substrate and vegetated bottoms, if present. Because they rarely leave their natal rivers, shortnose sturgeon are considered to be freshwater amphidromous (i.e. adults spawn in freshwater but regularly enter saltwater habitats during their life).

Despite the life span of adult sturgeon, the viability of sturgeon populations is highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population. This relationship caused Secor et al. (2002) to conclude sturgeon populations can be grouped into two demographic categories: populations having reliable (albeit periodic) natural recruitment and those that do not. The shortnose sturgeon populations without reliable natural recruitment are at more risk. Several authors have also demonstrated that sturgeon populations generally, and shortnose sturgeon populations in particular, are much more sensitive to adult mortality than other species of fish. Sturgeon populations cannot survive fishing related mortalities exceeding five percent of an adult spawning run and they are vulnerable to declines and local extinction if juveniles die from fishing related mortalities (Secor et al. 2002).

Shortnose sturgeon populations are at risk from incidental bycatch, loss of habitat, dams, dredging and pollution. These threats are likely to continue into the future. We conclude that the shortnose sturgeon’s resilience to further perturbation is low.

Threats. The 1998 recovery plan for shortnose sturgeon (NMFS 1998) identify Habitat degradation or loss (resulting, for example, from dams, bridge construction, channel dredging, and pollutant discharges), and mortality (for example, from impingement on cooling water intake screens, dredging, and incidental capture in other fisheries) as principal threats to the species' survival. Introductions and transfers of indigenous and nonindigenous sturgeon, intentional or accidental, may threaten wild shortnose sturgeon populations by imposing genetic threats, increasing competition for food or habitat, or spreading diseases. Sturgeon species are susceptible to viruses enzootic to the west coast and fish introductions could further spread these diseases.

Designated critical habitat. No critical habitat has been designated for shortnose sturgeon.

3.3 Atlantic sturgeon (5 DPSs)

Status. The range of Atlantic sturgeon includes the St. John River in Canada, to St. Johns River in Florida. EPA has NPDES permitting authority throughout New Hampshire, Massachusetts, the District of Columbia, Federally operated facilities in Delaware and Tribal lands in Connecticut, Rhode Island, New York, North Carolina, and Florida The five DPSs of Atlantic sturgeon are Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic.

Life history. Although the Atlantic sturgeon DPSs are genetically distinct, their life history characteristics are the same and are discussed together. As Acipensieriformes, Atlantic sturgeon are anadromous and iteroparus. Like shortnose sturgeon, male Atlantic sturgeon tend to sexually mature earlier than females, and sturgeon residing in more northern latitudes reach maturity later than those at southerly latitudes. Evidence of Atlantic sturgeon spawning has been found in many of the same rivers as shortnose sturgeon (see discussion above). Atlantic sturgeon eggs are between 2.5-3.0mm, and larvae are about 7mm long upon hatching. Generally, sturgeon are benthic omnivores, feeding on benthic invertebrates that are abundant in the substrate in that area. Atlantic sturgeon commonly eat polychaetes and isopods.

As juveniles, Atlantic sturgeon migrate downstream from the spawning grounds into brackish water. Unlike shortnose sturgeon, subadult Atlantic sturgeon (76-92cm) may move out of the estuaries and into coastal waters where they can undergo long range migrations. At this stage in the coastal waters, individual subadult and adult Atlantic sturgeon originating from different DPSs will mix, but adults return to their natal river to spawn.

Threats. Of the stressors evaluated in the 2007 status review (ASSRT 2007), bycatch mortality, water quality, lack of adequate state and/or Federal regulatory mechanisms, and dredging activities were most often identified as the most significant threats to the viability of Atlantic sturgeon populations. Additionally, some populations were affected by unique stressors, such as habitat impediments (e.g., Cape Fear and Santee-Cooper rivers) and apparent ship strikes (e.g., Delaware and James rivers).

Designated critical habitat. The proposed designated critical habitat for Atlantic sturgeon includes tidally-affected accessible waters of coastal estuaries where the species occurs. The essential features of the proposed designated critical habitat for the Atlantic sturgeon DPSs within these rivers do not include plant or animal life that may be affected by the stressors of the action.

From north to south, the rivers and waterways that make up the spatial extent of designated critical habitat are detailed in Table 4.

Table 3. River Systems Included in Proposed Designated Critical Habitat for Atlantic Sturgeon.

Distinct Population Unit	River/Waterway		
Gulf of Maine	Penobscot Piscataqua	Kennebec Merrimack	Androscoggin
New York Bight	Connecticut Housatonic Delaware	Housatonic	Hudson
Chesapeake Bay	Susquehanna York James	Potomac Mattaponi	Rappahannock Pamunkey
Carolina	Roanoke Cape Fear Waccamaw Santee South Santee Wateree Santee Lake Moultrie	Tar - Pamlico Northeast Cape Fear Bull Creek Rediversion Canal Tailrace Canal Cooper Broad Lake Marion	Neuse Pee Dee Black North Santee Cooper Congaree Diversion Canal
South Atlantic	North Fork Edisto North Edisto Savannah Ocmulgee St. Marys	South Fork Edisto South Edisto Ogeechee Altamaha	Edisto Combahee - Salkehatchie Oconee Satilla

3.4 Green sturgeon, southern DPS

Status. The most recent 5-year status review was published in August of 2015. Green sturgeon occur in coastal Pacific waters from San Francisco Bay to Canada. The Southern DPS of green sturgeon includes populations south of (and exclusive of) the Eel River, coastal and Central Valley populations, and the spawning population in the Sacramento River, CA (Adams et al. 2007). We used information available in the 2002 Status Review and 2005 Status Review Update (GSSR 2002, 2005, 2015), and the proposed and final listing rules to summarize the status of the species.

The 2015 status update indicates that DPS structure of the North American green sturgeon has not changed and that many of the principle factors considered when listing Southern DPS green sturgeon as threatened are relatively unchanged. Loss of spawning habitat and bycatch in the white sturgeon commercial fishery are two major causes for the species decline. Spawning in the Feather River is encouraging and the decommissioning of Red Bluff Diversion Dam and breach of Shanghai Bench makes spawning conditions more favorable. The prohibition of retention in commercial and recreational fisheries has eliminated a known threat and likely had a very positive effect on the overall population, although recruitment indices are not presently available.

Life history. As members of the family Acipenseridae, green sturgeon share similar reproductive strategies and life history patterns with other sturgeon species; see discussion for shortnose sturgeon above. The Sacramento River is the location of the single, known spawning population for the green sturgeon Southern DPS (Adams et al. 2007). Green sturgeon have relatively large eggs compared to other sturgeon species (4.34 mm) and grow rapidly, reaching 66 mm in three weeks. Generally, sturgeon are benthic omnivores, feeding on benthic invertebrates that are abundant in the substrate in that area. Little is known specifically about green sturgeon foraging habits; generally, adults feed upon invertebrates like shrimp, mollusks, amphipods and even small fish, while juveniles eat opossum shrimp and amphipods. Juvenile green sturgeon spend 1-3 years in freshwater, disperse widely in the ocean, and return to freshwater as adults to spawn (about age 15 for males, age 17 for females).

Threats. The 2015 status review (NMFS 2015a) for the southern DPS of green sturgeon indicates that many of the principle factors considered when listing Southern DPS green sturgeon as threatened are relatively unchanged. Current threats to the Southern DPS include entrainment by water projects, contaminants, incidental bycatch and poaching. Given the small population size, the species’ life history traits (e.g., slow to reach sexual maturity), and that the threats to the population are likely to continue into the future, the Southern DPS is not resilient to further perturbations. The spawning area for the species is still small, as the species still encounters impassible barriers in the Sacramento, Feather and other rivers that limit their spawning range. Entrainment threat includes stranding in flood diversions during high water events.

Designated critical habitat. Critical habitat for the Southern DPS of green sturgeon was designated on October 9, 2009 (74 FR 52300), including coastal United States marine waters within 60 fathoms deep from Monterey Bay, California to Cape Flattery, Washington, including the Strait of Juan de Fuca, and numerous coastal rivers and estuaries: see the Final Rule for a complete description (74 FR 52300). Essential features identified in this designation that may be affects by the stressors of the action include acceptably low levels of contaminants (e.g., pesticides, PAHs, heavy metals that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon) and abundant prey items (benthic invertebrates and fish) for subadults and adults.

4 *MARINE FISH*

4.1 Bocaccio Puget Sound/Georgia Basin DPS

The bocaccio that occur in the Georgia Basin are listed as an endangered “species,” which, in this case, refers to a distinct segment of a vertebrate population. The listing includes bocaccio throughout Puget Sound, which encompasses all waters south of a line connecting Point Wilson on the Olympic Peninsula and Partridge on Whidbey Island; West Point on Whidbey Island, Deception Island, and Rosario Head on Fidalgo Island; and the southern end of Swinomish Channel between Fidalgo Island and McGlinn Island (United States Geological Survey 1979), and the Strait of Georgia, which encompasses the waters inland of Vancouver Island, the Gulf Islands, and the mainland coast of British Columbia.

Status. Bocaccio have always been rare in recreational fisheries that occur in North Puget Sound and the Strait of Georgia; however, there have been no confirmed reports of bocaccio in Georgia Basin for several years. Although their abundance cannot be estimated directly, NMFS’ BRT estimated that the populations of bocaccio and yelloweye rockfish are small in size, probably numbering fewer than 10,000 individuals in Georgia Basin and fewer than 1,000 total individuals in Puget Sound (Drake et al. 2010). Georgia Basin bocaccio are most common at depths between 50 and 250 meters (160 and 820 feet).

Life history. Preferred bocaccio habitat is largely dependent upon the life stage of an individual. Larvae and young juveniles tend to be found in deeper offshore regions (1-148 km offshore), but associated with the surface and occasionally with floating kelp mats (Hartmann 1987, Love et al. 2002, Emery et al. 2006). Mating occurs between August and November, with larvae born between January and April (Lyubimova 1965, Moser 1967, Westrheim 1975, Echeverria 1987, Love et al. 2002, MacCall and He 2002).As individuals mature into older juveniles and adults, they transition into shallow waters and settle to the bottom, preferring algae-covered rocky, eelgrass, or sand habitats and aggregating into schools (Eschmeyer et al. 1983, Love et al. 1991). After a few weeks, fish move into slightly deeper waters of 18-30 m and occupy rocky reefs (Feder et al. 1974, Carr 1983, Eschmeyer et al. 1983, Johnson 2006, Love and Yoklavich 2008). As adults, bocaccio may be found in depths of 12-478 m, but tend to remain in shallow waters on the continental shelf (20-250 m), still associating mostly with reefs or other hard substrate, but may move over mud flats (Feder et al. 1974, Kramer and O'Connell 1995, Love et al. 2002, Love et al. 2005, Love and York 2005, Love et al. 2006). Artificial habitats, such as platform structures, also appear to be suitable habitat for bocaccio (Love and York 2006). Adults may occupy territories of 200-400 hectares, but can venture outside of this territory (Hartmann 1987). Adults tend to occupy deeper waters in the southern population compared to the northern population (Love et al. 2002). Adults are not as benthic as juveniles and may occur as much as 30 m above the bottom and move 100 m vertically during the course of a day as they move between different areas (Starr 1998, Love et al. 2002). Prior to severe population reductions, bocaccio appeared to frequent the Tacoma Narrows in Washington State (DeLacy et al. 1964, Haw and Buckley 1971, Miller and Borton 1980).

Prey of bocaccio vary with fish age, with bocaccio larvae starting with larval krill, diatoms, and dinoflagellates (Love et al. 2002). Pelagic juveniles consume fish larvae, copepods, and krill, while older, nearshore juveniles and adults prey upon rockfishes, hake, sablefish, anchovies, lanternfish, and squid (Reilly et al. 1992, Love et al. 2002).

Threats. The 2016 draft recovery plan for rockfish indicates that historical overfishing is recognized as the primary cause of the decline of rockfishes in Puget Sound (Palsson et al. 2008, Drake et al. 2010, Williams et al. 2010), there is some uncertainty about the relative impact of some fisheries today, and of the additional remaining threats, which include degraded water quality and habitat, contaminants, derelict fishing gear, and other threats (Palsson et al. 2008, Drake et al. 2010, WDFW 2013).

Designated critical habitat. NMFS proposed critical habitat designation of approximately 1,185 mi^2 of marine habitat for bocaccio in Puget Sound, Washington. Physical or biological features essential to adult bocaccio include the benthic habitats or sites deeper than 30m (98 ft) that possess or are adjacent to areas of complex bathymetry consisting of rock and or highly rugose habitat are essential to conservation because these features support growth, survival, reproduction, and feeding opportunities by providing the structure for rockfish to avoid predation, seek food and persist for decades. Several attributes of these sites determine the quality of the habitat and are useful in considering the conservation value of the associated feature, and whether the feature may require special management considerations or protection. These attributes are also relevant in the evaluation of the effects of a proposed action in a section 7 consultation if the specific area containing the site is designated as critical habitat. These attributes include: (1) Quantity, quality and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities, (2) water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities, and (3) the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance.

4.2 Rockfish, Yelloweye and Canary (Puget Sound/Georgia Basin)

Status. In July of 2016 NMFS petitioned to delist the canary rockfish based on newly obtained genetic information that demonstrates that the Puget Sound/Georgia Basin canary rockfish population does not meet the DPS criteria and therefore does not qualify for listing under the ESA. Georgia Basin yelloweye rockfish occur through Puget Sound, which encompasses all waters south of a line connecting Point Wilson on the Olympic Peninsula and Partridge on Whidbey Island; West Point on Whidbey Island, Deception Island, and Rosario Head on Fidalgo Island; and the southern end of Swinomish Channel between Fidalgo Island and McGlinn Island (United States Geological Survey 1979), and the Strait of Georgia, which encompasses the waters inland of Vancouver Island, the Gulf Islands, and the mainland coast of British Columbia.

The frequency of yelloweye rockfish in collections from Puget Sound appears to have been highly variable; frequencies were less than 1 percent in the 1960s and 1980s and about 3 percent in the 1970s and 1990s. In North Puget Sound, however, the frequency of yelloweye rockfish has been estimated to have declined from a high of greater than 3 percent in the 1970s to about 0.65 percent in more recent samples. This decline combined with their low intrinsic growth potential, threats from bycatch in commercial and recreational fisheries, loss of nearshore rearing habitat, chemical contamination, and the proportion of coastal areas with low dissolved oxygen levels led to this species' listing as threatened under the ESA.

Although their abundance cannot be estimated directly, NMFS' BRT estimated that the populations of bocaccio, yelloweye rockfish and canary rockfish are small in size, probably numbering fewer than 10,000 individuals in Georgia Basin and fewer than 1,000 total individuals in Puget Sound (Drake et al. 2010).

Georgia Basin yelloweye rockfish are most common at depths between 91 and 180 meters (300 to 580 feet), although they may occur in waters 50 to 475 meters (160 and 1,400 feet) deep. Larval rockfish occur over areas that extend several hundred miles offshore where they are passively dispersed by ocean currents and remain in larval form and as small juveniles for several months (Auth and Brodeur 2006, Moser and Boehlert 1991). They appear to concentrate over the continental shelf and slope, but have been captured more than 250 nautical miles offshore of the Oregon coast (Richardson and Laroche 1979, Moser and Boehlert 1991). Larval rockfish have been reported to be uniformly distributed at depths of 13, 37 and 117 meters below surface. Densities were highest at the 37- and 177-meter depths (Lenarz et al. 1991).

Life history. As with bocaccio, yelloweye habitat varies based upon life stage. Larvae maintain a pelagic existence but as juveniles, move into shallow high relief rocky or sponge garden habitats (Eschmeyer et al. 1983, Richards et al. 1985, Love et al. 1991). Juveniles may also associate with floating debris or pilings (Lamb and Edgell 1986). As adults, yelloweye rockfish move in to deeper habitats. Individuals have been found in waters as deep as 549 m, but are generally found in waters of less than 180 m (Eschmeyer et al. 1983, Love et al. 2002). However, adults continue to associate with rocky, high relief habitats, particularly with caves and crevices, pinnacles, and boulder fields (Carlson and Straty 1981, Richards 1986, Love et al. 1991, O'Connell and Carlisle 1993, Yoklavich et al. 2000). Yelloweyes generally occur as individuals, with loose, residential aggregations infrequently found (Coombs 1979, DeMott 1983, Love et al. 2002). In the Puget Sound region, sport catch records from the 1970's indicate that Sucia Island and other islands of the San Juans as well as Bellingham Bay had the highest concentrations of catches (Delacy et al. 1972, Miller and Borton 1980).

Yelloweye rockfish prey upon different species and size classes throughout their development. Larval and juvenile rockfish prey upon phyto- and zooplankton (Lee and Sampson 2009). Adult yelloweyes eat other rockfish (including members of their own species), sand lance, gadids, flatfishes, shrimp, crabs, and gastropods (Love et al. 2005, Yamanaka et al. 2006).

Designated critical habitat. Physical or biological features essential to the conservation of both adult and juvenile yelloweye rockfish are the same as for adult bocaccio and adult canary rockfish.

4.3 Nassau Grouper

The Nassau grouper (*Epinephelus striatus*) is primarily a shallow-water, insular fish species found from inshore to about 330 feet (100m) depth. The species is distributed throughout the islands of the western Atlantic including Bermuda, the Bahamas, southern Florida and along the coasts of central and northern South America. It is not known from the Gulf of Mexico except at Campeche Bank off the coast of the Yucatan Peninsula, at Tortugas, and off Key West. Adults are generally found near coral reefs and rocky bottoms while juveniles are found in shallower waters in and around coral clumps covered with macroalgae (*Laurencia* spp.) and over seagrass beds. Their diet is mostly fishes and crabs, with diet varying by age/size. Juveniles feed mostly on crustaceans, while adults (>30 cm; 11.8 in) forage mainly on fish. The Nassau grouper usually forages alone and is not a specialized forager.

Under the authority of the Magnuson-Stevens Fisheries Act, NMFS classified the Nassau grouper as “overfished” in its October 1998 “Report to Congress on the status of Fisheries and Identification of overfished Stocks.”

Life History. Nassau grouper exhibit no sexual dimorphism in body shape or color. The species passes through a juvenile bisexual phase, with gonads consisting of both immature spermatogenic and immature ovarian tissue, before maturing directly as male or female. The minimum age at sexual maturity is between four and eight years when reaching a size of 400-500 mm standard length (Olsen and LaPlace 1979, Bush et al. 2006). The major determinant of maturity appears to be size rather than age, as fish raised in captivity reached maturity at 27-28 months (Tucker and Woodward 1994).

Nassau grouper reproduce in site-specific spawning aggregations. Spawning aggregations, of a few dozen up to perhaps thousands of individuals have been reported from the Bahamas, Jamaica, Cayman Islands, Belize, and the Virgin Islands. These aggregations occur in depths of 20-40 m (65.6-131.2 ft) at specific locations of the outer reef shelf edge. Spawning takes place in December and January, around the time of the full moon, in waters 25-26 degrees C (77-78.8 degrees F). Because Nassau grouper spawn in aggregations at historic areas and at very specific times, they are easily targeted during reproduction. Because Nassau grouper mature relatively late (4-8 years), many juveniles may be taken by the fishery before they have a chance to reproduce.

Designated critical habitat. Critical habitat has not been designated for this species.

5 Sea Turtles

Sea turtles share the common threats described below.

Bycatch: Fishing is the primary anthropogenic threat to sea turtles in the ocean. Fishing gear entanglement potentially drowns or seriously injures sea turtles. Fishing dredges can crush and entrap turtles, causing death and serious injury. Infection of entanglement wounds can compromise health. The development and operation of marinas and docks in inshore waters can negatively impact nearshore habitats. Turtles swimming or feeding at or just beneath the surface of the water are particularly vulnerable to boat and vessel strikes, which can result in serious propeller injuries and death.

Marine Debris: Ingestion or entanglement in marine debris is a cause of morbidity and mortality for sea turtles in the pelagic (open ocean) environment (Stamper et al. 2009). Consumption of non-nutritive debris also reduces the amount of nutritive food ingested, which then may decrease somatic growth and reproduction (McCauley and Bjorndal 1999). Marine debris is especially problematic for turtles that spend all or significant portions of their life cycle in the pelagic environment (e.g., leatherbacks, juvenile loggerheads, and juvenile green turtles).

Habitat Disturbance: Sea turtle nesting and marine environments are facing increasing impacts through structural modifications, sand nourishment, and sand extraction to support widespread development and tourism (Lutcavage et al. 1997, Bouchard et al. 1998, Hamann et al. 2006, Maison 2006, Hernandez et al. 2007, Santidrián Tomillo et al. 2007, Patino-Martinez 2013).

These factors decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings through direct loss of and indirect (e.g., altered temperatures, erosion) mechanisms (Ackerman 1997, Witherington et al. 2003, 2007). Lights from developments alter nesting adult behavior and are often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Witherington and Bjorndal 1991, Witherington 1992, Cowan et al. 2002, Deem et al. 2007, Bourgeois et al. 2009).

Beach nourishment also affects the incubation environment and nest success. Although the placement of sand on beaches may provide a greater quantity of nesting habitat, the quality of that habitat may be less suitable than pre-existing natural beaches. Constructed beaches tend to differ from natural beaches in several important ways. They are typically wider, flatter, more compact, and the sediments are more moist than those on natural beaches (Nelson et al. 1987) (Ackerman 1997, Ernest and Martin 1999). Nesting success typically declines for the first year or two following construction, even when more nesting area is available for turtles (Trindell et al. 1998, Ernest and Martin 1999, Herren 1999). Likely causes of reduced nesting success on constructed beaches include increased sand compaction, escarpment formation, and changes in beach profile (Nelson et al. 1987, Grain et al. 1995, Lutcavage et al. 1997, Steinitz et al. 1998, Ernest and Martin 1999, Rumbold et al. 2001). Compaction can inhibit nest construction or increase the amount of time it takes for turtles to construct nests, while escarpments often cause female turtles to return to the ocean without nesting or to deposit their nests seaward of the escarpment where they are more susceptible to frequent and prolonged tidal inundation. In short, sub-optimal nesting habitat may cause decreased nesting success, place an increased energy burden on nesting females, result in abnormal nest construction, and reduce the survivorship of eggs and hatchlings. In addition, sand used to nourish beaches may have a different composition than the original beach; thus introducing lighter or darker sand, consequently affecting the relative nest temperatures (Ackerman 1997, Milton et al. 1997).

In addition to effects on sea turtle nesting habitat, anthropogenic disturbances also threaten coastal foraging habitats, particularly areas rich in seagrass and marine algae. Coastal habitats are degraded by pollutants from coastal runoff, marina and dock construction, dredging, aquaculture, oil and gas exploration and extraction, increased under water noise and boat traffic, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999, Lee Long et al. 2000, Waycott et al. 2005).

Pollutants: Conant (2009) included a review of the impacts of marine pollutants on sea turtles: marine debris, oil spills, and bioaccumulative chemicals. Sea turtles at all life stages appear to be highly sensitive to oil spills, perhaps due to certain aspects of their biology and behavior, including a lack of avoidance behavior, indiscriminate feeding in convergence zones, and large pre-dive inhalations (Milton and Lutz 2003). Milton et al. (2003) state that the oil effects on turtles include increased egg mortality and developmental defects, direct mortality due to oiling in hatchlings, juveniles and adults, and impacts to the skin, blood, salt glands, and digestive and immune systems. Vargo et al. (1986) reported that sea turtles would be at substantial risk if they encountered an oil spill or large amounts of tar in the environment. In a review of available information on debris ingestion, Balazs (1985) reported that tar balls were the second most prevalent type of debris ingested by sea turtles. Physiological experiments showed that sea turtles exposed to petroleum products may suffer inflammatory dermatitis, ventilator disturbance, salt gland dysfunction or failure, red blood cell disturbances, immune response, and digestive disorders (Vargo et al. 1986, Lutcavage et al. 1995).

Natural Threats: A number of threats are common to all sea turtles.[1] Predation is a primary natural threat. While cold stunning is not a major concern for leatherback sea turtles, which can tolerate low water temperatures, it is considered a major natural threat to other sea turtle species. Disease is also a factor in sea turtle survival. Fibropapillomatosis (FP) tumors are a major threat to green turtles in some areas of the world and is particularly associated with degraded coastal habitat. Scientists have also documented FP in populations of loggerhead, olive ridley, and flatback turtles, but reports in green turtles are more common. Large tumors can interfere with feeding and essential behaviors, and tumors on the eyes can cause permanent blindness. FP was first described in green turtles in the Florida Keys in the 1930s. Since then it has been recorded in many green turtle populations around the world. The effects of FP at the population level are not well understood. The sand-borne fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* capable of killing greater than 90 percent of sea turtle embryos they infect, threatening nesting productivity under some conditions. These pathogens can survive on decaying organic matter and embryo mortality rates attributed to fusarium were associated with clay/silt nesting areas compared to sandy areas (Sarmiento-Ramırez et al. 2014).

Climate Change. Conant's (2009) review describes the potentially extensive impacts of climate change on all aspects of a sea turtle's life cycle, as well as impact the abundance and distribution of prey items. Rising sea level is one of the most certain consequences of climate change (Titus and Narayanan 1995), and will result in increased erosion rates along nesting beaches. This could particularly affect areas with low-lying beaches where sand depth is a limiting factor, as the sea will inundate nesting sites and decrease available nesting habitat (Fish et al. 2005, Baker et al. 2006). The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Baker et al. 2006). On some undeveloped beaches, shoreline migration will have limited effects on the suitability of nesting habitat. The Bruun rule specifies that during a sea level rise, a typical beach profile will maintain its configuration but will be translated landward and upward (Rosati et al. 2013). However, along developed coastlines, and especially in areas where erosion control structures have been constructed to limit shoreline movement, rising sea levels will cause severe effects on nesting females and their eggs. Erosion control structures can result in the permanent loss of dry nesting beach or deter nesting females from reaching suitable nesting sites (Council 1990). Nesting females may deposit eggs seaward of the erosion control structures potentially subjecting them to repeated tidal inundation. Non-native vegetation often out competes native species, is usually less stabilizing, and can lead to increased erosion and degradation of suitable nesting habitat. Exotic vegetation may also form impenetrable root mats that can prevent proper nest cavity excavation, invade and desiccate eggs, or trap hatchlings.

5.1 Leatherback Sea Turtle

Status. The leatherback sea turtle is unique among sea turtles for its large size, wide distribution (due to thermoregulatory systems and behavior), and lack of a hard, bony carapace. It ranges from tropical to subpolar latitudes, worldwide.

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[1] See hyperlink to NMFS information on sea turtles: http://www.nmfs.noaa.gov/pr/species/turtles/threats.htm, updated June 16, 2014

The global population of adult females has declined over 70 percent in less than one generation, from an estimated 115,000 adult females in 1980 to 34,500 adult females in 1995 (Pritchard 1982, Spotila et al. 1996). There may be as many as 34,000 – 94,000 adult leather backs in the North Atlantic, alone (TEWG 2007), but dramatic reductions (> 80 percent) have occurred in several populations in the Pacific, which was once considered the stronghold of the species (Sarti Martinez 2000). The 2013 five year review (NMFS and USFWS 2013b) reports that the East Pacific and Malaysia leatherback populations have collapsed, yet Atlantic populations generally appear to be stable or increasing. Many explanations have been provided to explain the disparate population trends, including fecundity and foraging differences seen in the Pacific, Atlantic, and Indian Oceans. Since the last 5-year review, studies indicate that high reproductive output and consistent and high quality foraging areas in the Atlantic Ocean have contributed to the stable or recovering populations; whereas prey abundance and distribution may be more patchy in the Pacific Ocean, making it difficult for leatherbacks to meet their energetic demands and lowering their reproductive output. Both natural and anthropogenic threats to nesting and marine habitats continue to affect leatherback populations, including the 2004 tsunami in the Indian Ocean, 2010 oil spill in the United States Gulf of Mexico, logging practices, development, and tourism impacts on nesting beaches in several countries.

In 2015, NMFS announced a new program to focus and redouble its efforts to protect some of the species that are currently among the most at risk of extinction in the near future with the goal of reversing their declining trend so that the species will become a candidate for recovery in the future. The leatherback sea turtle is one of the eight species identified for this initiative (NMFS 2015b). These species were identified as among the most at-risk of extinction based on three criteria (1) endangered listing, (2) declining populations, and (3) are considered a recovery priority #1. A priority #1 species is one whose extinction is almost certain in the immediate future because of a rapid population decline or habitat destruction, whose limiting factors and threats are well understood and the needed management actions are known and have a high probability of success, and is a species that is in conflict with construction or other developmental projects or other forms of economic activity.

Life history. Estimates of age at maturity ranges from 5 to 29 years (Spotila et al. 1996, Avens et al. 2009). Females nest every 1 to 7 years. Natal homing, at least within an ocean basin, results in reproductive isolation between five broad geographic regions: eastern and western Pacific, eastern and western Atlantic, and Indian Ocean. Leatherback sea turtles migrate long, transoceanic distances between their tropical nesting beaches and the highly productive temperate waters where they forage, primarily on jellyfish and tunicates. These gelatinous prey are relatively nutrient-poor, such that leatherbacks must consume large quantities to support their body weight (James et al. 2005, Wallace et al. 2006).

Designated critical habitat. On March 23, 1979, leatherback designated critical habitat was identified adjacent to Sandy Point, St. Croix, Virgin Islands from the 183 m isobath to mean high tide level between 17° 42’12” N and 65°50’00” W. This habitat is essential for nesting, which has been increasingly threatened since 1979, when tourism increased significantly, bringing nesting habitat and people into close and frequent proximity; however, studies do not support significant designated critical habitat deterioration. Additional designated critical habitat for the leatherback sea turtle includes approximately 43,798 km2 stretching along the California coast from Point Arena to Point Arguello east of the 3000 m depth contour; and 64,760 km^2 stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 m depth contour. The designated areas comprise approximately 108558 km2 of marine habitat and include waters from the ocean surface down to a maximum depth of 80 m. They were designated specifically because of the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (i.e., jellyfish), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

5.2 Hawksbill Sea Turtle

Status. The hawksbill sea turtle has a sharp, curved, beak-like mouth. It has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical oceans. The species was first listed under the Endangered Species Conservation Act (35 FR 8491) and listed as endangered under the ESA since 1973.

The hawksbill turtle was once abundant in tropical and subtropical regions throughout the world. Over the last century, this species has declined in most areas and stands at only a fraction of its historical abundance. According to the 2013 status review (NMFS and USFWS 2013a), nesting populations in the eastern Pacific, and the Nicaragua nesting population in the western Caribbean appears to have improved. However, the trends and distribution of the species throughout the globe largely is unchanged. Although greatly depleted from historical levels, nesting populations in the Atlantic in general are doing better than in the Indian and Pacific Oceans. In the Atlantic, more population increases have been recorded in the insular Caribbean than along the western Caribbean mainland or the eastern Atlantic. In general, hawksbills are doing better in the Indian Ocean (especially the southwestern and northwestern Indian Ocean) than in the Pacific Ocean. The situation for hawksbills in the Pacific Ocean is particularly dire, despite the fact that it still has more nesting hawksbills than in either the Atlantic or Indian Oceans.

Life history. Hawksbill sea turtles reach sexual maturity at 20 to 40 years of age. Females return to their natal beaches every 2 to 5 years to nest (an average of 3 to 5 times per season). Clutch sizes are large (up to 250 eggs). Sex determination is temperature dependent, with warmer incubation producing more females. Hatchlings migrate to and remain in pelagic habitats until they reach approximately 22 to 25 cm in straight carapace length. As juveniles, they take up residency in coastal waters to forage and grow. As adults, hawksbills use their sharp beak-like mouths to feed on sponges and corals.

Designated critical habitat. NMFS established designated critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico. Aspects of these areas that are important for hawksbill sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for hawksbill sea turtle prey.

5.3 Kemp's Ridley Sea Turtle

Status. The Kemp's ridley is the smallest of all sea turtle species and considered to be the most endangered sea turtle, internationally (Zwinenberg 1977, Groombridge 1982, TEWG 2000). According to the 2015 status review (NMFS and USFWS 2013a), population growth rate (as measured by numbers of nests) stopped abruptly after 2009. Given the recent lower nest numbers, the population is not projected to grow at former rates. An unprecedented mortality in subadult and adult females post-2009 nesting season may have altered the 2009 age structure and momentum of the population, which had a carryover impact on annual nest numbers in 2011-2014. The results indicate the population is not recovering and cannot meet recovery goals unless survival rates improve. The Deep Water Horizon oil spill that occurred at the onset of the 2010 nesting season and exposed Kemp's ridleys to oil in nearshore and offshore habitats may have been a factor in fewer females nesting in subsequent years, however this is still under evaluation. The long-term impacts from the Deep Water Horizon oil spill and response to the spill (e.g., dispersants) to sea turtles are not yet known. Given the Gulf of Mexico is an area of high-density offshore oil exploration and extraction, future oil spills are highly probable and Kemp's ridleys and their habitat may be exposed and injured. Commercial and recreational fisheries continue to pose a substantial threat to the Kemp's ridley despite measures to reduce bycatch. Kemp's ridleys have the highest rate of interaction with fisheries operating in the Gulf of Mexico and Atlantic Ocean than any other species of turtle.

Life history. Adult Kemp's ridley sea turtles have an average straight carapace length of 2.1 ft (65 cm). Females mature at 12 years of age. The average remigration is 2 years. Nesting occurs from April to July in large arribadas, primarily at Rancho Nuevo, Mexico. Females lay an average of 2.5 clutches per season. The annual average clutch size is 97 – 100 eggs per nest. The nesting location may be particularly important because hatchlings can more easily migrate to foraging grounds in deeper oceanic waters, where they remain for approximately 2 years before returning to nearshore coastal habitats. Juvenile Kemp's ridley sea turtles use these nearshore coastal habitats from April through November, but move towards more suitable overwintering habitat in deeper offshore waters (or more southern waters along the Atlantic coast) as water temperature drops. Adult habitat largely consists of sandy and muddy areas in shallow, nearshore waters less than 120 ft (37 m) deep, although they can also be found in deeper offshore waters. As adults, Kemp's ridleys forage on swimming crabs, fish, jellyfish, mollusks, and tunicates.

Designated critical habitat. Critical habitat has not been designated for this species.

5.4 Loggerhead Sea Turtle

Status. The loggerhead sea turtle is distinguished from other turtles by its large head and powerful jaws. The North Pacific Ocean DPS ranges throughout tropical to temperate waters in the North Pacific. Based on the 2009 status review (Conant et al. 2009), for three of five DPSs with sufficient data (Northwest Atlantic Ocean, South Pacific Ocean, and North Pacific Ocean), analyses indicate a high likelihood of quasi-extinction. Similarly, threat matrix analysis indicated that all other DPSs have the potential for a severe decline in the future.

North Pacific Ocean Loggerhead sea turtle DPS life history. Mean age at first reproduction for female loggerhead sea turtles is 30 years (SD = 5). Females lay an average of three clutches per season. The annual average clutch size is 112 eggs per nest. The average remigration interval is 2.7 years. Nesting occurs primarily on Japanese beaches, where warm, humid sand temperatures incubate the eggs. Temperature determines the sex of the turtle during the middle of the incubation period. Turtles spend the post-hatchling stage in pelagic waters. The juvenile stage is spent first in the oceanic zone (Kuroshio Extension Bifurcation Region) and later in the neritic zone (i.e., coastal waters) in the eastern and central Pacific. Coastal waters in the eastern and western North Pacific provide important foraging habitat, inter-nesting habitat, and migratory habitat for adult loggerheads.

Northwest Atlantic Ocean Loggerhead sea turtle DPS life history. Mean age at first reproduction for female loggerhead sea turtles is 30 years (SD = 5). Mating occurs in the spring, and eggs are laid throughout the summer. Northwest Atlantic females lay an average of five clutches per season. The annual average clutch size is 115 eggs per nest. The average remigration interval is 3.7 years (Tucker 2010). Nesting occurs primarily on beaches along the Southeastern Coast of the United States, from southern Virginia to Alabama. Additional nesting occurs on beaches throughout the Gulf of Mexico and Caribbean Sea. Temperature determines the sex of the turtle during the middle of the incubation period. Post- hatchling loggerheads from southeast United States nesting beaches may linger for months in waters just off the nesting beach or become transported by ocean currents within the Gulf of Mexico and North Atlantic, where they become associated with Sargassum habitats, driftlines, and other convergence zones.

The juvenile stage is spent first in the oceanic zone (e.g., waters around the Azores, Madeira, Morocco, and the Grand Banks off Newfoundland) and later in the neritic zone (i.e., continental shelf waters) from Cape Cod Bay, Massachusetts, south through Florida, the Caribbean, and the Gulf of Mexico. Neritic stage juveniles often inhabit relatively enclosed, shallow water estuarine habitats with limited ocean access. Juveniles are omnivorous and forage on crabs, mollusks, jellyfish and vegetation at or near the surface (Dodd 1988). Adults inhabit shallow water habitats with large expanses of open ocean access, as well as continental shelf waters. Sub-adult and adult loggerheads prey on benthic invertebrates such as mollusks and decapod crustaceans in hard bottom, coastal habitats.

Northwest Atlantic Ocean Loggerhead sea turtle DPS designated critical habitat. The final designated critical habitat for the Northwest Atlantic Ocean loggerhead DPS within the Atlantic Ocean and the Gulf of Mexico includes 36 occupied marine areas within the range of the Northwest Atlantic Ocean DPS. These areas contain one or a combination of nearshore reproductive habitat, winter area, breeding areas, and migratory corridors.

5.5 Green sea turtle

The green sea turtle is the largest of the hardshell marine turtles, growing to a weight of 350 lb (159 kg) and a straight carapace length of greater than 3.3 ft (1 m). It has a circumglobal distribution, occurring throughout nearshore tropical, subtropical, and, to a lesser extent, temperate waters. The species was separated into two listing designations: endangered for breeding populations in Florida and the Pacific coast of Mexico, and threatened in all other areas throughout its range. On August 1, 2012, NMFS found that a petition to identify the Hawaiian population of green turtle as a DPS, and to delist the DPS, may be warranted (77 FR 45571). In April 2016, we removed the range-wide and breeding population listings of the green sea turtle, and in their place, listed 8 DPSs as threatened and 3 DPSs as endangered (81 FR 20057). Among these, only the North Atlantic DPS occurs in waters where EPA has permitting authority.

Life history throughout range. Age at first reproduction for females is 20 - 40 years. They lay an average of three nests per season with an average of 100 eggs per nest. The remigration interval (i.e., return to natal beaches) is 2 – 5 years. Nesting occurs primarily on beaches with intact dune structure, native vegetation, and appropriate incubation temperatures during summer months. After emerging from the nest, hatchlings swim to offshore areas and go through a post-hatchling pelagic stage where they are believed to live for several years. During this life stage, green sea turtles feed close to the surface on a variety of marine algae and other life associated with drift lines and debris. Adult turtles exhibit site fidelity and migrate hundreds to thousands of kilometers from nesting beaches to foraging areas. Green sea turtles spend the majority of their lives in coastal foraging grounds, which include open coastlines and protected bays and lagoons. Adult green turtles feed primarily on seagrasses and algae, although they also eat jellyfish, sponges, and other invertebrate prey.

Status. Once abundant in tropical and subtropical waters, globally, green sea turtles exist at a fraction of their historical abundance, as a result of over-exploitation. The North Atlantic DPS is characterized by geographically widespread nesting with eight sites having high levels of abundance (i.e., <1,000 nesters). Nesting is reported in 16 countries and/or United States Territories at 73 sites. This region is data rich and has some of the longest running studies on nesting and foraging turtles anywhere in the world. All major nesting populations demonstrate long-term increases in abundance. The prevalence of FP has reached epidemic proportions in some parts of the North Atlantic DPS.

The extent to which this will affect the long-term outlook for green turtles in the North Atlantic DPS is unknown and remains a concern, although nesting trends across the DPS continue to increase despite the high incidence of the disease. There are still concerns about future risks, including habitat degradation (particularly coastal development), bycatch in fishing gear, continued turtle and egg harvesting, and climate change.

Designated critical habitat. On September 2, 1998, NMFS designated critical habitat for green sea turtles (63 FR 46694), which include coastal waters surrounding Culebra Island, Puerto Rico. Seagrass beds surrounding Culebra provide important foraging resources for juvenile, subadult, and adult green sea turtles. Additionally, coral reefs surrounding the island provide resting shelter and protection from predators. This area provides important developmental habitat for the species.

6 CORALS

There are currently 22 coral species listed as threatened under the ESA, 16 of which occur in the action area (Table 5). Information from the proposed listings and status reports (ABRT 2005) were used to summarize the status of these species

Table 4: Threatened coral species occurring in the PGP action area

Threatened Corals	Currently Known in These United States Geographic Areas			
	Caribbean Waters: Puerto Rico			
Acropora cervicornis (Staghorn)*and designated critical habitat*	X			
Acropora palmata (Elkhorn) *and designated critical habitat*	X			
Mycetophyllia ferox	X			
Dendrogyra cylindrus	X			
Orbicella annularis	X			
Orbicella faveolata	X			
Orbicella franksi	X			
	Pacific Waters			
	Guam	**Commonwealth of Northern Mariana Islands**	**Pacific Remote Island Areas**	**American Samoa**
Acropora globiceps	X	X	X	X
Acropora jacquelineae				X
Acropora retusa	X		X	X
Acropora rudis				X
Acropora speciosa			X	X
Euphyllia paradivisa				X
Isopora crateriformis				X
Pavona diffluens	X	X		X
Seriatopora aculeata	X			

Life history. The threatened coral species include true stony corals (class Anthozoa, order Scleractinia), the blue coral (class Anthozoa, order Helioporacea), and fire corals (class

Hydrozoa, order Milleporina). All threatened species are reef-building corals, because they secrete massive calcium carbonate skeletons that form the physical structure of coral reefs.

Reef-building coral species are capable of rapid calcification rates because of their symbiotic relationship with single-celled dinoflagellate algae, zooxanthellae, which occur in great numbers within the host coral tissues. Zooxanthellae photosynthesize during the daytime, producing an abundant source of energy for the host coral that enables rapid growth. At night, polyps extend their tentacles to filter-feed on microscopic particles in the water column such as zooplankton, providing additional nutrients for the host coral. In this way, reef-building corals obtain nutrients autotrophically (i.e., via photosynthesis) during the day, and heterotrophically (i.e., via predation) at night.

Most coral species use both sexual and asexual propagation. Sexual reproduction in corals is primarily through gametogenesis (i.e., development of eggs and sperm within the polyps near the base). Some coral species have separate sexes (gonochoric), while others are hermaphroditic. Strategies for fertilization are by either “brooding” or “broadcast spawning” (i.e., internal or external fertilization, respectively). Brooding is relatively more common in the Caribbean, where nearly 50 percent of the species are brooders, compared to less than 20 percent of species in the Indo-Pacific. Asexual reproduction in coral species most commonly involves fragmentation, where colony pieces or fragments are dislodged from larger colonies to establish new colonies, although the budding of new polyps within a colony can also be considered asexual reproduction. In many species of branching corals, fragmentation is a common and sometimes dominant means of propagation.

Reef-building corals do not thrive outside of an area characterized by a fairly narrow mean temperature range (typically 25 °C-30 °C). Two other important factors influencing suitability of habitat are light and water quality.

Threats. Massive mortality events from disease conditions of corals and the keystone grazing urchin *Diadema antillarum* have precipitated widespread and dramatic changes in reef community structure. Large-scale coral bleaching reduces population viability. In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and bleaching events from ocean warming have added to the poor state of coral populations and yielded a remnant coral community with increased dominance by weedy brooding species, decreased overall coral cover, and increased macroalgal cover. Additionally, iron enrichment may predispose the basin to algal growth. Further, coral growth rates in many areas have been declining over decades. Such reductions prevent successful recruitment as a result of reduced density. Finally, climate change is likely to result in the endangerment of many species as a result of temperature increases (and resultant bleaching), sea level rises, and ocean acidification.

Designated critical habitat. On November 26, 2008, NMFS designated critical habitat for elkhorn and staghorn coral. They designated marine habitat in four specific areas: Florida (1,329 square miles), Puerto Rico (1,383 square miles), St. John/St. Thomas (121 square miles), and St. Croix (126 square miles). These areas support the following physical or biological features that are essential to the conservation of the species: substrate of suitable quality and availability to support successful larval settlement and recruitment and reattachment and recruitment of fragments.

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APPENDIX B

COMPREHENSIVE ENVIRONMENTAL BASELINE

The Environmental Baseline is defined as: "past and present impacts of all Federal, State, or private actions and other human activities in an action area, the anticipated impacts of all proposed Federal projects in an action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process" (50 CFR 402.02). The key purpose of the Environmental Baseline is to describe the natural and anthropogenic factors influencing the status and condition of ESA-listed species and designated critical habitat in the action area. Since this is a programmatic consultation on what is primarily a continuing action with a large geographic scope, this Environmental Baseline focuses more generally on the status and trends of the aquatic ecosystems in the U.S. and the consequences of that status for listed resources.

Activities that negatively impact water quality also threaten aquatic species. The deterioration of water quality is a contributing factor that has led to the endangerment of some aquatic species under NMFS jurisdiction. Declines in populations of listed species leave them vulnerable to a multitude of threats. Due to the cumulative effects of reduced abundance, low or highly variable growth capacity, and the loss of essential habitat, these species are less resilient to additional disturbances. In larger populations, stressors that affect only a limited number of individuals could once be tolerated by the species without resulting in population level impacts; in smaller populations, the same stressors are more likely to reduce the likelihood of survival. It is with this understanding of the environmental baseline that we consider the effects of the proposed action, including the likely effect that the PGP will have on endangered and threatened species and their designated critical habitat. There may be direct and indirect effects of activities associated with the proposed PGP in streams, wetlands, rivers, lakes, estuaries, irrigation canals, and drainage systems into, over, and in close proximity to which pesticides are applied. Areas adjacent to or downstream from these jurisdictional areas may be indirectly affected by activities authorized under the PGP.

1 *REGIONS WITHIN THE ACTION AREA*

We identified the following regions and states for inclusion in the Environmental Baseline section of this opinion: Pacific Coast (Washington, Idaho, Oregon, and California); New England (Maine, New Hampshire, Vermont, and Massachusetts); Mid-Atlantic (District of Columbia, Delaware, and Virginia); U.S. Caribbean (Puerto Rico) and U.S. Pacific Islands (excluding Hawaii). These regions/states cover the vast majority of the proposed action area. At the regional level, our baseline assessment focused on the natural and anthropogenic threats affecting the listed species (and their habitats) within the action area for each particular region: Pacific Coast – all listed ESUs and DPSs of Pacific salmon and steelhead, eulachon, Southern DPS green sturgeon, and Southern Resident killer whale; New England – Atlantic salmon, Atlantic sturgeon (5 listed DPSs); Mid-Atlantic - Atlantic sturgeon (5 listed DPSs); Caribbean –

Nassau grouper, elkhorn coral, staghorn coral, lobed star coral, boulder star coral, mountainous star coral, pillar coral, and rough cactus coral; Pacific Islands – all listed Pacific Islands coral species.

While there are some Tribal lands and federal facilities in regions or states not mentioned above, in general these areas are either very small, far removed from listed species or habitat, or not affected by the proposed action. For example, any discharges of pesticide pollutants on Tribal lands in Florida would have to be transported through Everglades or Big Cypress National Parks, where they would be degraded by exposure to sunlight, microbial action and chemical processes. While all areas of overlap between ESA-listed species (and their critical habitat) and the PGP coverage area are evaluated in this opinion, the Environmental Baseline will focus specifically on the aquatic ecosystems in the regions/states (listed above) where the anticipated effects of the proposed action are considered more likely to adversely affect listed species.

The action area for this consultation covers a very large number of individual watersheds and an even larger number of specific water bodies (e.g., lakes, rivers, streams, estuaries). It is, therefore, not practicable to describe the environmental baseline and assess risk for each particular area where the PGP may authorize discharges and activities. Accordingly, this opinion approaches the Environmental Baseline more generally by describing the activities, conditions and stressors which adversely affect ESA-listed species and designated critical habitat. These include natural threats (e.g., parasites and disease, predation and competition, wildland fires), water quality, hydromodification projects, land use changes, dredging, mining, artificial propagation, non-native species, fisheries, vessel traffic, and climate changes. For each of these threats we start with a general overview of the problem, followed by a more focused analysis at the regional and state level for the species listed above, as appropriate and where such data are available.

Our summary of the Environmental Baseline complements the information provided in the Status of Listed Resources section of this opinion, and provides the background necessary to evaluate and interpret information presented in the Effects of the Proposed Action and Cumulative Effects sections to follow. We then evaluate the consequences of EPA's proposed action in combination with the status of the species, environmental baseline and the cumulative effects to determine whether EPA can insure that the likelihood of jeopardy or adverse modification of designated critical habitat will be avoided.

2 NATURAL THREATS

Natural mortality rates for some ESA listed species are already high due to a combination of contributing threats including parasites and/or disease, predation, water quality and quantity, wildland fire, oceanographic features and climatic variability. Natural mortality often varies for a given species depending on life stage or habitat. While species continuously co-evolve and adapt to changes in the natural environment, when combined with, and often compounded by, anthropogenic threats such natural threats can contribute significantly to the decline and endangerment of species.

2.1 Parasites and Disease

Fish disease and parasitic organisms occur naturally in the water. Many fish species are highly susceptible to parasites and disease, particularly during early life stages. Native fish have co-evolved with such organisms and individuals can often carry diseases and parasites at less than lethal levels. However, outbreaks may occur when stress from disease and parasites is compounded by other stressors such as diminished water quality, flows, and crowding (Spence and Hughes 1996, Guillen 2003). At higher than normal water temperatures salmonids may become stressed and lose their resistance to diseases (Spence and Hughes 1996). Consequently, diseased fish become more susceptible to predation and are less able to perform essential functions, such as feeding, swimming, and defending territories (McCullough 1999).

Salmonids are susceptible to numerous bacterial, viral, and fungal diseases. The more common bacterial diseases in New England waters include furunculosis, bacterial kidney disease, enteric redmouth disease, coldwater disease, and vibriosis (Olafesen and Roberts 1993), (Egusa and Kothekar 1992). There are over 30 identified parasites of Atlantic salmon including external parasites (Scott and Scott 1988, Hoffman 1999). Several species sea lice, a marine ectoparasite found in Atlantic and Pacific coastal waters, can cause deadly infestations of farm-grown salmon and may also affect wild salmon. While captive fish in aquaculture have the highest risk for transmission and outbreaks of such diseases, wild fish that must pass near aquaculture facilities are at risk of encountering both parasites and pathogens from hatchery operations. Although substantial progress has been made in recent years to reduce the risks to wild fish, this remains a potential threat.

Parasites also occur in both wild-caught and cultivated Nassau grouper, predominantly in the viscera and gonads. These include encysted larval tapeworms, nematode, isopods, and trematodes (Manter 1947, Thompson and Munro 1978).

Coral diseases are a common and significant threat affecting most or all coral species and regions to some degree, although the scientific understanding of the causes and mechanisms of coral diseases remains very poor. Disease adversely affects various coral life history events by, among other processes, causing adult mortality, reducing sexual and asexual reproductive success, and impairing colony growth. A diseased state results from a complex interplay of factors including the cause or agent (e.g., pathogen, environmental toxicant), the host, and the environment. All coral disease impacts are presumed to be attributable to infectious diseases or to poorly-described genetic defects. Coral disease often produces acute tissue loss. Other manifestations of disease in the broader sense, such as coral bleaching from ocean warming, are discussed under other the anthropogenic threats of ocean warming as a result of global climate change. Increased prevalence and severity of diseases is correlated with increased water temperatures and bleaching, which may correspond to increased virulence of pathogens, decreased resistance of hosts, or both (Bruno et al. 2007, Muller and Woesik 2012, Rogers and Muller 2012). Moreover, the expanding coral disease threat may result from opportunistic pathogens that become damaging only in situations where the host integrity is compromised by physiological stress or immune suppression. Coral resistance to disease can also be diminished by other stressors such as predation and nutrients. White band disease is thought to be the major factor responsible for the rapid loss of Atlantic Acropora due to mass mortalities. Significant population declines of star coral species have been linked to disease impacts, both with and without prior bleaching (Bruckner and Bruckner 2006, Miller et al. 2009). Disease outbreaks can persist for years in a population—star coral colonies suffering from yellow-band in Puerto Rico still manifested similar disease signs four years later (Bruckner and Bruckner 2006). Pillar coral and rough cactus coral are susceptible to extensive impacts and rapid tissue loss from white plague disease (Dustan 1977, Miller et al. 2006). The incidence of coral disease also appears to be expanding geographically in the Indo-Pacific, and there is evidence that corals with massive morphology damage are not recovering from disease events.

Although little is known about the threat of infectious diseases to killer whale populations in the wild, deaths of captive individuals have been attributed to pneumonia, systemic mycosis, other bacterial infections, and mediastinal abscesses (Gaydos et al. 2004). Marine *Brucella*, *Edwardsiella tarda*, and cetacean poxvirus, were detected in wild individuals. Marine *Brucella* and cetacean poxvirus have the potential to cause mortality in calves and marine *Brucella* has induced abortions in bottle-nose dolphins (Miller et al. 1999, Van Bressem et al. 1999). Pathogens identified from other species of toothed whales that are sympatric with the Southern Residents are potentially transmittable to killer whales (Palmer et al. 1991, Gaydos et al. 2004). Several, including porpoise morbillivirus, dolphin morbillivirus, and herpes viruses, are highly virulent and are capable of causing large-scale disease outbreaks in some related species. Killer whales are susceptible to other forms of disease, including Hodgkin’s disease and severe atherosclerosis of the coronary arteries (Roberts Jr et al. 1965, Yonezawa et al. 1989). Tumors and bone fusion have also been recorded (NMFS 2008b). Disease epidemics have never been reported in killer whales in the northeastern Pacific (Gaydos et al. 2004). No severe parasitic infestations have been reported in killer whales in the northeastern Pacific (NMFS 2008b).

2.2 Predation

Predation is a natural and necessary process in properly functioning aquatic ecosystems. In order to survive, species evolve a suite of strategies that allow them to co-exist with the numerous and diverse predators they encounter throughout their life cycle. However, natural predator-prey relationships in aquatic ecosystems have been substantially altered through the impacts of anthropogenic changes, often resulting in increased risk to populations of threatened and endangered species. High rates of predation may jeopardize viability of populations that are already experiencing significantly reduced abundance due to the cumulative effects of multiple stressors.

2.2.1 Salmonids

Salmonids are exposed to high rates of natural predation, during freshwater rearing and migration stages, as well as during ocean migration. Salmon along the U.S. west coast are prey for marine mammals, birds, sharks, and other fishes. In the Pacific Northwest, the increasing size of tern, seal, and sea lion populations in recent decades may have reduced the survival of some salmon ESUs/DPSs. Human barriers commonly aggregate fish, where they are subject to intense predation. Such locations include Ballard Locks in Seattle and the Bonneville Dam (Gustafson et al. 1997). Threatened Puget Sound Chinook adults are preferred prey (up to 78 percent of identified prey) of endangered Southern Resident killer whales during late spring to fall (Hanson et al. 2005, Ford et al. 2010). Several species of seals prey on Atlantic salmon in estuarine and marine areas and could exert a substantial impact on populations which have already been depleted due to other stressors (Cairns and Reddin 2000). Large numbers of fry and juvenile Pacific salmon are eaten by piscivorous birds such. Stream-type juveniles are vulnerable to bird predation in estuaries. Caspian terns and cormorants may be responsible for the mortality of up to 6 percent of the outmigrating stream-type juveniles in the Columbia River basin (Roby et al. 2007). Mergansers and kingfishers are likely the most important predators of Atlantic salmon in freshwater environments (Cairns and Reddin 2000). In estuarine environments, double crested cormorants are considered an important predator of smolts as they transition to life at sea because osmotic stress due to sea water entry likely enhances the predation risk at this life stage (Handeland et al. 1996). Avian predators of adult salmonids include bald eagles and osprey (Pearcy 1997). Overall freshwater fish predators native to Maine pose little threat to the Gulf of Maine DPS (Fay et al. 2006).

2.2.2 Non-salmonid Species

In estuarine and marine environments striped bass, Atlantic cod, pollock, porbeagle shark, Greenland shark, Atlantic halibut, and many other fish species have been recorded as predators of salmon at sea (Hvidsten and Møkkelgjerd 1987, Mills 1989, and Mills 1993 all cited in Fay, 2006). The primary fish predators in estuaries are probably adult salmonids or juvenile salmonids which emigrate at older and larger sizes than others (Beamish et al. 1992, Beamish and Neville 1995).

The impact of natural predation on sturgeon at various life stages is unknown. The presence of bony scutes is an effective adaptation for minimizing predation of sturgeon greater than 25 mm total length (Gadomski and Parsley 2005). Documented predators of sturgeon include sea lampreys, gar, striped bass, common carp, northern pikeminnow, channel catfish, smallmouth bass, walleye, grey seal, fallfish and sea lion (Scott and Crossman 1973, Dadswell et al. 1984, Kynard and Horgan 2002, Gadomski and Parsley 2005). Predation by non-native catfish species may also have an impact on early life stages of several Atlantic sturgeon DPSs. Pinnepeds are known predators of Southern DPS green sturgeon and populations of both Eastern DPS Steller and California sea lions have increased in recent decades (Caretta et al. 2009, NMFS 2013).

Predation of North American green sturgeon by white sharks has also been documented off Central California (Klimley 1985).

Large numbers of predators commonly congregate at eulachon spawning runs (Willson et al. 2006) and was identified as a moderate threat to eulachon in the Fraser River and mainland British Columbia rivers, and a low severity threat to eulachon in the Columbia and Klamath rivers. Information on predation on Nassau grouper is lacking. Sharks were reported to attack Nassau groupers at spawning aggregations in the Virgin Islands, and there is one report of cannibalism in this species (Olsen and LaPlace 1979 cited in NMFS, 2013). Although there is currently no legal directed fishery for Nassau grouper in the U.S. and possession is prohibited, they are still caught and released as bycatch in some fisheries. Predators can have important direct and indirect impacts on coral colonies. Predation on some coral genera by many corallivorous species of fish and invertebrates (e.g., snails and seastars) is a chronic threat that has been identified for most coral life stages. Prior to settlement and metamorphosis, coral larvae experience considerable mortality (up to 90 percent or more) from predation or other factors (Goreau et al. 1981). Because newly settled corals barely protrude above the substrate, juveniles need to reach a certain size to reduce damage or mortality from impacts such as grazing, sediment burial, and algal overgrowth (Bak and Elgershuizen 1976, Sammarco 1985). Predation of coral colonies can increase the likelihood of the colonies being infected by disease, and likewise diseased colonies may be more likely to be preyed upon. Predation impacts are typically greatest when population abundances are low as, in most cases, coral predators have not been subject to the same degrees of disturbance mortality and their broad diet breadth has allowed them to persist at high levels despite decreases in coral prey (FR 79 53852). Coral exposure to predation is naturally moderated by presence of predators of the corallivores. For example, corallivorous reef fish prey on corals, and piscivorous reef fish and sharks prey on the corallivores; thus, high abundances of piscivorous reef fish and sharks moderate coral predation.

Crown-of-thorns seastar can reduce living coral cover to less than one percent during outbreaks, dramatically changing coral community structure, promoting algal colonization, and affecting fish population dynamics (FR 79 53852).

The most important predators on Atlantic *Acropora* spp. are fireworm and muricid snail. Although these predators rarely kill entire colonies, there are several possible mechanisms of indirect impact. Because they prey on the growing tips (including the apical polyps), especially of *A. cervicornis*, growth of the colony may be arrested for prolonged periods of time. Another important coral predator is the gastropod, *Coralliophila abbreviata* which feeds on a wide range of corals, but seems to be particularly damaging to *Acropora* spp. (Baums et al. 2003) . Several species of damselfish establish algal nursery gardens within branching *Acropora* spp. (Itzkowitz 1978, Sammarco and Williams 1982). Although not predators in the strict sense, damselfish nip off living coral tissue, thus denuding the skeleton to make a place for their algal gardens. As with other predators, it is likely that the impacts of damselfish are proportionally greater when population abundances of *Acropora* are already reduced due to other stressors.

2.3 Wildland Fire

Wildland fires that are allowed to burn naturally in riparian or upland areas may benefit or harm aquatic species, depending on the degree of departure from natural fire regimes. Fire is one of the dominant habitat-forming processes in mountain streams (Bisson et al. 2003). The patchy, mosaic pattern burned by fires provides a refuge for those fish and invertebrates that leave a burning area or simply spares some fish that were in a different location at the time of the fire (Murphy 2000). Although most fires are small in size, large size fires increase the chances of adverse effects on aquatic species. Large fires that burn near the shores of streams and rivers can have biologically significant short-term effects. These include increased water temperatures, ash, nutrients, pH, sediment, toxic chemicals, and loss of large woody debris (Buchwalter et al. 2004, Rinne 2004). Such fires can result in fish kills and the indirect effects of displacement as fish are forced to swim downstream to avoid poor water quality conditions (Gresswell 1999, Rinne 2004). Small fires or fires that burn entirely in upland areas also cause ash to enter rivers and increase smoke in the atmosphere, contributing to ammonia concentrations in rivers as the smoke adsorbs into the water (Gresswell 1999). The presence of ash can have indirect effects on aquatic species depending on the quantity deposited into the water. All ESA-listed salmonids rely on macroinvertebrates as a food source for at least a portion of their life histories. When small amounts of ash enter the water, there are usually no noticeable changes to the macroinvertebrate community or water quality (Bowman and Minshall 2000). When significant amounts of ash are deposited into rivers, the macroinvertebrate community density and composition may be moderately to drastically reduced for a full year, with milder long-term effects lasting 10 years or more (Minshall et al. 2001, Buchwalter et al. 2004). Larger fires can also indirectly affect fish by altering water quality. Ash and smoke contribute to elevated ammonium, nitrate, phosphorous, potassium, and pH, which can remain elevated for up to four months after forest fires (Buchwalter et al. 2003). Within the action area for this opinion, wildland fires of the size and proximity to aquatic ecosystems that may result in adverse effects on listed species are concentrated in the Pacific Coast region.

2.4 Oceanographic Features and Climatic Variability

Oceanographic conditions and natural climatic variability may affect Pacific salmonids within the action area. There is evidence that Pacific salmon abundance may have fluctuated for centuries as a consequence of dynamic oceanographic conditions (Beamish and Bouillon 1993, Finney et al. 2002, Beamish et al. 2009). Sediment cores reconstructed for 2,200-year records have shown that Northeastern Pacific fish stocks have historically been regulated by these climate regimes (Finney et al. 2002). The long-term pattern of the Aleutian low pressure system corresponds with historical trends in salmon catches, copepod production, and other climatic indices, indicating that climate and the marine environment play an important role in salmon production. Pacific salmon abundance and corresponding worldwide catches tend to be large during naturally-occurring periods of strong Aleutian low pressure causing stormier winters and upwelling, positive Pacific decadal oscillation , and an above average Pacific circulation index

(Beamish et al. 2009). Periods of increasing Aleutian low pressure correspond with periods of high pink and chum salmon production and low coho and Chinook salmon production (Beamish et al. 2009). The abundance and distribution of salmon and zooplankton also relate to shifts in North Pacific atmospheric and oceanic climate (Francis and Hare 1994). Over the past century, regime shifts have occurred as a result of the North Pacific's natural climate regime. Reversals in the prevailing polarity of the Pacific Decadal Oscillation occurred around 1925, 1947, 1977, and 1989 (Mantua et al. 1997, Hare and Mantua 2000). The reversals in 1947 and 1977 correspond to dramatic shifts in salmon production regimes in the North Pacific Ocean (Mantua et al. 1997). Poor environmental conditions for salmon survival and growth may be more prevalent with projected increases in ocean warming and acidification. Anthropogenic climate change (discussed in more detail below) may exacerbate the effects that natural oceanographic conditions and climatic variability have on listed species, although the synergistic effects of these combined stressors is largely unknown at this time.

3 ANTHROPOGENIC THREATS

The quality of the biophysical components within aquatic ecosystems is affected by human activities conducted within and around coastal waters, estuarine and riparian zones, as well as those conducted more remotely in the upland portion of the watershed. Industrial activities can result in discharge of pollutants, changes in water temperature and levels of dissolved oxygen, and the addition of nutrients. In addition, forestry and agricultural practices can result in erosion, run-off of fertilizers, herbicides, insecticides or other chemicals, nutrient enrichment and alteration of water flow. Chemicals such as chlordane, DDE, DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the river bottom and are later consumed by benthic feeders, such as macroinvertebrates, and then work their way higher into the food web (e.g., to sturgeon and sea turtles). Some of these compounds may affect physiological processes and impede a fish's ability to withstand stress, while simultaneously increasing the stress of the surrounding environment by reducing dissolved oxygen, altering pH, and altering other physical properties of the water body. Coastal and riparian areas are also heavily impacted by development and urbanization resulting in storm water discharges, non-point source pollution and erosion. Section 2.1 *Status of Aquatic Ecosystem Health* describes the health status and trends of the U.S. coastal zone, rivers, streams and wetlands in the geographic areas covered by the PGP that overlap with ESA-listed species under NMFS jurisdiction. Section 1.2.2 focuses specifically on the effects of pesticides on aquatic ecosystems as is relevant to the proposed action in this opinion. Sections 2.3 through 2.8 describe other anthropogenic stressors and threats that result in both direct and indirect adverse effects on listed species and their critical habitats within the action area. These include hydromodification projects (dams, channelization, and water diversion), dredging, mining, population growth and land use changes, artificial propagation, non-native species introductions, direct harvest and bycatch, vessel related stressors (strikes, noise, harassment), and climate change.

3.1 Status of Aquatic Ecosystem Health

This section describes the current status and recent health trends of aquatic ecosystems within the action area. EPA sampling results (USEPA 2015) are summarized by region for the following biological, chemical, and physical indicators: 1) Biological – benthic macroinvertebrates; 2) Chemical – phosphorous, nitrogen, ecological fish tissue contaminants, sediment contaminants, sediment toxicity, and pesticides; and 3) Physical – dissolved oxygen, salinity, water clarity, pH, and Chlorophyll a. Cumulatively, these key indicators provide us with an overall picture of the ecological condition of aquatic ecosystems. Different thresholds, based on published references and the best professional judgment of regional experts, are used to evaluate each region as "good," "fair," or "poor" for each water quality indicator. EPA rates overall water quality from results of the five key indicators using the following guidelines: "poor" – two or more component indicators are rated poor; "fair" - one indicator is rated poor, or two or more are rated fair; "good" - no indicators are rated poor, and a maximum of one is rated fair.

The benthic macroinvertebrates (e.g., worms, mollusks, and crustaceans) that inhabit the bottom substrates of aquatic ecosystems are an important food source for a wide variety of fish, mammals, and birds. Benthic communities serve as reliable biological indicators of environmental quality because they are sensitive to chemical contamination, dissolved oxygen stresses, salinity fluctuations, and sediment disturbances. A good benthic index rating means that benthic habitats contain a wide variety of species, including low proportions of pollution-tolerant species and high proportions of pollution-sensitive species. A poor benthic index rating indicates that benthic communities are less diverse than expected and are populated by more pollution-tolerant species and fewer pollution-sensitive species than expected.

Chemical and physical components are measured as indicators of key stressors that have the potential to degrade biological integrity. Some of these are naturally occurring and others result only from human activities, but most come from both sources. EPA evaluates overall water quality based on the following primary indicators: surface nutrient enrichment—dissolved inorganic nitrogen and dissolved inorganic phosphorus concentrations; algae biomass—surface chlorophyll a concentration; and potential adverse effects of eutrophication—water clarity and bottom dissolved oxygen levels (USEPA 2015). Contaminants, including some pesticides, PCBs and mercury, also contribute to ecological degradation. Many contaminants adsorb onto suspended particles and accumulate in areas where sediments are deposited and may adversely affect sediment-dwelling organisms. As other organisms eat contaminated sediment-dwellers the contaminants can accumulate in organisms and potentially become concentrated throughout the food web.

3.1.1 Northeast Region (Maine to Virginia)

A wide variety of coastal environments are found in the Northeast region including rocky coasts, drowned river valleys, estuaries, salt marshes, and city harbors. The Northeast is the most populous coastal region in the U.S.. In 2010, the region was home to 54.2 million people, representing about a third of the nation's total coastal population (USEPA 2015). The population in this area has increased by ten million residents (~ 23 percent) since 1970. The coast from Cape

Cod to the Chesapeake Bay consists of larger watersheds that are drained by major riverine systems that empty into relatively shallow and poorly flushed estuaries. These estuaries are more susceptible to the pressures of a highly populated and industrialized coastal region.

A total of 238 sites were sampled to assess approximately 10,700 square miles of Northeast coastal waters. Figure 1 shows a summary of findings from the EPA's National Coastal Condition Assessment Report for the Northeast Region (USEPA 2015). Biological quality is rated as good in 62 percent of the Northeast coast region based on the benthic index. Poor biological conditions occur in 27 percent of the coastal area. About 11 percent of the region reported missing results, due primarily to difficulties in collecting benthic samples along the rocky coast north of Cape Cod. Based on the water quality index, 44 percent of the Northeast coast is in good condition, 49 percent is rated fair, and 6 percent is rated poor.

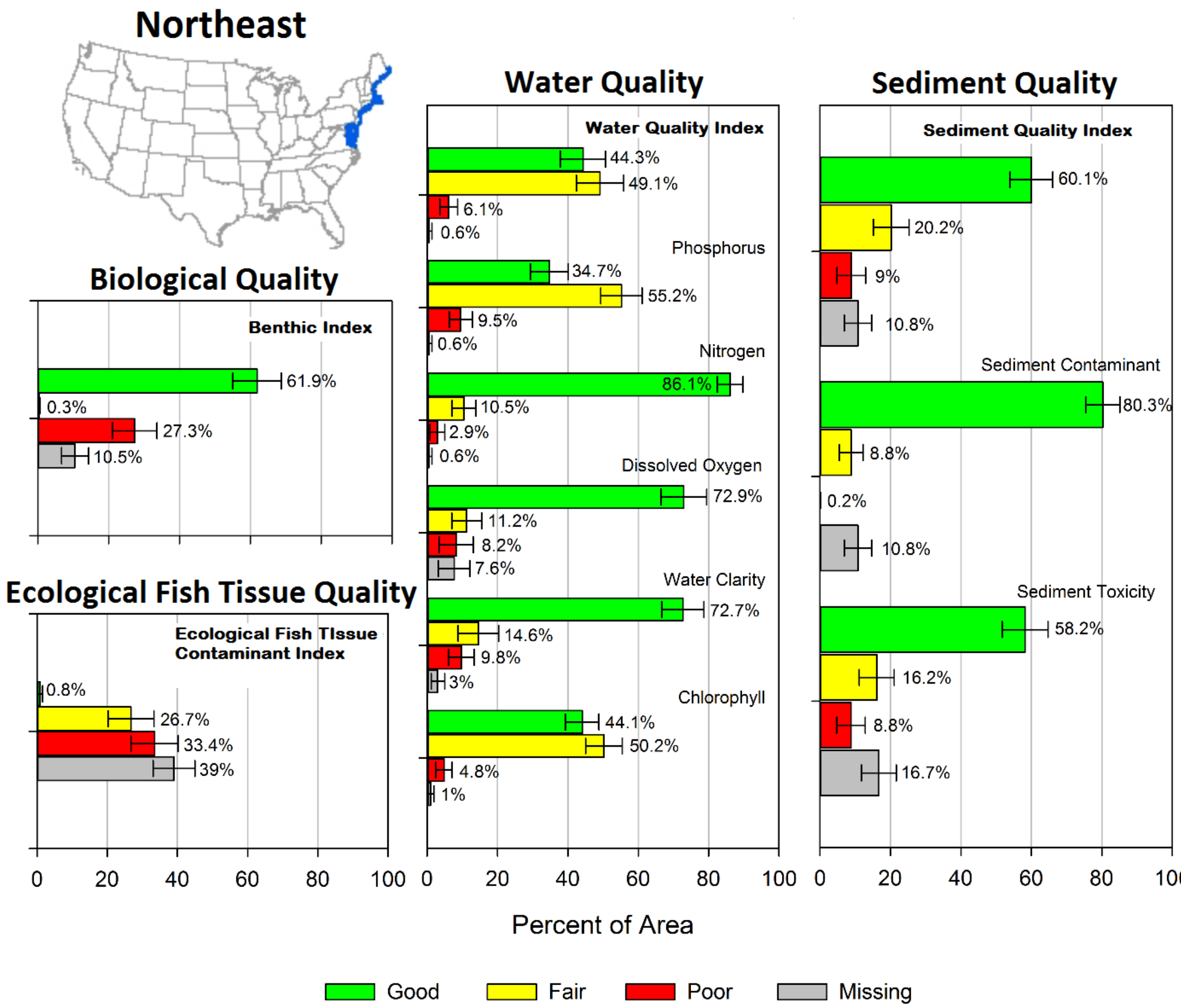

Figure 1. National Coastal Condition Assessment 2010 Report findings for the Northeast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 238 sites sampled). Error bars represent 95 percent confidence levels (USEPA 2015).

Based on the sediment quality index, 60 percent of the Northeast coastal area sampled is in good condition, 20 percent is in fair condition, and 9 percent is in poor condition (11 percent were reported “missing”). Compared to ecological risk-based thresholds for fish tissue contamination, less than 1 percent of the Northeast coast is rated as good, 27 percent is rated fair, and 33 percent is rated poor. Researchers were unable to evaluate fish tissue for 39 percent of the region, including almost the entire Acadian Province, because target species were not caught for analysis. The contaminants that most often exceed the thresholds for a “poor” rating in the assessed areas of the Northeast coast are selenium, mercury, arsenic, and, in a small proportion of the area, total PCBs.

New Hampshire

New Hampshire conducted site specific water quality assessments on 42 percent of rivers, 81 percent of aquatic estuarine waters, and 85 percent of ocean waters within the state. Results reported in the New Hampshire 2012 Surface Water Quality Report indicate that approximately 0.8 percent of freshwater rivers and stream mileage is fully supportive of aquatic life, 26.0 percent is not supportive, and 73.2 percent could not be assessed due to insufficient information (NHDES 2012). In estuarine waters, approximately 0.8 percent of the square mileage is fully supportive of aquatic life, 91.9 percent is not supportive and 7.2 percent could not be assessed due to insufficient information. Twenty-six percent of estuarine waters fully met the water quality standards, 54 percent were impaired, and 19 percent could not be assessed due to insufficient information. In ocean waters, approximately 94.1 percent of the square mileage is fully supportive of aquatic life, 0.0 percent is not supportive and 5.9 percent could not be assessed due to insufficient information (NHDES 2012). Fifty-six percent of ocean waters fully met the water quality standards, 29 percent were impaired, and 15 percent could not be assessed due to insufficient information. All of New Hampshire waters are impaired by mercury contamination in fish tissue, with the source being atmospheric deposition. All of New Hampshire's bays and estuaries are impaired by dioxins and PCBs. The top five reasons for impairment in New Hampshire rivers for 2012 were: mercury (16,962 acres), pH (3,821 acres), E coli (1,306 acres), dissolved oxygen (688 acres), and aluminum (563 acres) (NHDES 2012). The top five reasons for impairment in New Hampshire estuaries for 2012 were: mercury (18 acres), dioxin (18 acres), PCBs (18 acres), estuarine bioassessments (15 acres), and nitrogen (14 acres). The top five reasons for impairment in New Hampshire ocean waters for 2012 were: PCBs (81 acres), mercury (81 acres), dioxin (81 acres), Enterococcus (0.5 acres), and fecal coliform (0.5 acres). Besides atmospheric deposition, sources of impairment in New Hampshire include forced drainage pumping, waterfowl, domestic wastes, combined sewer overflows, animal feeding operations, municipal sources, and other unknown sources (NHDES 2012).

Violation rates among EPA- permitted pollutant sources are relatively low in New Hampshire. A total of 386 (1.7 percent) of 23,192 permitted facilities are in violation of their permits, and only 58 (0.25 percent) of these violations are classified as a significant noncompliance. Of the 254 NPDES permits in New Hampshire, 28 currently have effluent violations and five of these are classified as significant noncompliance.

Massachusetts

In 2012, Massachusetts assessed the condition of 2,816 miles (28 percent) of the state's rivers and streams and found 63 percent to be impaired[1]. Four out of the top five impairment causes for rivers and streams in Massachusetts are attributed to pathogens and nutrients. The probable sources for these impaired waters include unknown sources, municipal discharges and unspecified urban stormwater. The distribution of impairment causes and probable sources suggest that eutrophication is a factor in Massachusetts rivers and stream impairments. PCBs in fish tissue from legacy sediment contamination is identified as a contributing factor in 14 percent of assessed river or stream miles. Both invasive species and atmospheric mercury deposition are

[1] Massachusetts 2014 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=MA

major contributors to impairments of lakes, reservoirs and ponds. Nearly the entire spatial area of Massachusetts' bays and estuaries were assessed (98 percent of 248 square miles), with 87 percent found to be impaired. Fecal coliform contamination from municipal discharges impair the entire extent of assessed bays and estuaries. PCBs in fish tissue are also a significant factor, occurring in 36 percent of assessed waters. The impairment classification "other cause" is identified in 27 percent of estuaries and bays. This reporting category is used for dissolved gases, floating debris and foam, leachate, stormwater pollutants, and many other uncommon causes lumped together. Among sources for pollutants, stormwater was a major factor for Massachusetts estuaries and bays as three of the top five identified sources of impairments are discharges from municipal separate storm sewer systems (53 percent of impaired area), wet weather discharges (27 percent) and unspecified urban stormwater (25 percent). Among the 29,788 discharge-permitted facilities located in Massachusetts, 956 (3 percent) are in violation, with 115 (0.39 percent) of these violations classified as a significant noncompliance. NPDES permits are held by 833 of these facilities. Effluent violations are identified at 77 of these facilities, with 33 violations classified as in significant noncompliance.

The Remaining East Coast

In 2014, the District of Columbia assessed the condition of 98.5 percent of its 39 miles of rivers and streams and 99 percent of its 6 square miles of bays and estuaries[2]. All waters assessed were found to be impaired by PCBs. By impairment group, pesticides accounted for the most causes for impairment for 303(d) listed waters assessed in D.C. The following pesticides were identified as causes for impairment in D.C. rivers/streams and bays/estuaries: heptachlor epoxide (21.9 miles), dieldrin (21.9 miles), chlordane (21.1 miles), DDT (19.4 miles), DDD (16.2 miles), and DDE (16.2 miles). Out of 2,729 facilities with pollutant-source permits in D.C., 48 permits (1.8 percent) are in violation, with three classified as significant noncompliance. Among the twenty-eight NPDES permits in D.C., two had effluent violations (7 percent), but none of the effluent violations were classified as a significant noncompliance.

The remaining East coast portion of the action area is very small. It includes Tribal and federal lands within 24 subwatersheds distributed among Maine, Vermont, Connecticut, and Delaware. Although 13 of these are in Maine, few river and stream aquatic impairments are reported in this state (8 out of 250 total assessed water bodies are impaired). Impairment causes in Maine are identified as low dissolved oxygen and dioxins. Microbial pollution of rivers and streams are indicated as major impairment causes in Vermont, Connecticut and Delaware, accounting for nearly 60 percent of the impaired river and stream miles among these states (EPA Water Quality Assessment and TMDL Information, https://iaspub.epa.gov/waters10/attains_index.home). Mercury, arsenic pollution and "unknown" are also among the top impairment causes for rivers and streams in these states. None of the 35 federally operated permitted facilities in Delaware and Vermont or the six facilities on Tribal land in Connecticut have permit violations (NMFS

[2] District of Columbia 2014 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=DC

2015a). The 9 facilities located in Maine include 5 with violations, 4 of which are classified as a significant noncompliance. There are no NPDES permits for sub-watersheds of Maine or Vermont within the action area. The single NPDES permitted facility in the Delaware portion of the action area is currently in compliance with its permit.

3.1.2 West Coast Region

The West Coast region contains 410 estuaries, bays, and sub-estuaries that cover a total area of 2,200 square miles (USEPA 2015). More than 60 percent of this area consists of three large estuarine systems—the San Francisco Estuary, Columbia River Estuary, and Puget Sound (including the Strait of Juan de Fuca). Sub-estuary systems associated with these large systems make up another 27 percent of the West Coast. The remaining West Coast water bodies, combined, compose only 12 percent of the total coastal area of the region.

The majority of the population in the West Coast states of California, Oregon, and Washington lives in coastal counties. In 2010, approximately 40 million people lived in these coastal counties, representing 19 percent of the U.S. population residing in coastal watershed counties and 63 percent of the total population of West Coast states (U.S. Census Bureau, http://www.census.gov/2010census/). Between 1970 and 2010, the population in the coastal watershed counties of the West Coast region almost doubled, growing from 22 million to 39 million people.

A total of 134 sites were sampled to characterize the condition of West Coast waters. Figure 2 shows a summary of findings from the EPA’s National Coastal Condition Assessment Report for the west Coast Region (USEPA 2015).

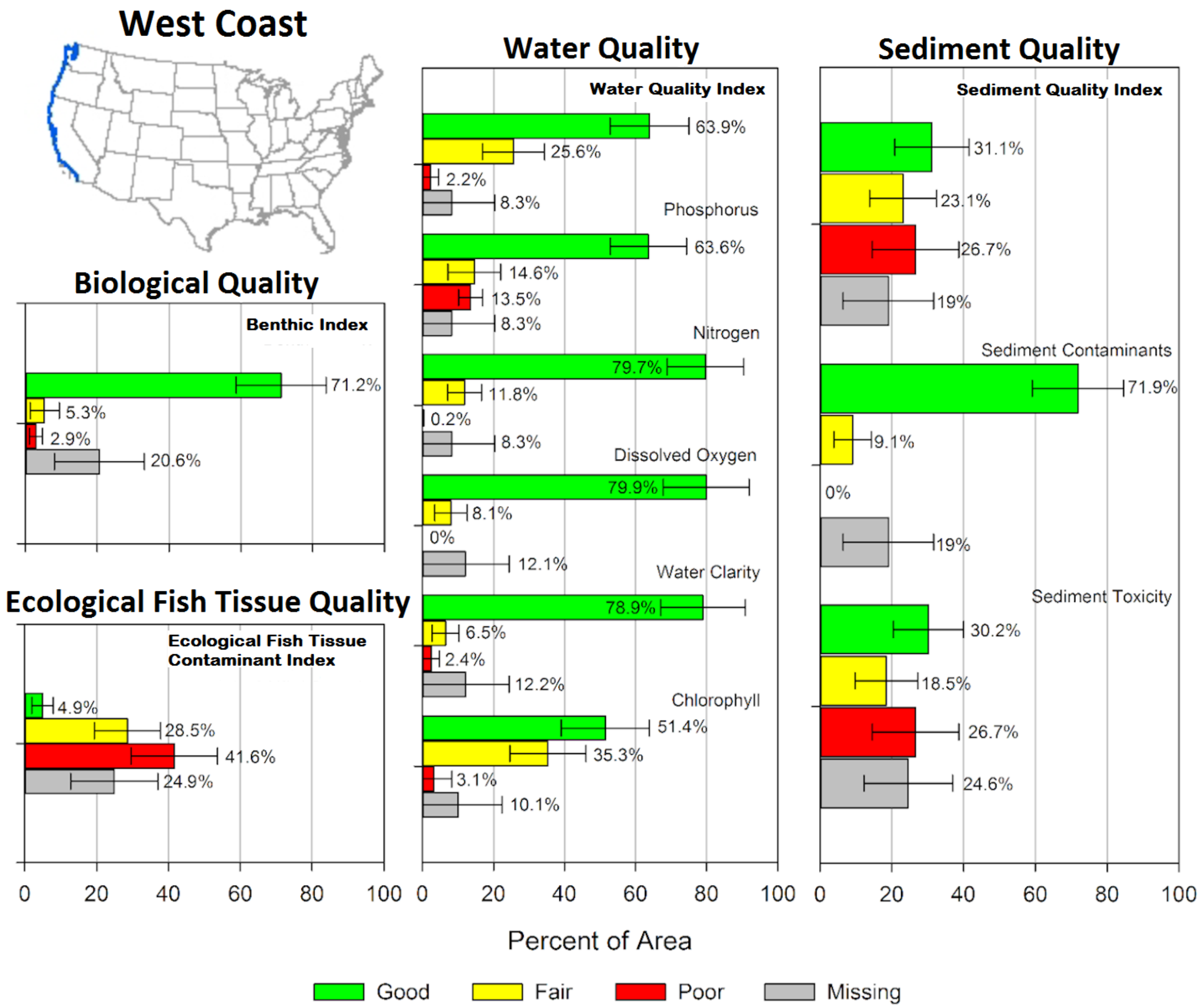

Figure 2. National Coastal Condition Assessment 2010 Report findings for the West Coast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 238 sites sampled). Error bars represent 95 percent confidence levels (USEPA 2015).

Biological quality is rated good in 71 percent of West Coast waters, based on the benthic index. Fair biological quality occurs in 5 percent of these waters, and poor biological quality occurs in 3 percent (data are missing for an additional 21 percent of waters due to difficulty obtaining samples). Based on the water quality index, 64 percent of waters in the West Coast region are in good condition, 26 percent are rated fair, and 2 percent are rated poor (USEPA 2015).

Based on the sediment quality index, 31 percent of West Coast waters sampled are in good condition, 23 percent in fair condition, and 27 percent in poor condition (data missing for 19 percent of waters sampled) (USEPA 2015). Based on the ecological fish tissue contaminant index, 42 percent of West Coast waters are in poor condition, 29 percent in fair condition, and 5 percent in good condition (data missing for 25 percent of waters sampled). The contaminants that most often exceed the thresholds for “poor” condition are selenium, mercury, arsenic, and, in a very small proportion of the area, hexachlorobenzene (USEPA 2015).

Washington

Subwatersheds associated with Washington State federal lands where PGP eligible activities may occur (e.g., Department of Defense, Bureau of Land Management, Bureau of Reclamation) or Tribal lands, are distributed throughout the state and along the coast line. Information from the 2008 state water quality assessment report for the entire state was used to infer conditions within the action area. For the 2008 reporting year, the state of Washington assessed 1,997 miles of rivers and streams, 434,530 acres of lakes, reservoirs, and ponds, and 376 square miles of ocean and near coastal waters (Washington 2008 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=WA). Among assessed waters, 80 percent of rivers and streams, 68 percent of lakes, reservoirs, and ponds, and 53 percent of ocean and near coastal waters were impaired. Temperature (39 percent of assessed waters) and fecal coliform (32 percent of assessed waters) are prominent causes of impairments. These are followed by low dissolved oxygen (19 percent), pH (9 percent), and instream flow impairments (2 percent). Ocean and near coastal impairment causes include fecal coliform in 17 percent of assessed waters, followed by low dissolved oxygen in 12 percent of these waters. The remaining contributors are invasive exotic species, sediment toxicity, and PCBs.

Among the 485 facilities located within Washington’s Tribal lands, 67 are in violation of their permits, with 7 of these violations classified as a significant noncompliance (NMFS 2015a). There are 349 NPDES permits within the action area, but only two of these facilities have effluent violations. There are no violations reported for the 11 EPA permitted facilities within the watersheds associated with federally operated facilities in Washington. Three of these permits are NPDES permits.

Oregon

The area covered by subwatersheds within Tribal lands in Oregon where EPA has permitting authority account for only 1.5 percent of the action area. Direct examination of these areas using EPA’s geospatial databases from 2006 indicate that 80 percent of the 376 km of rivers and streams assessed are impaired by elevated iron (NMFS 2015a). While the source of the iron is not identified, iron contamination can result from acid mine drainage. Eleven out of the 13 assessed lakes, reservoirs, and ponds in subwatersheds associated with these lands are impaired, with causes listed as temperature and fecal coliform bacteria. This amounts to impairment of 93 percent of the assessed area.

California

EPA also has permitting authority for Tribal lands in California. The subwatersheds associated with these lands account for about 6 percent of the total action area, but are dispersed widely and make up a very small fraction of the watersheds within the state. As such, we did not make generalizations about water quality in these areas based on the 2010 statewide water quality assessment report. Rather, information for the relevant watersheds was extracted from EPA Geospatial databases and analyzed separately. Seventy nine percent of the assessed rivers and streams within these Tribal land subwatersheds are impaired by nutrients, aluminum, arsenic, temperature, and chlordane (NMFS 2015a). Stressor sources are attributed to unknown sources, municipal point discharges, agriculture, natural background, and loss of riparian habitat. High impairment rates (93 percent) are also found for assessed lakes, reservoirs and ponds within the action area in California (NMFS 2015a). The predominant impairment for these waters is arsenic, affecting 45 percent of assessed waters, while mercury is a factor in about 9 percent of assessed waters. Arsenic is also the identified cause of impairment in 97 percent of assessed bays and estuaries (NMFS 2015a). Among the 204 facilities located in the California action area, a total of 25 facilities are in violation of their NPDES, Clean Air Act, or Resource Conservation and Recovery Act permit, with 2 of these violations classified as a significant noncompliance. The single NPDES permit listed among these permits is in compliance (NMFS 2015a).

3.1.3 Puerto Rico

Since the listed species under NMFS jurisdiction in Puerto Rico are strictly marine and do not occur in freshwaters or wetlands, this discussion will focus on water quality conditions reported for coastal shoreline and saltwater habitats. In 2014, Puerto Rico assessed the condition of 390 out of 550 miles of coastal shoreline (70.9 percent) and all 8.7 square miles of the surrounding bays and estuaries. The findings indicate that 77 percent of the coastline and 100 percent of the assessed estuaries and bays are impaired[3]. TMDLs are needed in 100 percent of coastal areas sampled but none have been completed. TMDLs are needed in 58.6 percent of bay/estuary areas sampled but are completed for less than 2 percent of assessed areas. Pathogens (e.g., fecal coliform, total coliform, Enterococcus) and pathogen sources dominate the impairment profiles for all three types of assessed waters These include onsite waste water systems, agriculture, concentrated animal feed operations, major municipal point sources, and urban runoff. Coastline impairment causes include pH, turbidity and Enterococcus bacteria. Many of these impairments are attributed to sewage and urban-related stormwater runoff. Rates of noncompliance among EPA-permitted pollution sources are fairly high. Among the 10,077 facilities located in Puerto Rico, 59 percent were in violation of at least one permit in 2012, and nearly all were classified as significant noncompliance. There are 522 facilities with NPDES permits and 84 (16 percent) of these were classified as in significant violation of permit effluent limits as of 2012.

3.1.4 Pacific Islands

The EPA has NPDES permitting authority in the Pacific islands of Guam, the Northern Marianas, and American Samoa. Because the listed species under NMFS jurisdiction in these areas are strictly marine and do not occur in freshwaters or wetlands, this discussion will focus on water quality conditions reported for coastal shoreline and saltwater habitats.

The population of American Samoa was 55,519 in 2010. Factors such as population density, inadequate land-use permitting, and increased production of solid waste and sewage, have detrimentally impacted water quality in streams and coastal waters of this U.S. territory. The

[3] Puerto Rico Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_index.control?p_area=PR#total_assessed_waters

total surface area of American Samoa is very small, only 76.1 sq. miles, which is divided into 41 watersheds with an average size of 1.8 sq. miles. Water quality monitoring, along with coral and fish benthic monitoring, covers 34 of the 41 watersheds, which includes areas populated by more than 95 percent of the total population of American Samoa. For the goal Protect and Enhance Ecosystems (Aquatic Life), of the 45.1 shoreline miles (out of 149.5 total) assessed in 2012-2013, 15.5 miles were found to be fully supporting, 12.8 miles were found to be partially supporting, and 16.8 miles were found to be not supporting (Tuitele et al. 2014). For the goal to Protect and Enhance Public Health, all 7.9 shoreline miles assessed in 2012-2013 for fish consumption were found to be not supporting. Eighty four percent of American Samoa's coastline was assessed in 2010 and 60 percent of the assessed waters were found to be impaired. Enterococcus is identified as causing impairments along 50 percent of the coastline evaluated, while 26 percent of assessed coastline had nonpoint source pollutants contributing to impairments. Of the 5.7 km^2 of reef flats assessed in 2010, 76 percent were fully supporting and 24 percent were not supporting the goal of Protect and Enhance Ecosystems (Tuitele et al. 2014). The major stressors identified were PCBs, metals (mercury), pathogen indicators, and other undetermined stressors (Tuitele et al. 2014). The major sources of impairment included sanitary sewer overflows and animal feed operations, each implicated for 50 percent of the waters assessed. Multiple nonpoint sources were identified as a stressor source for 26 percent of assessed waters, while contaminated sediments contributed to impairments in 6 percent of assessed waters. Among the 204 facilities with pollutant permits, a total of 21 (10.3 percent) facilities were in violation, with 17 of these violations classified as a significant noncompliance. Of the six facilities with NDPES permits, two have violated effluent limits, one of which is considered to be in significant noncompliance.

Guam assessed 3 percent of its 915 acres of bays/estuaries and 14 percent of its 117 miles of coastline in 2010[4]. Impairments are identified in 42 percent of assessed bays and estuaries and the entire extent of assessed coastline. PCBs levels in fish tissue was the cause of impairment in 33 percent of assessed bays and estuaries, followed by antimony, dieldrin, tetrachloroethylene, and trichloroethylene, each listed as causing impairments to 6 percent of assessed waters. Enterococcus bacteria is the cause of impairment in nearly all of Guam's coastal shoreline waters (96 percent), while PCB contamination is a minor contributor to impairment of the coastal shoreline (4 percent). Sources of impairment causes have not been identified for Guam. Among the 403 NPDES, Clean Air Act, or Resource Conservation and Recovery Act EPA-permitted facilities located in Guam, a total of 23 (5.7 percent) facilities are in violation, with 13 of these violations classified as a significant noncompliance. NPDES permits are held by 19 facilities, six of which have effluent violations classified as significant noncompliance.

In the Northern Marianas, 36 percent of the 235.5 miles of assessed shoreline were found to be impaired in 2014[5]. Phosphate is listed as a cause for all impaired areas. Other causes identified

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[4] Guam 2010 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=GU
[5] N. Mariana Islands Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=CN

among the impaired stretches of shoreline include microbiological contamination from Enterococcus bacteria (22 percent), dissolved oxygen saturation levels (16 percent), and mercury in fish tissue (1 percent). The presence of Enterococci bacteria was implicated for the impairment of 32.2 miles of Saipan's, 17.8 miles of Rota's, and 24.3 miles of Tinian's shoreline for recreational uses. In addition 15 percent of the assessed waters had impaired biological assemblages. Sources of impairments included sediments (15 percent), unknown sources (13 percent), on-site septic treatment systems (12 percent), urban runoff (12 percent), and livestock operations (7 percent). We did not find any NPDES permitted facilities in the Northern Marianas.

3.2 Baseline Pesticide Detections in Aquatic Environments

Pesticide detections for the environmental baseline are addressed as reported in the U.S. Geological Survey (USGS) National Water-Quality Assessment Program's (NAWQA) national assessment (Gilliom 2006). This approach was chosen because the NAWQA reports provide the same level of analysis for each geographic area. In addition, given the lack of uniform reporting standards and large action area for this opinion, it is not feasible to present a comprehensive basin-specific analysis of pesticide detections.

Over half a billion pounds of herbicides, insecticides, and fungicides were used annually from 1992 to 2011 to increase crop production and reduce insect-borne disease (Stone et al. 2014) During any given year, more than 400 different types of pesticides are used in agricultural and urban settings. The distributions of the most prevalent pesticides in streams and groundwater correlate with land use patterns and associated present or past pesticide use (Gilliom 2006). When pesticides are released into the environment they frequently end up as contaminants in aquatic environments. Depending on their physical properties, some are rapidly transformed via chemical, photochemical, and biologically mediated reactions into other compounds known as degradates. These degradates may become as prevalent as the parent pesticides depending on their rate of formation and their relative persistence. Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (Gilliom 2006). Mixtures result from the use of different pesticides for multiple purposes within a watershed or groundwater recharge area. Pesticides generally occur more often in natural water bodies as mixtures than as individual compounds. Fish exposed to multiple pesticides at once may also experience additive and synergistic effects. If the effects on a biological endpoint from concurrent exposure to multiple pesticides can be predicted by adding the potency of the pesticides involved, the effects are said to be additive. If, however, the response to a mixture leads to a greater than expected effect on the endpoint, and the pesticides within the mixture enhance the toxicity of one another, the effects are characterized as synergistic. These effects are of particular concern when the pesticides share a mode of action.

From 1992 to 2001, the USGS sampled water from 186 stream sites, bed sediment samples from 1,052 stream sites, and fish from 700 stream sites across the continental U.S. Pesticide concentrations were detected in streams and groundwater within most areas sampled with substantial agricultural or urban land uses. NAWQA results detected at least one pesticide or degradate in more than 90 percent of water samples, more than 80 percent of fish samples, and more than 50 percent of bed sediment samples from streams in watersheds with agricultural, urban, and mixed land use (Gilliom 2006). Compounds commonly detected included 11 agriculture-use herbicides and the atrazine degradate deethylatrazine; 7 urban-use herbicides; and 6 insecticides used in both agricultural and urban areas. Mixtures of pesticides were detected more often in streams than in ground water and at relatively similar frequencies in streams draining areas of agricultural, urban, and mixed land use. Water from streams in these developed land use settings had detections of two or more pesticides or degradates more than 90 percent of the time, five or more pesticides or degradates about 70 percent of the time, and 10 or more pesticides or degradates about 20 percent of the time (Gilliom 2006). NAWQA analysis of all detections indicates that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams (Gilliom 2006). The number of unique mixtures varied with land use. More than half of all agricultural streams and more than three-quarters of all urban streams sampled had concentrations of pesticides in water that exceeded one or more benchmarks for aquatic life. Exceedance of an aquatic life benchmark level indicates a strong probability that aquatic species are being adversely affected. However, aquatic species may also be affected at levels below benchmark criteria. In agricultural streams, most concentrations that exceeded an aquatic life benchmark involved chlorpyrifos (21 percent), azinphos methyl (19 percent), atrazine (18 percent), DDE (16 percent), and alachlor (15 percent) (Gilliom 2006). Organochlorine pesticides that were discontinued 15 to 30 years ago still exceeded benchmarks for aquatic life and fish-eating wildlife in bed sediment or fish tissue samples from many streams.

Stone et al. (2014) compared pesticide levels for streams and rivers across the conterminous U.S. for the decade 2002−2011 with previously reported findings from the decade of 1992−2001. Overall, the proportions of assessed streams with one or more pesticides that exceeded an aquatic life benchmark were very similar between the two decades for agricultural (69 percent during 1992−2001 compared to 61 percent during 2002−2011) and mixed-land-use streams (45 percent compared to 46 percent). Urban streams, in contrast, increased from 53 percent during 1992−2011 to 90 percent during 2002−2011, largely because of fipronil and dichlorvos. Agricultural use of synthetic organic herbicides, insecticides, and fungicides in the continental U.S. had a peak in the mid-1990s, followed by a decline to a low in the mid-2000s (Stone et al. 2014). During the late-2000s, overall pesticide use steadily increased, largely because of the rapid adoption of genetically modified crops and the increased use of glyphosate. The herbicides that were assessed by USGS represent a decreasing proportion of total use from 1992 to 2011 because glyphosate was not previously included in the national monitoring network.

3.2.1 ESA Section 7 Consultations

EPA has consulted with NMFS under Section 7(a)(2) of the ESA on the registration of several pesticides on the West Coast (NMFS Pesticide Consultations with EPA, http://www.nmfs.noaa.gov/pr/consultation/pesticides.htm). In a 2008 biological opinion NMFS concluded that current use of chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 listed salmonid ESUs/DPSs. In 2009, NMFS further determined that the current use of carbaryl and carbofuran is likely to jeopardize the continued existence of 22 ESUs/DPSs; and the current use of methomyl is likely to jeopardize the continued existence of 18 ESUs/DPSs of listed salmonids. In 2010 NMFS issued a biological opinion that concluded pesticide products containing azinphos methyl, disulfoton, fenamiphos, methamidophos, or methyl parathion are not likely to jeopardize the continuing existence of any listed Pacific salmon or destroy or adversely modify designated critical habitat. NMFS also concluded that the effects of products containing bensulide, dimethoate, ethoprop, methidathion, naled, phorate, or phosmet are likely to jeopardize the continued existence of some listed Pacific salmonids and to destroy or adversely modify designated habitat of some listed salmonids. In 2011, NMFS issued a biological opinion on the effects of four herbicides and two fungicides. NMFS concluded that products containing 2,4-D are likely to jeopardize the existence of all listed salmonids, and adversely modify or destroy the critical habitat of some of these ESUs and DPSs. Products containing chlorothalonil or diuron were also likely to adversely modify or destroy critical habitat, but not likely to jeopardize listed salmonids. NMFS also concluded that products containing captan, linuron, or triclopyr BEE do not jeopardize the continued existence of any ESUs/DPSs of listed Pacific salmonids or adversely modify designated critical habitat. In 2012 NMFS issued an opinion on oryzalin, pendimethalin, and trifluralin that concluded each of these chemicals are likely to jeopardize the continued existence of some listed Pacific salmonids, and adversely modify designated critical habitat of some listed salmonids. Also in 2012, NMFS concluded EPA's proposed registration of thiobencarb, an herbicide authorized for use in California only on rice, is not likely to jeopardize the continued existence or adversely modify the designated critical habitat of listed Pacific salmonid species. Finally, in 2015 NMFS concluded that the EPAs proposed registration of the pesticide active ingredient diflubenzuron is likely to jeopardize the continued existence of 23 ESA-listed Pacific salmonid species and is likely to destroy or adversely modify designated critical habitat of 23 listed Pacific salmonids. Also in this opinion, NMFS found that the active ingredients fenbutatin oxide and propargite are each likely to jeopardize the continued existence and likely to destroy or adversely modify designated critical habitat of 21 ESA-listed Pacific salmonid species.

3.3 Hydromodification

Hydromodification is generally defined as a change in natural channel form, watershed hydrologic processes and runoff characteristics (i.e., interception, infiltration, overland flow, interflow and groundwater flow) associated with alterations in stream and rivers flows and sediment transport due to anthropogenic activities. Such changes often result in negative impacts to water quality, quantity, and aquatic habitats.

3.3.1 Dams

While dams provide valuable services to the public, such as recreation, flood control, and hydropower, they also have detrimental impacts on aquatic ecosystems. Dams can have profound effects on anadromous species by impeding access to spawning and foraging habitat and altering natural river hydrology and geomorphology, water temperature regimes, and sediment and debris transport processes (Pejchar and Warner 2001, Wheaton et al. 2004). The loss of historic habitat ultimately affects anadromous fish in two ways: 1) it forces fish to spawn in sub-optimal habitats that can lead to reduced reproductive success and recruitment, and 2) it reduces the carrying capacity (physically) of these species and affects the overall health of the ecosystem (Patrick 2005). Additionally, a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delay in passage, exposure to high concentrations of dissolved gases, elevated water temperatures, and increased predation.

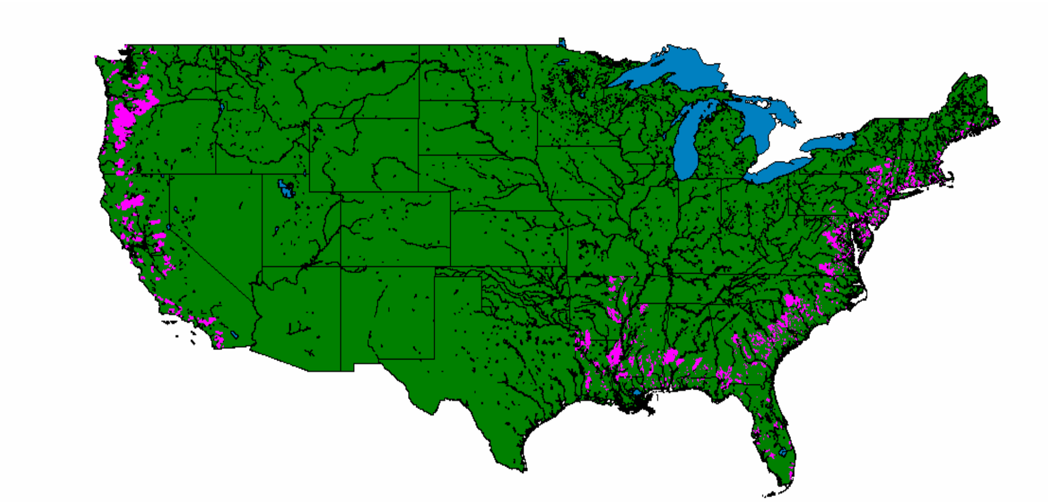

Figure 3. Map of River and Lake Habitat Impeded by Dams (Denoted in Purple) for the Continental U.S. (modified from Patrick 2005).

Nationwide, nearly 44,000 miles of river and lake habitat are blocked by terminal dams (those lowest in the watershed), which includes the area between the terminal dam and the next upstream impediment. This loss of habitat represents approximately 8.5 percent and 4.7 percent of the total riverine miles available (637,525 miles) along the Atlantic/Gulf Coast and Pacific Coast, respectively (Patrick 2005). Based on a non-random sample of dams affecting the largest areas (east and west coast) with diadromous fish runs, nearly 30 percent of diadromous fish habitat is blocked by terminal dams that have no fish passage (Patrick 2005).

The final rule listing Southern DPS green sturgeon indicates that the principle factor for the decline of this DPS is the reduction of spawning to a limited area, due largely to impassable barriers on the Sacramento River (Keswick Dam) and the Feather River (Oroville Dam) (71 FR 17757; April 7, 2006).

Comparative analyses of historic and contemporary hydrologic and thermal regimes indicate that aquatic habitats in the Sacramento, Yuba, and Feather rivers are different than they were before dam construction (NMFS 2015b). However, the impact of these changes on Southern DPS green sturgeon spawning and recruitment is not fully understood. (Mora et al. 2009) suggest that flow regulation has had mixed effects on habitat suitability. In the Sacramento River the removal of Red Bluff Diversion Dam as a barrier to migration has increased the use of upstream spawning habitat by Southern DPS green sturgeon. Modeling studies predict that Southern DPS green sturgeon would use additional areas on the Sacramento River in the absence of impassable dams (Mora et al. 2009). This modeling work also found that suitable spawning habitat historically existed on portions of the San Joaquin, lower Feather, American, and Yuba rivers, much of which is currently inaccessible to green sturgeon due to the presence of barriers. Flood bypass systems along the Sacramento River pose a challenge to Southern DPS green sturgeon during spawning migrations. Green sturgeon are particularly affected at the Yolo and Sutter bypasses and by Tisdale and Fremont weirs (Thomas et al. 2013).

3.3.2 Pacific Northwest Dams

There are more than 400 dams in the Pacific Northwest, ranging from mega dams that store large amounts of water to small diversion dams for irrigation (Panel on Economic Environmental and Social Outcomes of Dam Removal 2001). Every major tributary of the Columbia River, except the Salmon River, is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects which provide a significant source of power to the region. Of these, 18 dams are located on the mainstem Columbia River and its major tributary, the Snake River. Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on ecosystems within the Columbia River Basin, particularly the survival of anadromous salmonids (Williams et al. 1999). Approximately 80 percent of historical spawning and rearing habitat of Snake River fall-run Chinook salmon is now inaccessible due to dams. The Snake River spring/summer run has been limited to the Salmon, Grande Ronde, Imnaha, and Tuscanon rivers. Dams have cut off access to the majority of Snake River Chinook salmon spawning habitat. The Sunbeam Dam on the Salmon River is believed to have limited the range of Snake River sockeye salmon as well. Non-federal hydropower facilities on Columbia River tributaries have also partially or completely blocked higher elevation spawning (NMFS 2015b).

The Puget Sound region, which includes the San Juan Islands and south to Olympia is the second largest estuary in the U.S. and is fed by over 10,000 rivers and streams. More than 20 dams occur within this region's rivers and overlap with the distribution of salmonids. Dams were built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

Compared to other parts of the Northwest Region, the Oregon-Washington-Northern California coastal drainages are less impacted by dams and still have several remaining free flowing rivers.. Dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat (Palmisano et al. 1993 cited in NMFS, 2015). In the past, temporary splash dams were constructed throughout the region to transport logs out of mountainous reaches. Thousands of splash dams were constructed across the Northwest in the late 1800s and early 1900s. While these dams typically only temporarily blocked salmon habitat, in some cases dams remained long enough to wipe out entire salmon runs. The effects of the channel scouring and loss of channel complexity from splash dams also resulted in the long-term loss of salmon habitat (Salmonids 1996)

Several hydromodification projects in the Pacific Northwest have been designed to improve the productivity of listed salmonids. Improvements include flow augmentation to enhance water flows through the lower Snake and Columbia Rivers; providing stable outflows at Hells Canyon Dam during the fall Chinook salmon spawning season and maintaining these flows as minimums throughout the incubation period to enhance survival of incubating fall-run Chinook salmon; and reduced summer temperatures and enhanced summer flow in the lower Snake River (Corps et al. 2007, Appendix 1 cited in NMFS, 2008). Providing suitable water temperatures for over-summer rearing within the Snake River reservoirs allows the expression of productive “yearling” life history strategy that was previously unavailable to Snake River Fall-run Chinook salmon. The mainstem Federal Columbia River Power System corridor has also improved safe passage through the hydrosystem for juvenile steelhead and yearling Chinook salmon with the construction and operation of surface bypass routes at Lower Granite, Ice Harbor, and Bonneville dams and other configuration improvements. For salmon, with a stream-type juvenile life history, projects that have protected or restored riparian areas and breached or lowered dikes and levees in the tidally influenced zone of the estuary have improved the function of the juvenile migration corridor. The Federal Columbia River Power System action agencies recently implemented 18 estuary habitat projects that removed passage barriers to increase fish access to high quality habitat. The Army Corps estimates that hydropower configuration and operational improvements implemented from 2000 to 2006 resulted in an 11.3 percent increase in survival of yearling juvenile Lower Columbia River Chinook salmon from populations that pass Bonneville Dam.

Obstructed fish passage and degraded habitat caused by dams is considered the greatest impediment to self-sustaining anadromous fish populations in Maine (NRC 2004). Gulf of Maine DPS Atlantic salmon are not well adapted to the artificially created and maintained impoundments resulting from dam construction (NRC 2004). Other aquatic species that thrive in impounded riverine habitat have proliferated and significantly altered the prey resources available to salmon, as well as the abundance and species composition of salmon competitors and predators. The National Inventory of Dams Program lists 639 dams (over four feet high) in Maine, over half of which are located within the range of the Gulf of Maine DPS (USACOE National Inventory of Dams Program, http://nid.usace.army.mil/cm_apex/f?p=838:12). The larger hydroelectric dams and storage projects within the Gulf of Maine DPS are primarily located in the Penobscot, Kennebec, and Androscoggin watersheds. Gulf of Maine DPS salmon habitat is also degraded as a result of bypassed reaches of natural river channels that re-route river flows through forebays or penstocks. Many smaller dams still remain on smaller rivers and streams within Gulf of Maine DPS range.

3.3.3 East Coast Dams

The prevalence of dams throughout East Coast rivers means that all Atlantic sturgeon life stages generally occur downstream of dams, leaving them vulnerable to perturbations of natural river conditions. Atlantic sturgeon spawning sites remain unknown for the majority of rivers in their range. However, they have been observed spawning hundreds of miles upstream in Southern non-tidal rivers that are unobstructed by dams, suggesting that dams may prevent them from reaching preferred spawning areas. Observations of Atlantic sturgeon spawning immediately below dams, further suggests that they are unable to reach their preferred spawning habitat upriver. Overall, 91 percent of historic Atlantic sturgeon habitat seems to be accessible, but the quality of the remaining portions of habitat as spawning and nursery grounds is unknown, therefore estimates of percentages of availability do not necessarily equate to functionality (ASSRT 2007). Access to 50 percent or more of historical sturgeon spawning habitat have been eliminated or restricted. Thus, dams may one of the primary causes of the extirpation of several Atlantic sturgeon subpopulations.

Due to their upriver locations, most dams in the Chesapeake Bay watershed have large freshwater tailways (unobstructed habitat downstream of the dam). Several dams within the Atlantic sturgeon historic range have been removed or naturally breached.Sturgeon appear unable to use some fishways (e.g., ladders) but have been transported in fish lifts (Kynard 1998). Data on the effects of the fish lift at the Holyoke Hydroelectric Project on the Connecticut River suggest that fish lifts that successfully attract other anadromous species (i.e., shad, salmon etc.) do a poor job of attracting sturgeon: attraction and lifting efficiencies for shortnose sturgeon at the Holyoke Project are estimated around 11 percent (ASSRT 2007). Despite decades of effort, fish passage infrastructure retrofitted at hydroelectric dams has largely failed to restore diadromous fish to historical spawning habitat (Brown et al. 2013). While improvements to fish passage are often required when hydroelectric dams go through Federal Energy Regulatory Commission relicensing, the relicensing process occurs infrequently, with some licenses lasting up to 50 years. Over 95 percent of dams on the eastern seaboard are not hydroelectric facilities and are thus not subject to continual relicensing or fish passage improvement measures (ASMFC 2008).

3.3.4 Water Diversions

Like many regions throughout the world, the U.S. is experiencing increasing demand for fresh, clean water. Increasing population growth and agricultural needs frequently conflict with water availability. The twentieth century saw increased dam construction, increased irrigation practices for agriculture, increased recreational use of waterbodies, and increased use of waterways for waste disposal, both sanitary and industrial. Water use in the western U.S. presents a particular concern because the western states are characterized by low precipitation and extended periods of draught. Moreover, agricultural uses dominate the water needs in these states (Anderson and

Woosley 2008). Although the western states contain the headwaters of some of the continent's major river systems, these water sources have been utilized to the point that there are few undeveloped resources to draw upon to satisfy new demands or to restore depleted rivers and aquifers (USACE and CBI 2012). Groundwater has become an increasingly important source of water as surface water resources have been depleted. Water remains a finite resource, however, and there are consequences to pumping ground water including depleting aquifer storage, supplying poorer quality water to wells, diminishing flow to springs and streams, and land subsidence (Anderson and Woosley 2008).

The amount and extent of water withdrawals or diversions for agriculture impacts streams and their inhabitants by reducing water flow/velocity and dissolved oxygen levels, which can have negative effects on listed species and their designated critical habitat. Water diversions and withdrawals for agricultural irrigation or other purposes can directly impact fish populations by constraining available spawning and rearing habitat. Adequate water quantity and quality are critical to all salmonid life stages, especially adult migration and spawning, fry emergence, and smolt emigration. Low flow events may delay salmonid migration or lengthen fish presence in a particular water body until favorable flow conditions permit fish migration along the migratory corridor or into the open ocean. Survival of eggs, fry, and juveniles are also mediated by streamflow. Water withdrawals may dewater redds thus reducing egg survival. During summer and winter, the two periods of low flow annually, juvenile salmon survival is directly related to discharge, with better survival in years with higher flows during these two seasons (Gibson 1993, Ghent and Hanna 1999). Summer water withdrawals have the potential to limit carrying capacity and reduce parr survival.

Other potential detrimental impacts of water diversions include increases in nutrient loading, sediments (from bank erosion), and water temperature. Flow management, in combination with the effects of climate change (i.e., droughts), has further decreased the delivery of suspended particulate matter and fine sediment to estuaries. Low river flows may constrain conditions necessary for important salmonid refuge habitat (shade, woody debris, overhanging vegetation), making fish more vulnerable to predation, elevated temperatures, crowding, and disease. In addition, some listed fish species have been shown to be susceptible to entrainment through unscreened diversion pipes. Although many diversion pipes are now screend, the effectiveness of screening for green sturgeon requires further study given that screen criteria were designed to reduce salmon entrainment and impingement. Thousands of diversions exist in the Sacramento River and Delta that could potentially entrain Southern DPS green sturgeon (Mussen et al. 2014). By the early 1900s, agricultural opportunities within the Columbia River basin began increasing rapidly with the creation of more irrigation canals and the passage of the Reclamation Act of 1902. Today, agriculture represents the largest water user within the basin (>90 percent). Approximately 6 percent of the annual flow from the Columbia River is diverted for the irrigation of over seven million acres of croplands (Hinck et al. 2004). The vast majority of these agricultural lands are located along the lower Columbia River, the Willamette, Yakima, Hood, and Snake rivers, and the Columbia Plateau.

In general, the southern basins in California have a warmer and drier climate while the more northern, coastal-influenced basins are cooler and wetter. About 75 percent of the runoff occurs in basins in the northern third of the state (north of Sacramento), while 80 percent of the demand occurs in the southern two-thirds of the state. Two major water diversion projects meet these demands—the federal Central Valley Project and the California State Water Project. Combined these two water storage and transport systems irrigate about four million acres of farmland and deliver drinking water to roughly 22 million residents.

Water withdrawal may also impact Gulf of Maine DPS Atlantic salmon habitat in the main stem areas of the Penobscot, Kennebec, and Androscoggin Rivers including headwater areas and tributaries of these watersheds (Fay et al. 2006). There are a variety of consumptive water uses in these large watersheds including municipal water supplies, snow making, mills, golf course and agricultural irrigation, and industrial cooling. Increased levels of agricultural irrigation have been occurring throughout the range of the Gulf of Maine DPS for several years. Approximately 6,000 acres of blueberries are irrigated annually with water withdrawn from Pleasant, Narraguagus, and Machias river watersheds (Fay et al. 2006).

3.3.5 Dredging

Riverine, nearshore, and offshore coastal areas are often dredged to support commercial shipping, recreational boating, construction of infrastructure, and marine mining. Dredging in spawning and nursery grounds modifies habitat quality, and limits the extent of available habitat in some rivers where anadromous fish habitat has already been impacted by the presence of dams. Negative indirect effects of dredging include changes in dissolved oxygen and salinity gradients in and around dredged channels ((Jenkins et al. 1993, Secor and Niklitschek 2001, Campbell and Goodman 2004). Dredging operations may also pose risks to anadromous fish species by destroying or adversely modifying benthic feeding areas, disrupting spawning migrations, and filling spawning habitat with resuspended fine sediments. As benthic omnivores, sturgeon in particular may be sensitive to modifications of the benthos which affect the quality, quantity and availability of prey species.

Dredging and filling impact important habitat features of Atlantic sturgeon as they disturb benthic fauna, eliminate deep holes, and alter rock substrates (Smith and Clugston 1997). (Hatin et al. 2007) reported avoidance behavior by Atlantic sturgeon during dredging operations. Dredging operations are also capable of destroying macroalgal beds that may be used as Nassau grouper nursery areas. The eulachon biological review team identified dredging as a low to moderate threat to the species in the Fraser and Columbia rivers, and a low threat in mainland British Columbia rivers due to less dredging activity there (FR 75 13012). They noted that dredging during eulachon spawning was particularly detrimental, as eggs associated with benthic substrates are likely to be destroyed. In addition to indirect impacts, hydraulic dredging can directly harm listed fish species by lethally entraining fish up through the dredge drag-arms and impeller pumps. Atlantic sturgeon have been reported as taken in hydraulic pipeline and bucketand-barge operations (Moser and Ross 1995), mechanical dredges (i.e., clamshell) (Hastings 1983), and hopper dredges (Dickerson 2006).

Dredging and filling activities can adversely affect colonies of reef-building organisms by burying them, releasing contaminants such as hydrocarbons into the water column, reducing light penetration through the water, and increasing the level of suspended particles in the water column. Corals are sensitive to even slight reductions in light penetration or increases in suspended particulates, and the adverse effects of such activities lead to a loss of productive coral colonies. Among corals, Atlantic Acropora species are considered to be particularly environmentally sensitive, requiring relatively clear, well-circulated water (Jaap 1989). Acropora spp. are almost entirely dependent upon sunlight for nourishment compared to massive, boulder-shaped species in the region, with these latter types of corals more dependent on zooplankton (Porter 1976). Thus, Acropora are considered more susceptible to increases in water turbidity and reductions in water clarity that can result from dredging operations.

3.4 Mining

Mining operations can negatively impact aquatic ecosystems and decrease the viability of threatened and endangered fish populations. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations. Extraction methods such as suction dredging, hydraulic mining, and strip mining may cause water pollution problems and increased levels of harmful contaminants. Metal contamination reduces the biological productivity within a basin. Metal contamination can result in fish kills at high levels or sublethal effects at low levels, including reduced feeding, activity level, and growth. Sand and gravel mined from riverbeds (gravel bars and floodplains) may result in substantial changes in channel elevation and patterns, in-stream sediment loads, and in-stream habitat conditions. In some cases, in-stream or floodplain mining has resulted in large-scale river avulsions.

California has a long history of mining that dates back to the Gold Rush of the mid-1800s. The Sacramento Basin and the San Francisco Bay watershed are two of the most heavily impacted basins from mining activities. The Iron Metal Mine in the Sacramento Basin releases large quantities of copper, zinc, and lead into the Keswick Reservoir below Shasta Dam (Cain et al. 2000). Methyl mercury contamination remains a persistent problem within San Francisco Bay (Conaway et al. 2003). Many of the streams and river reaches in the Pacific Northwest are impaired from mining. Metal mining (zinc, copper, lead, silver, and gold) peaked in Washington state between 1940 and 1970 (Palmisano et al. 1993 cited in NMFS, 2015). Several abandoned and former mining sites are designated as Superfund cleanup areas (Benke and Cushing 2011). An estimated 200 abandoned mines within the Columbia River Basin pose a potential hazard to the environment due elevated levels of lead and other trace metals (Quigley 1997 cited in Hinck, 2004).

3.5 Population Growth, Development and Land Use Changes

The 2010 Census reported 308.7 million people in the U.S., a 9.7 percent increase from the 2000 Census population of 281.4 million (U.S. Census Bureau, www.census.gov). From 2000 to 2010, regional growth was much faster in the South and West (14.3 and 13.8 percent, respectively) than in the Midwest (3.9 percent) and Northeast (3.2 percent). Puerto Rico's population declined by 2.2 percent from 2000 to 2010. Several coastal states within the action area experienced faster growth than the nation as a whole including Oregon (12 percent), Washington (14.1 percent), Delaware (14.5 percent), and Virginia (13.0 percent). Population trends by state and decade from 1980 to 2010 are shown in Figure 4. Some of the highest population densities in the U.S. are found in coastal counties within the action area, particularly in central and southern California, Washington, and Massachusetts through New Jersey.

Many stream and riparian areas within the action area have been degraded by the effects of land and water use resulting from urbanization, road construction, forest management, agriculture, mining, transportation, and water development. Development activities have contributed to many interrelated factors causing the decline of listed anadromous fish species considered in this opinion. These include reduced in- and off-channel habitat, restricted lateral channel movement, increased flow velocities, increased erosion, decreased cover, reduced prey sources, increased contaminants, increased water temperatures, degraded water quality, and decreased water quantity.

Urbanization and increased human population density within a watershed result in changes in stream habitat, water chemistry, and the biota (plants and animals) that live there. The most obvious effect of urbanization is the loss of natural vegetation which results in an increase in impervious cover and dramatic changes to the natural hydrology of urban and suburban streams. Urbanization generally results in land clearing, soil compaction, modification and/or loss of riparian buffers, and modifications to natural drainage features. The increased impervious cover in urban areas leads to increased volumes of runoff, increased peak flows and flow duration, and greater stream velocity during storm events. Runoff from urban areas also contains chemical pollutants from vehicles and roads, industrial sources, and residential sources. Urban runoff is typically warmer than receiving waters and can significantly increase temperatures in small urban streams. Wastewater treatment plants replace septic systems, resulting in point discharges of nutrients and other contaminants not removed in the processing. Additionally, some cities have combined sewer/stormwater overflows and older systems may discharge untreated sewage following heavy rainstorms. These urban nonpoint and point source discharges affect the water quality and quantity in basin surface waters. Dikes and levees constructed to protect infrastructure and agriculture have isolated floodplains from their river channels and restricted fish access. The many miles of roads and rail lines that parallel streams with the action area have degraded stream bank conditions and decreased floodplain connectivity by adding fill to floodplains. Culvert and bridge stream crossings have similar effects and create additional problems for fish when they act as physical or hydraulic barriers that prevent fish access to spawning or rearing habitat, or contribute to adverse stream morphological changes upstream and downstream of the crossing itself.

3.5.1 USGS Land Cover Trends Project

The USGS Land Cover Trends Project (http://landcovertrends.usgs.gov/) was a research project focused on understanding the rates, trends, causes, and consequences of contemporary U.S. land use and land cover change. The project spanned from 1999 to 2011, producing statistical and geographic summaries of land cover change using time series land cover data. The project was designed to document the types and rates, causes, and consequences of land cover change from 1973 to 2000 within 84 ecoregions, as defined by EPA, that span the conterminous U.S.. Research objectives of this project were as follows:

- Develop a comprehensive methodology using sampling, change analysis techniques, and Landsat Multispectral Scanner and Thematic Mapper data for estimating regional land cover change.
- Characterize the spatial and temporal characteristics of conterminous U.S. land cover change for five periods from 1973-2000 (1973, 1980, 1986, 1992, and 2000).
- Document the regional driving forces and consequences of change.
- Prepare a national synthesis of land cover change.

For this opinion we summarized the results of the Land Cover Trends Project for project areas that overlap with PGP coverage. The Northeastern coastal zone covers approximately 37,158 km2 in eight states (Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, Connecticut, New York, and New Jersey). Primary land-cover classes are forests and developed land which account for more than 70 percent of the ecoregion. Water, wetlands, and agriculture are secondary land covers classes found in smaller, less frequent concentrations in the Northeast coastal zone. Developed land increased an estimated 4 percent (1,510 km2) from 1973 to 2000, to approximately 27 percent of the ecoregion's area. Much of the new development came from forest loss, with a decrease of 3.7 percent (1,361 km2) during this same time period. Agricultural land-cover decreased by 0.8 percent. Other land cover changes in the Northeastern coastal zone from 1973 to 2000 included slight decreases in wetlands and slight increases in mechanically disturbed lands and mining.

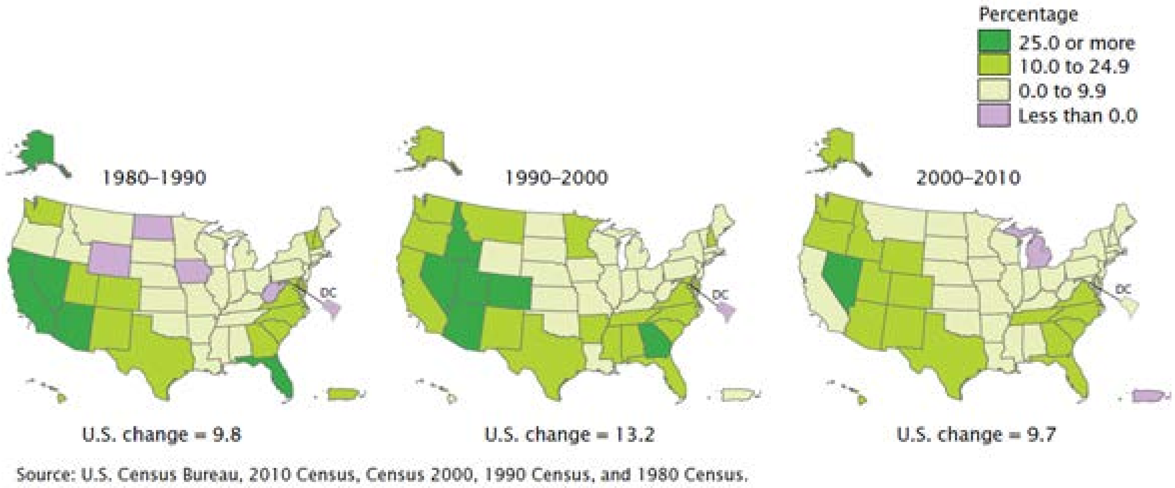

Figure 4. Percentage Change in Population by State and Decade from 1980 to 2010 (Source: U.S. Census Bureau)

The Puget lowland ecoregion is located in western Washington State and covers an area of approximately 17,541 km² (Omernik 1987). Puget Sound is in the center of the ecoregion, which is bordered on the west by the Olympic Mountains and on the east by the Cascade Mountains. The dominant land-cover class in 2000 for Puget lowland was forest (48.4 percent), followed by developed (19.3 percent), agriculture (10.6 percent), and water (10.6 percent). Puget lowland experienced one of the highest percentages of land use change of any ecoregion nationwide from 1973 to 2000. The largest net change for any land-cover class between 1973 and 2000 was the loss of 1,767 km² of forest, which is 10 percent of the land area of the ecoregion. Agriculture decreased by 0.7 percent during this period, while developed land increased by 6.7 percent or 1,186 km².

The Willamette Valley ecoregion covers approximately 14,400 km² and includes the Willamette River watershed, with headwaters in the Cascades draining northward into the Columbia River near the ecoregion's northern boundary in Washington State (Omernik 1987). The dominant land-cover class in 2000 for Willamette Valley was agriculture (45.1 percent), followed by forest/woodland (33.5 percent), developed/urban (12.6 percent), and mechanically disturbed (4.0 percent). The largest net change for any land-cover class between 1973 and 2000 was the loss of 597 km² (-4.1 percent) of forest, followed by the loss of 320 km² of agricultural land. Most of the land use increases were for development (+3.1 percent) and mechanically disturbed land (+2.8 percent).

The Central California Valley ecoregion is an elongated basin extending approximately 650 km north to south through central California (Omernik 1987). The ecoregion is bound by the Sierra Nevada mountain range to the east and the Coast Range to the west. Agriculture land cover, which accounted for more than 70 percent of the ecoregion area, remained relatively stable from 1973 to 2000 with a net increase of 357 km² or 0.8 percent . The largest change in any one land cover class between 1973 and 2000 was a 3.9 percent loss (1,777 km²) of grasslands and shrublands in the ecoregion. Developed lands increased in cover from 6.5 percent to 9.0 percent of the total ecoregion area during this time frame.

3.6 Artificial Propagation

Each year approximately 380 million hatchery salmon and steelhead are released by government agencies on the Pacific coast and in New England (Kostow 2009). The introduction of hatchery produced fish can be a major cause of ecological perturbation in wild salmonid populations. Potential adverse effects of hatchery practices include: loss of genetic variability within and among populations (Hard et al. 1992, Reisenbichler 1997); disease transfer; increased competition for food, habitat, or mates; increased predation; altered migration; and the displacement of natural fish (Steward and Bjornn 1990 cited in NMFS, 2015, Hard et al. 1992, Fresh 1997). Recent research has demonstrated that the ecological effects of hatchery programs may significantly reduce wild population productivity and abundance even where genetic risks do not occur (Kostow 2009). Long-term domestication has eroded the fitness of hatchery reared fish in the wild and has reduced the productivity of wild stocks where significant numbers of hatchery fish spawn with wild fish.

Hatchery practices are cited as one of the key factors contributing to large reductions in salmonid populations in the Pacific Northwest over the past several decades, and remain a continuing threat to the recovery of many listed ESUs and DPSs. Hatcheries have been used for more than 100 years in the Pacific Northwest to produce fish for harvest and replace natural production lost to dam construction. Hatcheries have only minimally been used to protect and rebuild naturally produced salmonid populations. Hatchery contribution to naturally-spawning fish remains high for a number of Columbia River salmon populations, and it is likely that many returning unmarked adults are the progeny of hatchery-origin parents, especially where large hatchery programs operate (NWFSC 2015). For many populations the proportion of hatchery origin fish exceeds recovery goal criteria set for primary and contributing populations (Good et al. 2005, NWFSC 2015).

The Pacific Northwest Hatchery Reform Project was established in 2000. In their 2015 report to Congress the project's independent scientific review panel concluded that the widespread use of artificial propagation programs has contributed to the overall decline of wild salmonid populations. The states of Oregon and Washington have initiated a comprehensive program of hatchery and associated harvest reforms designed to manage hatchery broodstocks to achieve proper genetic integration with, or segregation from, natural populations, and to minimize adverse ecological interactions between hatchery and natural origin fish[6].

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[6] (WDFW, http://wdfw.wa.gov/hatcheries/esa.html; ODFW, http://www.dfw.state.or.us/fish/HGMP/final.asp).

Atlantic salmon have been stocked in at least 26 rivers in Maine from 1871 to 2003. Over 106 million fry and parr and over 18 million smolts have been stocked during this period (Fay et al. 2006). Currently there are two federal hatcheries that spawn and rear progeny of anadromous, captive reared Atlantic salmon, and four permanent feeding/rearing stations that raise progeny of captive reared and domestic broodstock obtained from the federal hatcheries for recovery and restoration stocking.

3.6.1 Non-native Species

When non-native plants and animals are introduced into habitats where they do not naturally occur they can have significant impacts on ecosystems and native fauna and flora. Non-native species can be introduced through infested stock for aquaculture and fishery enhancement, ballast water discharge, and from the pet and recreational fishing industries. Non-native species can reduce native species abundance and distribution, and reduce local biodiversity by out-competing native species for food and habitat. They may also displace food items preferred by native predators, disrupting the natural food web. The introduction of non-native species is considered one of the primary threats to ESA-listed species (Wilcove and Chen 1998). Non-native species were cited as a contributing cause in the extinction of 27 species and 13 subspecies of North American fishes over the past 100 years (Miller et al. 1989).

The introduction of invasive blue and flathead catfish along the Atlantic coast has the potential to adversely affect ongoing anadromous fish restoration programs and native fish conservation efforts, including Atlantic sturgeon restoration in mid-Atlantic and south Atlantic river basins (Brown et al. 2005, , J. Kahn, NMFS OPR, pers. comm. to R. Salz NMFS OPR, June 2016). Recent studies suggest that invasive species may reduce prey resources for Southern DPS green sturgeon. Green sturgeon may have difficulty feeding in substrate that has been invaded by Japanese eelgrass, which negatively impacts habitat for burrowing shrimp a common sturgeon prey item (Mary Moser, NMFS, pers. comm., June 18, 2015 cited in NMFS, 2015). Similarly, the invasive isopod (*U. pugettensis*) could also impact blue mud shrimp, another green sturgeon prey item (Olaf Langness, WDFW, and Brett Dumbauld, USDA-ARS, pers. comm. May 22, 2013 cited in NMFS, 2015).

Natural predator-prey relationships in aquatic ecosystems in Maine have been substantially altered by non-native species interactions. Several non-native fish species have been stocked throughout the range of Gulf of Maine DPS of Atlantic salmon. Those that are known to prey upon Atlantic salmon include smallmouth bass, largemouth bass, chain pickerel, northern pike, rainbow trout, brown trout, splake, yellow perch, and white perch (van den Ende 1993 cited in Fay, 2006, Baum 1997). Yellow perch, white perch, and chain pickerel were historically native to Maine, although their range has been expanded by stocking and subsequent colonization. Dams create slow water habitat that is preferred by chain pickerel and concentrate emigrating smolts in these head ponds by slowing migration speeds (McMenemy and Kynard 1988, Spicer et al. 1995). Brown trout, capable of consuming large numbers of stocked Atlantic salmon fry, have contributed to the decline of several native salmonid populations in North America (Alexander 1977, Alexander 1979, Taylor et al. 1984 all cited in Fay, 2006 #2827, Moyle 1976).

Introduction of non-native species on the West Coast has resulted in increased salmonid predation in many river and estuarine systems. Native resident salmonid populations have also been affected by releases of non-native hatchery reared salmonids (See 1.2.7 Artificial Propagation). The introduced northern pikeminnow is a significant predator of yearling juvenile Chinook migrants. Chinook salmon represented 29 percent of northern pikeminnow prey in lower Columbia reservoirs, 49 percent in the lower Snake River, and 64 percent downstream of Bonneville Dam (Friesen and Ward 1999). An ongoing northern pikeminnow management program has been in place since 1990 to reduce predation-related juvenile salmonid mortality. The rapid expansion of pikeminnow populations in the Pacific Northwest is believed to have been facilitated by alterations in habitat conditions (particularly increased water temperatures) that favor this species (Brown et al. 1994).

Predation of invasive lionfish on small reef fish and early life stages is a general concern throughout the Caribbean and could have an impact on Nassau grouper populations (Albins and Hixon 2008).

3.7 Fisheries

Commercial, recreational, and subsistence fisheries can result in substantial detrimental impacts on populations of ESA listed species. Past fisheries contributed to the steady decline in the population abundance of many ESA listed anadromous fish species. Although directed fishing for the species covered in this opinion is prohibited under the ESA, many are still caught as a result of ongoing fishing operations targeting other species (i.e., “bycatch”). Bycatch occurs when fishing operations interact with marine mammals, sea turtles, fish species, corals, sponges, or seabirds that are not the target species for commercial sale.

3.7.1 Directed Harvest

While directed fisheries for Atlantic salmon in the U.S. are at present illegal, impacts from past fisheries are an important factor contributing to the present low abundance of the Gulf of Maine DPS. The most complete records of commercial harvest of Atlantic salmon in the U.S. are for the Penobscot River, although historical records also mention commercial salmon fisheries in the Dennys, Androscoggin and Kennebec rivers (Kendall 1935, Beland et al. 1982, Beland 1984 all cited in Fay, 2006, Stolte 1981) reported that nearly 200 pound nets were operating in Penobscot Bay in 1872. A record commercial catch of 200,000 pounds of salmon was recorded for the Penobscot River in 1888. By 1898, landings had declined to 53,000 pounds and continued to decline in the following decades. The directed commercial fishery for Atlantic salmon in the Penobscot was eliminated by the Atlantic Sea Run Salmon Commission after the 1948 season when commercial harvest was reduced to only 40 fish. Directed fisheries for Atlantic salmon were further regulated by the adoption of the Atlantic Salmon Fishery Management Plan in 1987 which prohibits possession of Atlantic salmon in the U.S. Exclusive Economic Zone (NEFMC, http://www.nefmc.org/management-plans/atlantic-salmon).

The West Greenland fishery is one of the last directed Atlantic salmon commercial fisheries in the Northwest Atlantic. Greenland implemented a 45 mt quota for this fishery for 2015-2017. The West Greenland fishery is a mixed stock fishery and genetic analysis on captures from 2002 to 2004 indicate that Maine-origin salmon contribute between 0.1 and 0.8 percent to this fishery (ICES 2006). Based upon historic tag returns, the commercial fisheries of Newfoundland and Labrador historically intercepted far greater numbers of Maine-origin salmon than the West Greenland fishery (Baum 1997). A small commercial salmon fishery occurs off St. Pierre et Miquelon, a French territory south of Newfoundland. Historically, the fishery was very limited (2 to 3 mt per year). Genetic analysis on 134 samples collected in 2004 indicate that all samples originated from North American salmon, with roughly 2 percent of U.S. origin, presumably from the Gulf of Maine DPS.

Sport fishing for Atlantic salmon in Maine dates back to the mid-1800s. Recreational harvest regulations were not very restrictive through the 1970s. Increasingly restrictive regulations on the recreational harvest of Maine Atlantic salmon began in the 1980s as run sizes decreased notably. In 1995 regulations were promulgated for catch and release fishing only (i.e., zero harvest) of sea run Atlantic salmon throughout the state (Fay et al. 2006). By 2000, directed recreational fishing for sea run Atlantic salmon in Maine was prohibited. Illegal harvest ("poaching") of Maine Atlantic salmon has been reported (MASTF 1997 cited in Fay, 2006) but the level of this activity and the impact on the Gulf of Maine DPS has not been quantified.

During the mid-1800s, an estimated 10 to 16 million adult salmonids entered the Columbia River each year. Large annual harvests of returning adult salmon and steelhead during the late 1800s, ranging from 20 million to 40 million pounds, significantly reduced population productivity (ODFW 2002). The largest known harvest of Chinook salmon occurred in 1883 when Columbia River canneries processed 43 million pounds (Lichatowich and Lichatowich 2001). Commercial landings declined steadily from the 1920s to a low in 1993 when just over one million pounds of Chinook salmon were harvested (ODFW 2002). Harvest levels increased to 2.8 million pounds by the early 2000s, but almost half the harvest was hatchery produced fish. In the early 2000's, commercial harvest by tribal fisheries in the Columbia River ranged from between 25,000 and 110,000 fish. Recreational catches in both ocean and river fisheries have ranged from about 140,000 to 150,000 individuals over the same time frame. Non-Indian fisheries in the lower Columbia River are limited to a harvest rate of 1 percent. Treaty Indian fisheries are limited to a harvest rate of 5 percent to 7 percent, depending on the run size of upriver Snake River sockeye stocks. Snake River steelhead were historically taken in tribal and non-tribal gillnet fisheries, and in recreational fisheries in the mainstem Columbia River and its tributaries. In the 1970s, retention of steelhead in non-tribal commercial fisheries was prohibited, and in the mid 1980s tributary recreational fisheries in Washington adopted mark-selective regulations. Steelhead are still harvested in tribal fisheries and in mainstem recreational fisheries. Columbia River chum salmon were historically abundant and subject to substantial harvest until the 1950s (Johnson 1997). Illegal high seas driftnet fishing also likely contributed to past declines in Pacific salmon abundance although the extent of this activity is largely unknown.

Many grouper species are highly susceptible to overfishing, whether intentionally or as bycatch, due to a combination of life history traits including large size, late maturity, and tendency to form large spawning aggregations. Puerto Rico had significant commercial landings of Nassau grouper from the 1950s through the 1970s with fishermen targeting spawning aggregations (Schärer 2007). Landings subsequently dropped to negligible levels before the species was fully protected (in Commonwealth and federal waters) in 2004 (Sadovy 1997) (Matos-Caraballo 1997). Nassau grouper were considered “commercially extinct” in Puerto Rico by 1990 (Sadovy 1997); although the species still appeared in landings reports where it averaged approximately 11,000 pounds per year from 1994-2006.

Commercial harvest of eulachon in the Columbia and Fraser rivers was identified as a “low to moderate” threat by the Southern DPS eulachon biological review team. Current harvest levels are orders of magnitude lower than historic harvest levels, and a relatively small number of vessels still operate in this fishery. However, it is possible that even a small harvest of the remaining stock may slow recovery (75 FR 13012). Commercial fishing for eulachon is allowed in the Pacific Ocean, Columbia River, Sandy River, Umpqua River, and Cowlitz River. Commercial fishing in the Columbia River is managed according to the joint Washington and Oregon Eulachon Management Plan (WDFW and ODFW 2001). Under this plan, three eulachon harvest levels can be authorized based on the strength of the prior years’ run, resultant juvenile production estimates, and ocean productivity indices.

In the final listing rule, past and present commercial and recreational fishing, as well as poaching, were recognized as factors that pose a threat to the Southern DPS green sturgeon (71 FR 17757). Current regulations prohibit retention of green sturgeon in California, Oregon, and Washington state fisheries and in federal fisheries in the U.S. and Canada. These regulations apply to the range of both Southern and Northern DPS green sturgeon to address the possibility of capture of the threatened Southern DPS throughout the coast. Estimates based on past encounters suggest that Washington commercial fisheries outside of the lower Columbia River annually encounter 311 Southern DPS green sturgeon (pers. comm. with Kirt Hughes, WDFW January 30, 2015 cited in NMFS 2015c). An estimated 271 Southern DPS green sturgeon are annually encountered in lower Columbia River commercial fisheries (NMFS 2008a). Prior to the recreational retention limit, as many as 553 (1985) green sturgeon were harvested by anglers fishing in the lower Columbia River. A small number of green sturgeon (≤10) are still annually retained in this fishery due to misidentification or poaching.

Harvest records indicate that fisheries for sturgeon were conducted in every major coastal river along the Atlantic coast at one time, with fishing effort concentrated during spawning migrations (Smith 1985). Approximately 3,350 mt (7.4 million lbs) of sturgeon (Atlantic and shortnose combined) were landed in 1890 (Smith and Clugston 1997). The sturgeon fishery during the early years (1870 to 1920) was concentrated in the Delaware River and Chesapeake Bay systems. During the 1970s and 1980s sturgeon fishing effort shifted to the South Atlantic which accounted for nearly 80 percent of total U.S. landings (64 mt). By 1990 sturgeon landings were prohibited in Pennsylvania, District of Columbia, Virginia, South Carolina, Florida, and waters managed by the Potomac River Fisheries Commission. From 1990 through 1996 sturgeon fishing effort shifted to the Hudson River (annual average 49 mt) and coastal areas off New York and New Jersey (Smith and Clugston 1997). By 1996, closures of the Atlantic sturgeon fishery had been instituted in all Atlantic Coast states except for Rhode Island, Connecticut, Delaware, Maryland, and Georgia, all of which adopted a seven-foot minimum size limit. Poaching of Atlantic sturgeon continues and is a potentially significant threat to the species, but the present extent and magnitude of such activity is largely unknown.

3.7.2 Bycatch

Commercial bycatch is not thought to be a major source of mortality for Gulf of Maine DPS Atlantic salmon. Beland (1984 cited in Fay, 2006) reported that fewer than 100 salmon per year were caught incidental to other commercial fisheries in the coastal waters of Maine. A more recent study found that bycatch of Maine Atlantic salmon in herring fisheries is not a significant mortality source (ICES 2004). Commercial fisheries for white sucker, alewife, and American eel conducted in state waters also have the potential to incidentally catch Atlantic salmon.

Recreational angling occurs for many freshwater fish species throughout the range of the Gulf of Maine DPS Atlantic salmon. As a result Atlantic salmon can be incidentally caught (and released) by anglers targeting other species such as striped bass or trout. The potential also exists for anglers to misidentify juvenile Atlantic salmon as brook trout, brown trout, or landlocked salmon. A maximum length for landlocked salmon and brown trout (25 inches) has been adopted in Maine in an attempt to avoid the accidental harvest of sea-run Atlantic salmon due to misidentification.

Fisheries directed at unlisted Pacific salmonid populations, hatchery produced fish, and other species have caused adverse impacts to threatened and endangered salmonid populations. Incidental harvest rates for listed Pacific salmon and steelhead vary considerably depending on the particular ESU/DPS and population units. Bycatch represents one of the major threats to recovery as incidental harvest rates still remain as high as 50 percent-70 percent for some populations (NWFSC 2015). Freshwater fishery impacts on naturally-produced salmon have been markedly reduced in recent years through implementation of mark-selective fisheries (NWFSC 2015).

Take of Southern DPS green sturgeon in federal fisheries was prohibited as a result of the ESA 4(d) protective regulations issued in 2010 (75 FR 30714; June 2, 2010). Green sturgeon are occasionally encountered as bycatch in Pacific groundfish fisheries (Al-Humaidhi 2011), although the impact of these fisheries on green sturgeon populations is estimated to be small (NMFS 2012). (NMFS 2012) estimates between 86 and 289 Southern DPS green sturgeon are annually encountered as bycatch in the state-regulated California halibut bottom trawl fishery.

Approximately 50 to 250 green sturgeon are encountered annually by recreational anglers in the lower Columbia River (NMFS 2015c), of which 86 percent are expected to be Southern DPS green sturgeon based on the higher range estimate of Israel (Israel et al. 2009). In Washington, recreational fisheries outside of the Columbia River may encounter up to 64 Southern DPS green sturgeon annually (Kirt Hughes, WDFW, pers. comm., January 30, 2015 cited in NMFS, 2015). Southern DPS green sturgeon are also captured and released by California recreational anglers. Based on self-reported catch card data, an average of 193 green sturgeon were caught and released annually by California anglers from 2007-2013 (green sturgeon 5-year review). Recreational catch and release can potentially result in indirect effects on green sturgeon, including reduced fitness and increased vulnerability to predation. However, the magnitude and impact of these effects on Southern DPS green sturgeon are not well studied.

Directed harvest of Atlantic sturgeon is prohibited by the ESA. However, sturgeon are taken incidentally in fisheries targeting other species in rivers, estuaries, and marine waters along the east coast, and are probably targeted by poachers throughout their range (Collins et al. 1996) (ASSRT 2007). Commercial fishery bycatch is a significant threat to the viability of listed sturgeon species and populations. Bycatch could have a substantial impact on the status of Atlantic sturgeon, especially in rivers or estuaries that do not currently support a large subpopulation (< 300 spawning adults per year). Reported mortality rates of sturgeon (Atlantic and shortnose) captured in inshore and riverine fisheries range from 8 percent to 20 percent (Collins et al. 1996) (Bahn et al. 2012).

Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. Atlantic sturgeon originating from the five DPSs considered in this consultation are at risk of bycatch-related mortality in fisheries operating in the action area and beyond. Sturgeon are benthic feeders and as a result they are generally captured near the seabed unless they are actively migrating (Moser and Ross 1995). Atlantic sturgeon are particularly vulnerable to being caught in commercial gill nets, therefore fisheries using this type of gear account for a high percentage of Atlantic sturgeon bycatch and bycatch mortality. An estimated 1,385 individual Atlantic sturgeon were killed annually from 1989-2000 as a result of bycatch in offshore gill net fisheries operating from Maine through North Carolina (Stein et al. 2004b). Sturgeon are also taken in trawl fisheries, though recorded captures and mortality rates are thought to be low.

From 2001-2006 an estimated 649 Atlantic sturgeon were killed annually in offshore gill net and otter trawl fisheries From 2006-2010 an estimated 3,118 Atlantic sturgeon were captured annually in Northeast fisheries, resulting in approximately 391 mortalities (Miller and Shepherd 2011).

3.8 Vessel Related Stressors

Both large and small vessels can adversely affect listed species within the action area. The detrimental effects of vessel traffic can be both direct (i.e., ship strikes) and indirect (i.e., noise, harassment, displacement, avoidance).

Atlantic sturgeon are susceptible to vessel collisions. The Atlantic Sturgeon Status Review Team (ASSRT 2007) determined Atlantic sturgeon in the Delaware River are at a moderately high risk of extinction because of ship strikes, and sturgeon in the James River are at a moderate risk from ship strikes. Balazik (Balazik et al. 2012) estimated up to 80 sturgeon were killed between 2007 and 2010 in these two river systems. Ship strikes may also be threatening Atlantic sturgeon populations in the Hudson River where large ships move from the river mouth to ports upstream through narrow shipping channels. The channels are dredged to the approximate depth of the ships, usually leaving less than 6 feet of clearance between the bottom of ships and the river bottom. Any aquatic life along the bottom is sucked through the large propellers of these ships. Large sturgeon are most often killed by ship strikes because their size means they are unable to pass through ship propellers without making contact. Green sturgeon may also be susceptible to ship strikes but there is no data available indicating that this is a major source of mortality.

Collisions with ships are also one of the primary threats to marine mammals, particularly large whales. While interactions between killer whales and ships are known to occur, large migratory cetaceans including blue, fin, humpback, right, and gray whales are considered the most vulnerable to ship strikes, particularly along migratory routes that span thousands of miles. Only one killer whale ship strike was recorded the NMFS national large whale ship strike database from 1975-2002 (Jensen et al. 2004).

While ship strikes may be rare for this species, killer whales are likely more susceptible to other vessel related effects including noise and harassment. Reduced feeding behavior has been reported when vessels are present (Lusseau et al. 2009). However, there is insufficient data available to quantify the reduction in feeding for individual whales or to evaluate the cumulative behavioral effects of vessel traffic on killer whales. Commercial and recreational whale watching was identified as a “high severity” and “high likelihood” threat in the listing determination of Southern Resident killer whales and cited as a factor that could potentially affect recovery of this DPS. Other vessel traffic (not targeting killer whales) was identified as a “medium severity” and “high likelihood” threat. Current voluntary guidelines are in place regarding vessel activity around killer whales, but a vessel monitoring program has documented persistent violations of these guidelines for many years (Koski 2010 cited in NMFS, 2011). In 2009 NMFS proposed regulations under the ESA and MMPA to prohibit vessels from approaching killer whales within 200 yards, parking in the path of whales in inland waters of Washington State, and entering a conservation area during a defined season (74 FR 37674). NMFS has coordinated with the U.S. Coast Guard, Washington Department of Fish and Wildlife, and the Canadian Department of Fisheries and Oceans to evaluate the need for regulations or areas with vessel restrictions as described in the Southern Resident Killer Whales Recovery Plan.

3.9 Global Climate Change

The Intergovernmental Panel on Climate Change estimates that average global land and sea surface temperature has increased by 0.85°C (± 0.2) since the late 1800s, with most of the change occurring since the mid-1900s (IPCC 2013). This temperature increase is greater than what would be expected given the range of natural climatic variability recorded over the past 1,000 years (Crowley and Berner 2001). The Intergovernmental Panel on Climate Change estimates that the last 30 years were likely the warmest 30-year period of the last 1,400 years, and that global mean surface temperature change will likely increase in the range of 0.3 to 0.7°C by 2033.

Global climate change stressors, including consequent changes in land use, are major drivers of ecosystem alterations (Rahel and Olden 2008) (Bellard et al. 2012). Climate change is projected to have substantial direct effects on individuals, populations, species, and the community structure and function of marine, coastal, and terrestrial ecosystems in the foreseeable future (McCarty 2001, IPCC 2007, 2013). Increasing atmospheric temperatures have already contributed to changes in the quality of freshwater, coastal, and marine ecosystems and to the decline of endangered and threatened species populations (Mantua et al. 1997, Karl 2009) (Littell et al. 2009 cited in NMFS, 2015). All species discussed in this opinion are currently or are likely to be impacted by the direct and indirect effects of global climatic change.

Warming water temperatures attributed to climate change can have significant effects on survival, reproduction, and growth rates of aquatic organisms (Bellard et al. 2012). For example, warmer water temperatures have been identified as a factor in the decline and disappearance of mussel and barnacle beds in the Northwest (Harley 2011). Increasing surface water temperatures can cause the latitudinal distribution of freshwater and marine fish species to change as species move northward (Hiddink and Ter Hofstede 2008) (Britton et al. 2010). Cold water fish species and their habitat will begin to be displaced by warm water species (Hiddink and Ter Hofstede 2008, Britton et al. 2010). Fish species are expected to shift latitudes and depths in the water column, and the increasing temperatures may also result in expedited life cycles and decreased growth (Perry et al. 2005). Shifts in migration timing of pink salmon (*Oncorhynchus gorbuscha*), which may lead to high pre-spawning mortality, have already been connected to warmer water temperatures (Taylor 2008) . Climate-mediated changes in the global distribution and abundance of marine species are expected to reduce the productivity of the oceans by affecting keystone prey species in marine ecosystems such as phytoplankton, krill, and cephalopods. For example, climate change may reduce recruitment in krill by degrading the quality of areas used for reproduction (Walther et al. 2002).

Climate change will extend growing seasons and spatial extent of arable land in temperate and northern biomes. This would be accompanied by changes land use and pesticide application patterns to control pests (Kattwinkel et al. 2011). However modeling results indicate that predictions of mean trends in pesticide fate and transport is complicated by case specific and location specific conditions (Gagnon et al. 2016). Hellmann et al. (2008) described the consequences for climate change on the effectiveness of management strategies for invasive species. Such species are expected become more vigorous in areas where they had previously been limited by cold or ice cover. Increased vigor would make making mechanical control less effective and pesticide use likely. Some plant species may become more tolerant of herbicides due to elevated CO_2. Pesticide fate and transport, toxicities, degradation rates, and the effectiveness of biocontrol agents are expected to change with changing temperature and water regimes, driven largely by effects on rates in organism metabolism and abiotic reactions (Bloomfield et al. 2006, Schiedek et al. 2007, Noyes et al. 2009).

Warmer water also stimulates biological processes which can lead to environmental hypoxia. Oxygen depletion in aquatic ecosystems can result in anaerobic metabolism increasing, thus leading to an increase in metals and other pollutants being released into the water column (Staudinger et al. 2012). In addition to these changes, climate change may affect agriculture and other land development as rainfall and temperature patterns shift. Aquatic nuisance species invasions are also likely to change over time as oceans warm and ecosystems become less resilient to disturbances. If water temperatures warm in marine ecosystems, native species may shift poleward to cooler habitats, opening ecological niches that can be occupied by invasive species introduced via ships' ballast water or other sources (Ruiz et al. 1999, Philippart et al. 2011). Invasive species that are better adapted to warmer water temperatures could outcompete native species that are physiologically geared towards lower water temperatures. This scenario of native species displacement is currently occurring along central and northern California (Lockwood and Somero 2011).

Climate change is also expected to impact the timing and intensity of stream seasonal flows (Staudinger et al. 2012). Warmer temperatures are expected to reduce snow accumulation and increase stream flows during the winter, cause spring snowmelt to occur earlier in the year, and reduce summer stream flows in rivers that depend on snow melt. As a result, seasonal stream flow timing will likely shift significantly in sensitive watersheds (Littell et al. 2009 cited in NMFS, 2015). Warmer temperatures may also have the effect of increasing water use in agriculture, both for existing fields and the establishment of new ones in once unprofitable areas (ISAB 2007 cited in NMFS, 2015). This means that streams, rivers, and lakes will experience additional withdrawal of water for irrigation and increasing contaminant loads from returning effluent. Changes in stream flow due to use changes and seasonal run-off patterns may alter predator-prey interactions and change species assemblages in aquatic habitats.

Over the past 200 years, the oceans have absorbed about half of the CO_2 produced by fossil fuel burning and other human activities. This increase in CO_2 has led to a reduction of the pH of surface seawater of 0.1 units, equivalent to a 30 percent increase in the concentration of hydrogen ions in the ocean. If global emissions of CO_2 from human activities continue to increase at current rates, the average pH of the oceans is projected to fall by 0.5 units by the year 2100 (Raven et al. 2005). Although the scale of acidification changes would vary regionally, the resulting pH could be lower than the oceans have experienced over at least the past 420,000 years and the rate of change is probably one hundred times greater than the oceans have experienced at any time over that time interval. Acidification poses a significant threat to oceans because many major biological functions respond negatively to increased acidity of seawater. Ocean acidification, as a result of increased atmospheric CO_2, can interfere with fertilization, larval development, settlement success, and secretion of skeletons(Albright et al. 2010). Photosynthesis, respiration rate, growth rates, calcification rates, reproduction, and recruitment may be negatively impacted by increased ocean acidity (Raven et al. 2005). Marine species have already experienced stress related to the impacts of rising temperature. Corals, in particular, demonstrate extreme sensitivity to even small temperature increases. When sea temperatures increase beyond a coral's limit the coral "bleaches" by expelling the symbiotic organisms that not only give coral its color, but provide food for the coral through their photosynthetic capabilities. Bleaching events have steadily increased in frequency since the 1980s (Hoegh-Guldberg 2010). Kroeker Kroeker et al. (2010) reviewed 139 studies that quantified the effects of ocean acidification on aquatic life. Their analysis determined that the effects were variable depending on species, but effects were generally negative, with calcification being one of the most sensitive processes.

In summary, the direct effects of climate change include increases in atmospheric temperatures, decreases in sea ice, and changes in sea surface temperatures, ocean acidity, patterns of precipitation, and sea level. Indirect effects of climate change include altered reproductive seasons/locations, shifts in migration patterns, reduced distribution and abundance of prey, and changes in the abundance of competitors and/or predators. Climate change is most likely to have its most pronounced effects on species whose populations are already in tenuous positions (Williams et al. 2008).

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UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
1315 East-West Highway
Silver Spring, Maryland 20910
THE DIRECTOR

Ms. Debbie Edwards
Director, Office of Pesticide Programs
U.S. Environmental Protection Agency
One Potomac Yard
2777 S. Crystal Drive
Arlington, Virginia 22202

APR 2 0 2009

Dear Ms. Edwards:

Enclosed is the National Oceanic Atmospheric Administration (NOAA) National Marine Fisheries Service's (NMFS) final biological opinion (Opinion), issued under the authority of section 7(a)(2) of the Endangered Species Act (ESA), on the effects of the U.S. Environmental Protection Agency's (EPA) proposed registration of pesticide products containing the active ingredients carbaryl, carbofuran, and methomyl on endangered species, threatened species, and critical habitat that has been designated for those species. This Opinion assesses the effects of all pesticides containing carbaryl, carbofuran, or methomyl on 28 listed Pacific salmonids.

After considering the status of the listed resources, the environmental baseline, and the direct, indirect, and cumulative effects of EPA's proposed action on listed species, NMFS concludes that pesticide products containing carbaryl and carbofuran are likely to jeopardize the continuing existence of 22 listed Pacific salmonids as described in the attached Opinion. NMFS also concluded that the effects of carbaryl and carbofuran are likely to destroy or adversely modify designated habitat for 20 of 26 listed salmonids. NMFS has not designated critical habitat for two listed salmonids. NMFS determinations for no jeopardy and no adverse modification of critical habitat apply to Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon. We further conclude that pesticide products containing methomyl are likely to jeopardize 18 listed Pacific salmonids and likely to destroy or adversely modify critical habitat for 16 of 26 salmonids with designated critical habitat. NMFS determinations for no jeopardy and no adverse modification of designated critical habitat apply to California Coastal Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, and Snake River steelhead. As NMFS has not designated critical habitat for the Lower Columbia River coho salmon or Puget Sound steelhead, the action area contains no designated critical habitat for these species. Thus, the Opinion presents no further critical habitat analysis for the Lower Columbia River coho salmon and Puget Sound steelhead.

As required by section 7 of the ESA, NMFS provides an incidental take statement with the Opinion. The incidental take statement describes reasonable and prudent measures NMFS considers necessary or appropriate to minimize incidental take associated with this action. The incidental take statement also sets forth nondiscretionary terms and conditions, including reporting requirements that EPA and any person who performs the action must comply with to carry out the reasonable and prudent measures. Incidental take from actions by EPA and the applicants that meets these terms and conditions will be exempt from the ESA section 9 prohibitions for take.

This Opinion assesses effects to listed Pacific salmonids pursuant to the ESA. It does not address EPA's obligation under the Magnuson-Stevens Fishery Conservation and Management Act to consult on effects to essential fish habitat (EFH) for salmonids and other federally-managed species. Please contact Mr. Tom Bigford or Ms. Susan-Marie Stedman in NMFS' Office of Habitat Conservation at 301-713-4300 regarding the EFH consultation process.

If you have questions regarding this Opinion please contact me or Ms. Angela Somma, Chief of our Endangered Species Division at (301) 713-1401.

Sincerely,

for James H. Lecky
Director
Office of Protected Resources

Enclosure

National Marine Fisheries Service

Endangered Species Act Section 7 Consultation

Biological Opinion

Environmental Protection Agency Registration of Pesticides Containing Carbaryl, Carbofuran, and Methomyl

Photo Credit: Desmond Maynard NMFS

Photo Credit: Tom Maurer USFWS

April 20, 2009

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National Marine Fisheries Service

Endangered Species Act Section 7 Consultation

Biological Opinion

Agency: United States Environmental Protection Agency

Activities Considered: Authorization of pesticide products containing the a.i.s carbaryl, carbofuran, and methomyl, and their formulations in the United States and its affiliated territories

Consultation Conducted by: Endangered Species Division of the Office of Protected Resources, National Marine Fisheries Service

Approved by: Helen M Golde for James H. Lecky

Date: April 20, 2009

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. §1531 *et seq.*) requires each federal agency to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a federal agency's action "may affect" a protected species, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action "may affect but is not likely to adversely affect" endangered species, threatened species or designated critical habitat (50 CFR §420.14(b)).

The United States (U.S.) Environmental Protection Agency (EPA) initiated consultation with NMFS on its proposal to authorize use, pursuant to the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. 136 *et seq.*, of pesticide products containing the a.i.s (a.i.s) of carbaryl, and methomyl on April 1, 2003, and of carbofuran on December 1, 2004. EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (reregistrations and special review), 18 (emergency use), or 24(c) [Special Local Needs (SLN)]. At that time, EPA determined that uses of pesticide products containing these ingredients "may affect" most of the 26 Evolutionarily Significant Units (ESUs) of Pacific salmonids listed as endangered or threatened and designated critical habitat for the ESUs. This document represents NMFS' biological opinion (Opinion) on the impacts of EPA's authorization of pesticide products containing the above-mentioned a.i.s on the listed ESUs, plus on two newly listed salmonids. This is a partial consultation because pursuant to the court's order, EPA sought consultation on only this group of listed species under NMFS' jurisdiction. However, even though the court's order did not address the two more recently listed salmonids, NMFS analyzed the impacts of EPA's action to them because they belong to the same taxon. NMFS analysis requires consideration of the same information. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species and consults with NMFS as necessary.

This Opinion is prepared in accordance with section 7(a)(2) of the ESA and implementing regulations at 50 CFR §402. However, consistent with the decision in <u>Gifford Pinchot Task Force v. USFWS, 378 F.3d 1059 (9th Cir. 2004)</u>, we did not apply the regulatory definition of "destruction or adverse modification of critical habitat" at 50 CFR §402.02. Instead, we relied on the statutory provisions of the ESA to complete our analysis of the effects of the action on designated critical habitat.

This Opinion is based on NMFS' review of the package of information the EPA submitted with its 2003 and 2004 requests for formal consultation on the proposed authorization of the above a.i.s. It also includes our review of recovery plans for listed Pacific salmonids, past and current research and population dynamics modeling efforts, monitoring reports from prior research, Opinions on similar research, published and unpublished scientific information on the biology and ecology of threatened and endangered salmonids in the action area, and other sources of information gathered and evaluated during the consultation on the proposed authorization of a.i.s for carbaryl, carbofuran, and methomyl. NMFS also considered information and comments provided by EPA and by the registrants identified as applicants by EPA.

Background

On January 30, 2001, the Washington Toxics Coalition, Northwest Coalition for Alternatives to Pesticides, Pacific Coast Federation of Fishermen's Associations, and Institute for Fisheries Resources filed a lawsuit against EPA in the U.S. District Court for the Western District of Washington, Civ. No. 01-132. This lawsuit alleged that EPA violated section 7(a)(2) of the ESA by failing to consult on the effects to 26 ESUs of listed Pacific salmonids of its continuing approval of 54 pesticide a.i.s.

On July 2, 2002, the court ruled that EPA had violated ESA section 7(a)(2) and ordered EPA to initiate interagency consultation and make determinations regarding effects to the salmonids on all 54 a.i.s by December 2004.

In December 2002, EPA and the USFWS and NMFS (referred to as the Services) began interagency discussions for streamlining EPA's court ordered consultations.

On January 24, 2003, EPA and the Services published an Advance Notice of Proposed Rulemaking seeking public comment on improving the process by which EPA and the Services work together to protect listed species and critical habitat (68 FR 3785).

Between May and December 2003, EPA and the Services reviewed EPA's ecological risk assessment methodology and earlier drafts of EPA's "Overview of the Ecological Risk Assessment Process in the Office of Pesticide Programs, U.S. Environmental Protection Agency (Overview Document)". EPA and the Services also developed counterpart regulations to streamline the consultation process.

On January 22, 2004, the court enjoined application of pesticides within 20 (for ground) and 100 (for aerial) feet (ft) of streams supporting salmon. Washington Toxics Coalition v. EPA, 357 F.Supp. 2d 1266 (W.D. Wash. 2004). The court imposed several additional restrictions on pesticide use in specific settings.

On January 23, 2004, EPA finalized its Overview Document which specified EPA's conduct of ecological risk assessment on pesticide registrations.

On January 26, 2004, the Services approved EPA's procedures and methods for conducting ecological risk assessments and approved interagency counterpart regulations for EPA's pesticide registration program.

On January 30, 2004, the Services published in the Federal Register (69 FR 4465) proposed joint counterpart regulations for consultation under the ESA for regulatory actions under the FIFRA, codified at 50 C.F.R. Part 402 Subpart D.

On August 5, 2004, the Services promulgated final joint counterpart regulations for EPA's ESA-related actions taken pursuant to FIFRA. These regulations and the Alternative Conservation Agreement (ACA) under the regulations allowed EPA to conduct independent analyses of potential impacts of pesticide registration on listed species and their designated critical habitats. The ACA outlined procedures to ensure EPA's risk assessment approach will produce effect determinations that reliably assess the effects of pesticides on listed species and designated critical habitat. Additionally, EPA and the Services agreed to meet annually, or more frequently as may be deemed appropriate. The intention of these meetings was to identify new research and other activities that may improve EPA's current approach for assessing the potential ecological risks posed by use of a pesticide to listed species or designated critical habitat.

On September 23, 2004, the Washington Toxics Coalition and others challenged the counterpart regulations in the U.S. District Court for the Western District of Washington, Civ. No. 04-1998, alleging that the regulations were not authorized by the ESA and that the Services had not complied with the Administrative Procedure Act and the National Environmental Policy Act (NEPA) in promulgating these counterpart regulations.

In January 2006, EPA and the Services developed a draft joint interagency research agenda to address several critical areas of scientific and procedural uncertainties in EPA's current effects determination process. The jointly developed document identified eight areas of risk assessment and research uncertainties.

On August 24, 2006, the court determined the Services did not implement NEPA procedures properly during their promulgation of the joint counterpart regulations for EPA actions under FIFRA. Additionally, the court determined that the "not likely to adversely affect" and emergency consultation provisions of the counterpart regulations were arbitrary and capricious and contrary to the substantive requirements of ESA section 7(a)(2). The court determined that EPA may conduct its own formal consultation with the Services' involvement. Washington Toxics Coalition v. Department of the Interior, 457 F.Supp. 2d 1158 (W.D.Wash. 2006).

On November 5, 2007, the Northwest Coalition for Alternatives to Pesticides and others filed a legal complaint in the U.S. District Court for the Western District of Washington, Civ. No. 07-1791, against NMFS for its unreasonable delay in completing the section 7 consultations for EPA's registration of 54 pesticide a.i.s.

On July 30, 2008, NMFS and the plaintiffs entered into a settlement agreement with the Northwest Coalition for Alternatives to Pesticides. NMFS agreed to complete consultation within four years on 37 a.i.s. (EPA had concluded that 17 of the 54 a.i.s at issue in the first litigation would not affect any listed salmonid species or any of their designated critical habitat, and so did not initiate consultation on those a.i.s.)

On November 18, 2008, NMFS issued its first Opinion for three organophosphates: chlorpyrifos, diazinon, and malathion. This second consultation evaluates three carbamate insecticides: carbaryl, carbofuran, and methomyl. EPA consultations on pesticide products currently focus on their effects to listed Pacific salmonids. EPA consultations remain incomplete until all protected species under NMFS' jurisdiction are covered.

Consultation History

On April 1, 2003, the EPA sent a letter to NMFS' Office of Protected Resources (OPR) requesting section 7 consultation for the registration of the a.i. carbaryl and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's Office of Pesticide Programs (OPP) determined that the use of carbaryl will have "no effect" for 4 ESUs, "may affect but is not likely to adversely affect" 2 ESUs, and "may affect" 20 ESUs of listed salmonids. EPA's "no effect" determinations for carbaryl applied to Northern California steelhead, SONCC coho salmon, Hood Canal Summer-run chum salmon, and Ozette Lake sockeye salmon.

On April 1, 2003, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the a.i. methomyl and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, the EPA's OPP determined that the use of methomyl will have "no effect" for 2 ESUs, and "may affect" 24 ESUs of listed salmonids. EPA's "no effect" determinations for methomyl applied to the Northern California steelhead and California Coastal Chinook salmon ESUs.

On December 1, 2004, the EPA sent a letter to NMFS' OPR requesting section 7 consultation for the registration of the a.i. carbofuran and detailing its effects determinations on 26 ESUs of Pacific salmonids listed at that time. In that same letter, EPA's OPP determined that the use of carbofuran will have "no effect" for 3 ESUs; "may affect but is not likely to adversely affect" 18 ESUs, and "may affect" 3 ESUs of listed salmonids. EPA's "no effect" determinations applied to the California Coastal Chinook salmon, Central California coho salmon, and Northern California steelhead.

On June 28, 2005, NMFS listed the Lower Columbia River coho salmon ESU as endangered. Given this recent listing, EPA's 2003 and 2004 effects determinations for carbaryl, carbofuran, and methomyl on listed Pacific salmonids lack an effects determination for the Lower Columbia River coho salmon.

On May 22, 2007, NMFS listed the Puget Sound Steelhead Distinct Population Segment (DPS) as threatened. Given this recent listing, EPA's 2003 and 2004 effects determinations for carbaryl, carbofuran, and methomyl on listed Pacific salmonids lack an effects determination for the Puget Sound steelhead.

On December 10-12, 2007, EPA and the Services met and discussed approaches for moving forward with ESA consultations and pesticide registrations. The agencies agreed to develop methodologies for filling existing data gaps. In the interim, the Services will develop approaches within their Opinions to address these gaps. The agencies identified communication and coordination mechanisms to address technical and policy issues and procedures for conflict resolution.

On February 11, 2008, NMFS listed the Oregon Coast coho salmon ESU as threatened. EPA's 2003 and 2004 initiation packages for carbaryl, carbofuran, and methomyl provided an effects determination for the Oregon Coast coho salmon ESU. This ESU was previously listed in 1998 and its ESA status was in-flux until 2008.

On August 20, 2008, NMFS met with EPA and requested EPA to identify applicants for this and subsequent pesticide consultations. NMFS also requested information on EPA's cancellation of carbofuran and of existing stocks of carbofuran.

On August 29, 2008, NMFS met with EPA and the applicants for chlorpyrifos, diazinon, and malathion. At that meeting, NMFS asked EPA to identify applicants for this and subsequent pesticide consultations.

On September 16, 2008, NMFS requested EPA to confirm the status of EPA's cancellation of carbofuran and for existing stocks of that same compound during a conference call.

On September 17, 2008, NMFS requested EPA approval of Confidential Business Information (CBI) clearance for certain staff members in accordance with FIFRA regulations and access to EPA's incident database so NMFS staff may evaluate CBI materials from the applicants and incident reports for the a.i.s under consultation. EPA conveyed to NMFS that no access to the incident database would be authorized and the reports will be sent directly from EPA to NMFS.

On September 23, 2008, NMFS staff received notification of CBI clearance from EPA.

On September 26, 2008, NMFS sent correspondence to EPA informing it of the roles of the action agency and applicants during formal consultation. NMFS also requested incident reports and label information for subsequent pesticide consultations from EPA. The specified timeline for NMFS' receipt of incident report and label information for carbaryl, carbofuran, and methomyl was November 3, 2008.

On October 3, 2008, NMFS received post-2002 incident reports for carbaryl, carbofuran, and methomyl from EPA.

On November 5, 2008, NMFS sent an e-mail to EPA requesting it to identify applicants for upcoming pesticide consultations and label and incident report information for carbaryl, carbofuran, and methomyl. NMFS also requested information regarding whether final cancellation of carbofuran or any if its uses had occurred.

On November 13, 2008, EPA provided an interim e-mail response to NMFS' November 5, 2008 query. EPA stated that it was developing a process to identify applicants for carbaryl, carbofuran, and methomyl. No applicants were identified in EPA's response. EPA also stated that incident data for carbaryl, carbofuran, and methomyl were sent via FedEx to NMFS on October 2, 2008, which we received. Finally, EPA confirmed that no final cancellations for carbofuran have occurred subsequent to the Scientific Advisory Panel (SAP) meeting required for action relative to the Notice of Intent to cancel this compound. The SAP meeting occurred on February 5, 2008.

On December 1, 2008, NMFS repeated its request to EPA to identify applicants for carbaryl, carbofuran, and methomyl via e-mail. NMFS also requested EPA to provide technical staff contact information for these same chemicals so NMFS staff may request information from them during this consultation.

On December 15, 2008, EPA informed NMFS via e-mail that it would send letters to the technical registrants of carbaryl, carbofuran, and methomyl. EPA also stated that it would inform the parties that they may submit information relative to the consultation directly to NMFS with a copy to EPA. EPA also offered to include additional information requested by NMFS pertinent to the consultation into that same letter.

On December 16, 2008, NMFS and EPA discussed each agency's notification strategy of prospective applicants for this consultation. As the action agency, EPA indicated it would identify and contact prospective applicants. EPA limited applicant status to those technical registrants who have all information pertinent to the consultation.

On December 18, 2008, EPA sent formal correspondence to four technical registrants. EPA's letter requested confirmation on their desire to have applicant status and for parties to submit data not already provided with EPA's consultations that may inform the outcome of the consultation. That information includes any toxicity data, field studies or mesocosm studies not part of the consultation package, or EPA's Interim Registration Eligibility Decision (IRED) or Registration Eligibility Decision (RED) documents for the pesticide a.i.; and current labels for end use products or if available, a master label that includes all use instructions for all products containing the a.i.s. These data would be submitted to NMFS and EPA.

On December 19, 2008, EPA identified technical staff contact information and four applicants to NMFS for this consultation via formal correspondence. In that same letter, EPA referred NMFS to the IRED and RED documents for any changes to the three subject a.i.s since consultation was initiated in 2003 and 2004.

On that same date, NMFS received electronic copies of EPA letters sent to the applicants. EPA identified the following applicants: Bayer CropScience LP, Drexel Chemical Company, E.I., duPont de Nemours and Company (DuPont), and FMC Corporation.

On January 5, 2009, NMFS requested clarification on EPA's registration action for carbofuran via e-mail. Questions pertained to the six carbofuran uses that have not been proposed for voluntary cancellation; the duration of the cancellation process for these same six uses as well as for 22 crop uses proposed for voluntary cancellation; and whether other current or pending registrations for carbofuran (FIFRA sections 3, 4, 18, or 24 (c)) are anticipated.

On January 7, 2009, EPA notified NMFS via email that it was conducting an internal policy review and would provide a full response to NMFS' query on carbofuran as soon as possible.

On January 8 and 9, 2009, EPA and NMFS exchanged e-mails scheduling a meeting at the end of January with identified applicants for this consultation.

On January 9, 2009, NMFS provided EPA with an electronic draft of the Description of the Proposed Action and associated Appendix for this consultation. NMFS requested EPA comments on these documents by January 23, 2009.

On January 21, 2009, the agencies agreed to meet with the applicants on January 30, 2009, at OPP Headquarters in Crystal City, Virginia.

On January 22, 2009, NMFS requested the following information from EPA: the bibliography for the carbofuran IRED (August 3, 2006), the report study cited as Table 16 within the carbofuran IRED; and the methomyl study cited in the Science Chapter – Master Record Identification Number (MRID) #00131255.

On that same date, DuPont informed NMFS that it would submit information to NMFS and EPA to support the consultation for methomyl.

On that same date, EPA informed NMFS that it would provide comments to NMFS on the draft Description of the Proposed Action by January 30, 2009.

On January 23, 2009, EPA instructed DuPont to send any data in support of the methomyl consultation to both EPA and NMFS.

On January 26, 2009, NMFS received data on methomyl from DuPont. The package included a cover letter, analysis of risk to methomyl to listed Pacific salmonids; copies of four studies, including three toxicity tests with formulated material and an environmental fate study (dissipation of methomyl in a simulated pond); and copies of methomyl product labels held by DuPont.

On that same date, NMFS also received confidential information on sales of Lannate (DuPont methomyl product) in Washington and Oregon from 2004-2006.

On January 27, 2009, NMFS asked EPA via e-mail to identify agenda topics and a list of participants/applicants for the January 30, 2009, meeting. EPA provided the requested information on that same date.

On January 29, 2009, NMFS received information from Bayer CropScience, the registrant for carbaryl. The information included a summary of Section 3 label restrictions, a "master label table", and copies of current 24(c) and Section 3 labels.

On January 30, 2009, NMFS met with EPA, Bayer CropScience, DuPont, and FMC Corporation. At this meeting, NMFS explained the consultation procedure and timelines for this consultation. The applicants also presented information to NMFS and EPA on these a.i.s. DuPont presented the risk of methomyl on endangered salmonids. FMC presented the U.S. registration status and use in the Pacific Northwest for carbofuran. Bayer CropScience presented use pattern summaries for carbaryl. This venue facilitated a question-answer session between the applicants and the agencies.

On that same date, NMFS received electronic files of applicant presentation materials on carbaryl, carbofuran, and methomyl. EPA also provided NMFS with a PDF file of the Environmental Fate and Effects Division's (EFED) RED Science Chapter for methomyl.

On February 2, 2009, NMFS received carbofuran data from FMC Corporation. Materials included the status on furadan registration and use in the relevant Pacific Northwest, the proposed federal label for furadan, current special local needs label for potato use in Oregon and spinach grown for seed use in Washington, and the Federal Register notice for proposed voluntary cancellation of most uses.

On that same date, NMFS queried FMC Corporation via e-mail regarding when EPA's response on the proposed federal label for furadan is expected. NMFS also requested information whether the proposed permitted uses for foliar application of furadan on cotton would apply in California, Idaho, Oregon, and Washington.

On February 3, 2009, NMFS received comments from EPA on the draft Description of the Proposed Action relevant to carbofuran and methomyl. EPA disagreed with FMC's statement that use of carbofuran has been discontinued on field corn in the Pacific Northwest. According to EPA, carbofuran use is specified on federal labels and is used in California, Idaho, Oregon, and Washington. Carbofuran is also used on sunflowers grown in these states. EPA feedback for methomyl pertained to the SLN in California and the status of some section 24(c) actions. EPA provided no response regarding carbaryl.

On that same date, NMFS requested clarification on registered carbofuran uses and EPA's verification of NMFS' draft Description of the Proposed Action for this ingredient.

On February 5, 2009, NMFS received the two early life stage studies for fish (MRID 131255 and 126862) from EPA. On that same date, FMC Corporation responded to NMFS' questions regarding uses of furadan raised in it February 2, 2009, e-mail.

On February 9, 2009, EPA provided responses via e-mail regarding questions posed in NMFS' February 3, 2009, e-mail, on carbaryl. EPA provided no comments on NMFS' Description of the Proposed Action for carbaryl and no response towards NMFS' questions on carbofuran.

On February 11, 2009, NMFS received a study on carbaryl from Bayer CropScience.

On February 13, 2009, NMFS contacted EPA by phone and requested EPA comments on the draft Description of the Proposed Action.

On that same date, NMFS received a copy of a position paper on key issues related to carbaryl use, ecotoxicology, and aquatic exposure from Bayer CropScience. Bayer Crop Science also sent this document to EPA.

On February 20, 2009, EPA e-mailed NMFS information on carbaryl use patterns for Idaho, Oregon, and Washington. NMFS previously requested this information for the draft Description of the Proposed Action.

On February 20, 2009, EPA e-mailed NMFS information on carbaryl use patterns for Idaho, Oregon, and Washington.

On March 5, 2009, NMFS received an extension from the court for this consultation from March 31, 2009, to April 20, 2009.

On March 12, 2009, NMFS requested clarification from EPA on the carbaryl use data provided in its February 20, 2009, e-mail. NMFS also requested EPA forward cancellation notices for carbaryl for several crop uses once they are available.

On March 13, 2009, NMFS met with and provided EPA a list of questions pertaining to past, ongoing, and future pesticide consultations with EPA.

On March 17, 2009, NMFS e-mailed EPA instructions to access a pdf copy of the draft Opinion from NMFS' ftp site. NMFS also FedExed a pdf copy of the draft Opinion to EPA on that same day. Although this draft did not include Reasonable and Prudent Alternatives (RPAs), NMFS conveyed to EPA that RPAs will be provided on the following day.

On that same date, EPA downloaded the draft Opinion file from NMFS' ftp site and requested NMFS provide word version files for future draft Opinions.

On March 18, 2009, NMFS provided EPA instructions to access a pdf copy of the draft Opinion, including RPAs from NMFS' ftp site. NMFS also FedExed a word file of a full draft Opinion, including RPAs and a separate RPA file to EPA. On this same day, EPA responded to NMFS' query of SLN use of carbaryl in Washington State. EPA's e-mail further stated that Bayer CropScience does not support carbaryl use for adult mosquito treatments although carbaryl is registered for use on a number of sites where it could kill mosquitoes.

On that same date, EPA e-mailed NMFS information on the voluntary cancellation of carbofuran. NMFS replied to EPA's message and reminded EPA of NMFS questions raised in a February 3, 2009, e-mail regarding uses of carbofuran.

On March 24, 2009, EPA notified NMFS via e-mail of receipt of the cd containing the word versions of a complete draft Opinion and separate RPA document.

On March 30, 2009, EPA and NMFS confirmed their availability to meet with the three identified applicants for this consultation on April 7, 2009.

On April 7, 2009, EPA, NMFS, and the applicants met at EPA's office in Crystal City, VA. NMFS presented its assessment and conclusions on the draft Opinion. The applicants provided a combined presentation on their general and specific comments on NMFS' draft Opinion. Afterwards, all parties discussed potential Reasonable and Prudent Alternatives (RPAs) as NMFS concluded that EPA's proposed action will likely result in jeopardy of listed species and adverse modification of critical habitat for listed Pacific salmonids under NMFS' jurisdiction.

On that same date, NMFS and EPA discussed RPAs for this consultation and for NMFS' November 18, 2008, Opinion for chlorpyrifos, diazinon, and malathion.

On April 9, 2009, NMFS received copies of the three presentations made by the applicants at the April 7, 2009, meeting.

On April 10, 2009, NMFS received written comments from EPA on the March 18, 2009 draft Opinion for carbaryl, carbofuran, and methomyl. On that same date, NMFS received comments on the draft Opinion from Bayer CropScience on carbaryl and from DuPont on methomyl.

On April 15, 2009, NMFS received additional comments from Bayer CropScience on the draft Opinion.

On April 18, 2009, NMFS received additional comments from Bayer CropScience on the draft Opinion.

Description of the Proposed Action

The Federal Action

The proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing carbaryl, carbofuran, and methomyl. The purpose of the proposed action is to provide tools for pest control that do not cause unreasonable adverse effects to the environment throughout the U.S. and its affiliated territories. Pursuant to FIFRA, before a pesticide product may be sold or distributed in the U.S. it must be exempted or registered with a label identifying approved uses by EPA's OPP. Once registered, a pesticide may not legally be used unless the use is consistent with directions on its approved label (http:www.epa.gov/pesticides/regulating/registering/index.htm). EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (re-registrations and special review), 18 (emergency use), or 24(c) SLN.

EPA's pesticide registration process involves an examination of the ingredients of a pesticide, the site or crop on which it will be used, the amount, frequency and timing of its use, and its storage and disposal practices. Pesticide ingredients may include active and other ingredients, adjuvants, and surfactants (described in greater detail below). The EPA evaluates the pesticide to ensure that it will not have unreasonable adverse effects on humans, the environment, and non-target species. An unreasonable adverse effect on the environment is defined in FIFRA as, "(1) any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of the pesticide, or (2) a human dietary risk from residues that result from a use of a pesticide in or on any food inconsistent with the standard under section 408 of the Federal Food, Drug, and Cosmetic Act (FFDCA) (21 U.S.C. §346a)." 7 U.S.C. 136(b).

After registering a pesticide, EPA retains discretionary involvement and control over such registration. EPA must periodically review the registration to ensure compliance with FIFRA and other federal laws (7 U.S.C. §136d). A pesticide registration can be cancelled whenever "a pesticide or its labeling or other material… does not comply with the provisions of FIFRA or, when used in accordance with widespread and commonly recognized practice, generally causes unreasonable adverse effects on the environment."

On December 12, 2007, EPA, NMFS, and FWS agreed that the federal action for EPA's FIFRA registration actions will be defined as the "authorization for use or uses described in labeling of a pesticide product containing a particular pesticide ingredient." In order to ensure that EPA's action will not jeopardize listed species or destroy or adversely modify critical habitat, NMFS' analysis necessarily encompasses the impacts to Pacific salmonid ESUs/DPSs of all uses authorized by EPA, regardless of whether those uses have historically occurred.

Pesticide Labels. For this consultation, EPA's proposed action encompasses all approved product labels containing carbaryl, carbofuran, or methomyl; their degradates, metabolites, and formulations, including other ingredients within the formulations; adjuvants; tank mixtures; and their individual and collective interactions when applied in agricultural, urban, and residential landscapes throughout the U.S. and its territories. These activities comprise the stressors of the action (Figure 1). The three biological evaluations (BEs) indicate that carbaryl, carbofuran, and methomyl are labeled for a variety of uses including applications to residential areas and crop lands (EPA 2003; EPA 2003; EPA 2004). Modifications have been made or are planned for new product labels containing carbaryl, carbofuran, and methomyl as a result of reregistration activities that have occurred since the release of the BEs.

The Food Quality Protection Act (FQPA) of 1996 required EPA to complete an assessment of the cumulative effects on human health resulting from exposure to multiple chemicals that have a common mechanism of toxicity. In 2001, EPA identified the *N*-methyl carbamate (NMC) pesticides as a group which shares a common mechanism of toxicity. This group includes carbaryl, carbofuran, methomyl, and seven other cholinesterase-inhibiting pesticides. (http://www.epa.gov/pesticides/cumulative/carbamate_risk_mgmt.htm). EPA published a preliminary Cumulative Risk Assessment for NMC pesticides in 2005 and revised the risk assessment in 2007 (EPA 2007). Concurrent with completing the revised assessments, EPA completed tolerance reassessments and REDs for the NMC pesticides (EPA 1998; EPA 2006; EPA 2007; EPA 2008). EPA has identified measures to address cumulative risk of NMC pesticides (http://www.epa.gov/pesticides/cumulative/carbamate_risk_mgmt.htm). Some of the risk reduction measures for carbaryl, carbofuran, and methomyl follow:

Carbaryl **–** EPA intends to evaluate the revised worker assessment, which may require an amendment to the RED. EPA continues to respond to petitions requesting that carbaryl be cancelled and its tolerances revoked.

Carbofuran **–** EPA is pursuing cancellation of all carbofuran uses in the U.S. EPA has received a request from the carbofuran registrant, FMC Corporation, for voluntary cancellation of 22 crop uses of this pesticide. FMC Corporation has six uses not proposed for voluntary cancellation that EPA indicates still present risk concerns and are subject to future regulatory action by EPA. In July 2008, EPA initiated action to revoke existing carbofuran tolerances (residue limits in food) due to unacceptable dietary risks, especially to children, from consuming food or water alone or from a combination of food and water with carbofuran residues. Following resolution of the tolerance revocations, EPA plans to proceed with cancellation of remaining carbofuran uses due to unreasonable ecological and worker risks (Federal Register / Vol. 73, No. 245 / December 19, 2008 /77690-77693). In March 2008, EPA announced an order for the cancellation of some registrations and termination of certain uses, voluntarily requested by the registrant (Federal Register/Vol. 74, No. 51/ March 18, 2009/11551-11553).

Methomyl **–** The intent of registrants for voluntary cancellation of methomyl use on strawberry and grapes were incorporated in the *N*-methyl carbamate revised cumulative risk assessment. With these and other mitigation measures for these individual pesticides, EPA concluded that the cumulative risks to humans associated with the *N*-methyl carbamates are below EPA's regulatory level of concern (http://www.epa.gov/pesticides/cumulative/carbamate_risk_mgmt.htm). The FQPA does not address cumulative risk of pesticides to aquatic resources.

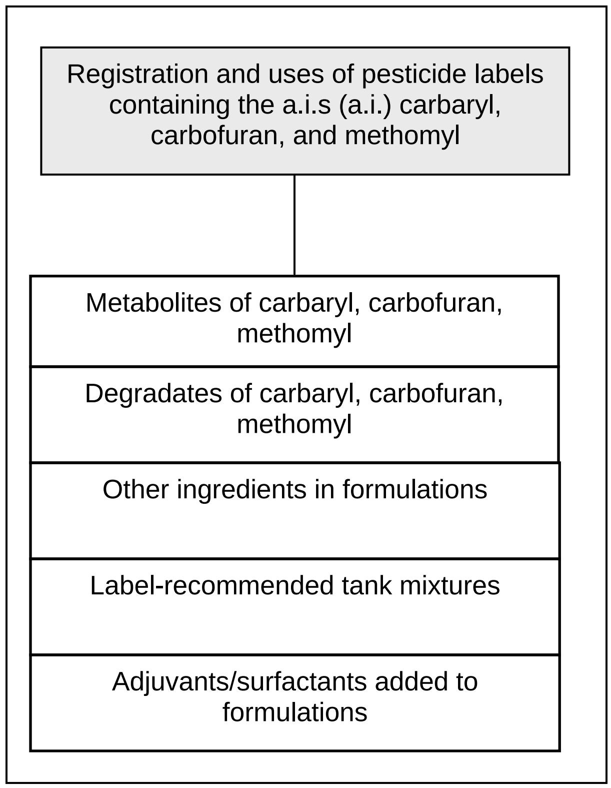


Figure 1. Stressors of the Action

Mode of Action of Carbamate Insecticides. NMC insecticides are neurotoxicants, affecting the central and peripheral nervous systems of animals. Similar to other carbamate and organophosphate (OP) insecticides, these a.i.s inhibit the enzyme acetylcholinesterase (AChE) found in brain and muscle tissue of invertebrates and vertebrates. Thus, NMCs belong to a class of insecticides known as AChE inhibitors. Inhibition of AChE results in a build-up of the neurotransmitter, acetylcholine, which can lead to continued stimulation. Normally, acetylcholine is broken down rapidly in the nerve synapse by AChE. Chemical neurotransmission and communication are impaired when acetycholine is not quickly degraded in animals, which ultimately may result in a number of adverse responses from physiological and behavioral modification to death.

NMFS batched the consultations on carbaryl, carbofuran, and methomyl into one Opinion because these compounds have the same mechanism of action, *i.e.*, they target the same site of action in the exact same way. However, NMFS evaluated the effects of each a.i. independently. Additionally, cumulative exposure to the three a.i.s is expected given they have overlapping uses and detections in surface water samples.

Active and Other Ingredients. Carbaryl, carbofuran, and methomyl are the a.i.s that kill or otherwise affect targeted organisms (listed on the label). However, pesticide products that contain these a.i.s also contain inert ingredients. Inert ingredients are ingredients which EPA defines as not "pesticidally" active. EPA also refers to inert ingredients as "other ingredients". The specific identification of the compounds that make up the inert fraction of a pesticide is not required on the label. However, this does not necessarily imply that inert ingredients are non-toxic, non-flammable, or otherwise non-reactive. EPA authorizes the use of chemical adjuvants to make pesticide products more efficacious. An adjuvant aides the operation or improves the effectiveness of a pesticide. Examples include wetting agents, spreaders, emulsifiers, dispersing agents, solvents, solubilizers, stickers, and surfactants. A surfactant is a substance that reduces surface tension of a system, allowing oil-based and water-based substances to mix more readily. A common group of non-ionic surfactants is the alkylphenol polyethoxylates (APEs), which may be used in pesticides or pesticide tank mixes, and also are used in many common household products. Nonylphenol (NP), one of the APEs, has been linked to endocrine-disrupting effects in aquatic animals.

Formulations. Pesticide products come in a variety of solid and liquid formulations. Examples of formulation types include dusts, dry flowables, emulsifiable concentrates, granulars, solutions, soluble powders, ultra-low volume concentrates, water-soluble bags, and powders. The formulation type can have implications for product efficacy and exposure to humans and other non-target organisms.

Tank Mix. A tank mix is a combination by the user of two or more pesticide formulations as well as any adjuvants or surfactants added to the same tank prior to application.

Typically, formulations are combined to reduce the number of spray operations or to obtain better pest control than if the individual products were applied alone. The compatibility section of a label may advise on tank mixes known to be incompatible or provide specific mixing instructions for use with compatible mixes. Labels may also recommend specific tank mixes. Pursuant to FIFRA, EPA has the discretion to prohibit tank mixtures. Applicators are permitted to include any combination of pesticides in a tank mix as long as each pesticide in the mixture is permitted for use on the application site and the label does not explicitly prohibit the mix.

Pesticide Registration. The Pesticide Registration Improvement Act (PRIA) of 2003 became effective on March 23, 2004. The PRIA directed EPA to complete REDs for pesticides with food uses/tolerances by August 3, 2006, and to complete REDs for all remaining non-food pesticides by October 3, 2008. The goal of the reregistration program is to mitigate risks associated with the use of older pesticides while preserving their benefits. Pesticides that meet today's scientific and regulatory standards may be declared "eligible" for reregistration. The results of EPA's reviews are summarized in RED documents. EPA issued REDs for carbaryl and methomyl in 2007 and 1998, respectively. The IRED for carbofuran was issued in August 2006. EPA considered the registration eligibility determination for carbofuran complete upon issuance of the cumulative assessment for the NMC pesticides in 2007 (http://www.epa.gov/pesticides/reregistration/REDs/carbofuran_red.pdf). Accordingly, EPA treated the carbofuran IRED as its RED. The REDs for all three a.i.s include various mitigation measures, including the cancellation of all registered uses of carbofuran due to ecological and occupational risks. These mitigation components were considered part of the proposed action.

Duration of the Proposed Action. EPA's goal for reassessing currently registered pesticide a.i.s is every 15 years. Given EPA's timeframe for pesticide registration reviews, NMFS' evaluation of the proposed action is also 15 years.

Interrelated and Interdependent Activities. No interrelated and interdependent activities are associated with the proposed action.

Registration Information of Pesticide A.i.s under Consultation. As discussed above, the proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing carbaryl, carbofuran, or methomyl. However, EPA did not provide copies of all product labels containing these a.i.s. The following descriptions represent information acquired from review of a sample of current product labels as well as information conveyed in the BEs, EPA REDs, and other documents.

Carbaryl

Carbaryl, also known by the trade name *Sevin*, is an NMC insecticide which was first registered in 1959 for use on cotton. In 2001, EPA identified the NMC insecticides as a group which shares a common mechanism of toxicity. Therefore, EPA was required to consider the cumulative effects on human health resulting from exposure to this group of chemicals when considering whether to establish, modify, or revoke a tolerance for pesticide residues in food, in accordance with the FQPA (EPA 2008).

Several regulatory documents concerning carbaryl were issued after EPA's BE of the analysis of risk of carbaryl to threatened and endangered salmonids (EPA 2003). An IRED for carbaryl that addressed the potential human health and ecological risks was signed on June 30, 2003. EPA amended the IRED on October 22, 2004, to incorporate clarifications and corrections, updated the residential risk assessment to reflect the voluntary cancellation of the liquid broadcast use of carbaryl on residential turf to address post-application risk to toddlers identified in the 2003 IRED, and addressed issues regarding labeling of carbaryl formulations for mitigating potential hazards to bees. In addition, mitigation measures required in the 2004 amended IRED included cancellation of certain uses and application methods, reduction of application rates, application prohibitions, personal protective equipment (PPE) and engineering control (EC) requirements, and extension of restricted-entry intervals (REIs) for post-application exposure (EPA 2008).

EPA also issued generic and product-specific data call-ins (DCIs) for carbaryl in March 2005. The carbaryl generic DCI required several studies for the a.i. carbaryl, including additional toxicology, worker exposure monitoring, and environmental fate data. The product DCI required acute toxicity and product chemistry data for all pesticide products containing carbaryl. In response to the 2005 DCIs, many carbaryl registrants chose to voluntarily cancel their carbaryl products. Approximately 80% of all of carbaryl end-use products registered at the time of the 2003 IRED have since been cancelled through this process or other voluntary cancellations (EPA 2008).

On September 26, 2007, EPA published a revised NMC cumulative risk assessment (EPA 2007), which concluded that the cumulative risks associated with the NMC pesticides meet the safety standard set forth in the FFDCA. Concurrently, on September 26, 2007, the RED for carbaryl was completed. The 2007 RED presents EPA's revised carbaryl human health risk assessment under FQPA and EPA's final tolerance reassessment decision for carbaryl. EPA amended the carbaryl RED in August of 2008. The amendment updated the 2007 RED to reflect the Revised Occupational Exposure and Risk Assessment, dated July 9, 2007 (EPA 2008).

The National Pesticide Information Retrieval System (NPIRS) website (http://ppis.ceris.purdue.edu/htbin/epachem.com) suggests that there are currently 24 registrants with active registrations of 87 pesticide products containing carbaryl. "Carbaryl is nationally registered for over 115 uses in agriculture, professional turf management, ornamental production, and residential settings (EPA 2007). Carbaryl is also registered for use as a mosquito adulticide. (http://www.umass.edu/fruitadvisor/NEAPMG/145-149.pdf)(EPA 2007)."

Several product labels indicate carbaryl is commonly formulated with other a.i.s. For example, there are active registrations of carbaryl products that also contain copper sulfate, rotenone, malathion, captan, metaldehyde, and bifenthrin (NPIRS website). According to EPA's BE, 26 carbaryl products are registered to individual states under

SLN provisions in Section 24(c) of FIFRA (EPA 2003). Section 24(c) registrations include control of shrimp in oyster beds in two tideland areas (Willapa Bay and Grays Harbor) in Washington and, in California, insecticidal use on fruits and nuts, prickly pear cactus, ornamental plants, and non-food crops. Idaho and Oregon do not have any 24(c) registrations for carbaryl (EPA 2003).

Usage Information.

The insecticide carbaryl is used in agriculture to control pests on terrestrial food crops including fruit and nut trees, many types of fruit and vegetables, and grain crops; cut flowers; nursery and ornamentals; turf, including production facilities; greenhouses; golf courses; and in oyster beds. Carbaryl is also registered for use on residential sites (*e.g.*, annuals, perennials, shrubs) by professional pest control operators and by homeowners on gardens, ornamentals, and turfgrass (EPA 2008). EPA estimated over 1.4 million pounds (lbs) of carbaryl are applied each year on agricultural crops and over 200,000 lbs are applied annually for turf, landscape, and horticultural uses in the U.S. (EPA 2008).

The California Department of Pesticide Regulation (CDPR) indicated approximately 150,000 – 250,000 lbs of carbaryl were applied annually in California between 2002 and 2006 based on agricultural and other "reportable uses" (CDPR 2007). The 1999 Oregon Legislature authorized development of the Oregon Pesticide Use Reporting System (PURS). In 2006, information on household pesticide use was collected through a pesticide use survey. The first full year of collecting non-household pesticide use in PURS was 2007. Over 37,000 lbs of carbaryl were reported as applied in Oregon in 2007 (ODA 2008). Approximately 189,600 lbs of carbaryl are used annually for agriculture in Washington State (WSDA 2004). Similar data on pesticide use were not found for Idaho.

Examples of Registered Uses.

***Agricultural Uses*.** Carbaryl is used on a myriad of crops. Examples of crops currently proposed for continued carbaryl use and which are grown in areas with Pacific salmon and steelhead include cranberries, cucumbers, beans, eggplant, grapefruit, grapes, hay, lemons, lettuce, nectarines, olives, onions, oranges, parsley, peaches, peanuts, pears, pecans, peppers, pistachios, plums, potatoes, prunes, pumpkins, rice, sod, spinach, squash, strawberries, sugar beets, sunflowers, sweet corn, sweet potatoes, tangelos, tangerines, tomatoes, walnuts, watermelons, wheat (EPA 2008). Carbaryl is also used to thin fruit in orchards to enhance fruit size and enhance repeat bloom.

*Non-agricultural Uses***.** Carbaryl is used extensively by homeowners, particularly for lawn care (EPA 2008). Examples of non-agricultural use sites include home and commercial lawns, flower beds around buildings, recreation areas, golf courses, sod farms, parks, rights-of-way, hedgerows, Christmas tree plantations, oyster beds, rural shelter belts, and applications to control ticks, grasshoppers, and adult mosquitoes (EPA 2007). Carbaryl is also used for pet care (pet collars, powders and dip, in kennels, and on pet sleeping quarters).

Examples of Registered Formulation Types. Carbaryl products are manufactured as granular, liquid, wettable powder, and dust formulations. All dry flowable (water dispersible granule) products have been voluntarily cancelled. The use of dust formulation in agriculture and backpack sprayers are not supported by Bayer CropScience, the carbaryl technical registrant, who is amending its carbaryl registrations to delete these uses (EPA 2008).

Methods and Rates of Application.

Methods. Groundboom, airblast, and aerial applications are typical for agricultural uses of carbaryl. Other applications can also be made using handheld equipment, such as low pressure hand wand sprayers, turf guns, and various ready-to-use products. Applications by aerosol cans, hand, spoon, shaker can, and front- and back-mounted spreaders are prohibited (EPA 2008).

Application Rates. The maximum single application rate allowed on the labels for agricultural uses in California, Idaho, Oregon, and Washington is 12 lb carbaryl/acre (Table 1). Many agricultural uses allow repeated application of carbaryl at intervals of 7-14 days. Application intervals are not specified for some uses (*e.g.*, flower beds, home use on fruits, and vegetables). Additionally, some uses do not specify the maximum number of applications (*e.g.*, prickly pear, ticks, and grasshoppers).

Table 1. Registered uses and application rates for carbaryl in California, Idaho, Oregon, and Washington (EPA 2007)

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
Home Lawn	9.1	2	7	Not specified
Fire ants	7.4	2	7	Not specified
Flower beds around buildings	8	4	None	6
Lawns, recreation areas, golf courses, sod farms, commercial lawns	8	2	7	8
Parks	8	1	-	8
Citrus	12	1	-	12
Citrus	7.5	8	14	20
Olives	7.5	2	14	Not specified
Almonds, chestnuts, pecans, filberts, walnuts, pistachio	5	4	7	15
Flowers, shrubs	4.3	3	7	Not specified
Apricot, cherries, nectarines, peaches, plums, prunes	4 (5 dormant)	3 + 1 dormant	15	9 + 5 dormant spray
Apple, pear, crabapple, oriental pear, loquat	3	8	14	Not specified
Sweet corn	2	8	3	Not specified
Caneberries, blueberries, grapes, strawberries	2	5	7	Not specified
Tomatoes, peppers, eggplant	2	7	7	8
Peanuts	2	5	7	8
Broccoli, cauliflower, cabbage, kohlrabi, Chinese cabbage, collards, kale, mustard greens, brussel sprouts, hanover salad	2	4	6	7
Sweet potato	2	8	7	8
Field corn, pop corn	2	4	14	8
Leaf lettuce, head lettuce, dandelion, endive, parsley, spinach, swiss chard	2	5	7	6
Celery, garden beets, carrots, horseradish, parsnip, rutabaga, potato, salsify, root turnip, radish	2	6	7	Not specified
Prickly pear	2	As needed	7	6
Rice	1.5	2	7	4
Fresh beans, dry beans, fresh peas, dry peas, cowpeas, fresh southern peas, soybeans	1.5	4	7	6
Sugar beets, pasture, grass for seed	1.5	2	14	3
Alfalfa, birdsfoot trefoil, clover	1.5	1/cutting	None	Not specified
Rangeland	1	1	-	1
Cucumber, melon, pumpkin, squash, roses, other herbaceous	1	6	7	Not specified

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
plants, woody plants				
CRP acreage, set-aside acreage, rights-of-way, hedgerows, ditch banks, roadsides, wasteland	1	2	14	3
Non-urban forests, tree plantations, Christmas trees, parks, rangeland trees, rural shelter belts	1	2	7	Not specified
adult mosquitoes[10]	1	*	*	Not specified
Ticks	2	As needed	None	Not specified
Grasshoppers	1.5	As needed	None	Not specified
flax	1.5	2	14	Not specified
Home fruits and vegetables	1.95	6	None	12.1
Proso millet, wheat	1.5	2	14	Not specified
lentils	1.5	4	7	Not specified
Oyster beds	8	Not specified	None	Not specified
Sunflower	1.5	2	7	Not specified
Tobacco	2	4	7	Not specified

*Number of applications and interval as specified for use site (pastures, rangeland, forests and wastelands, etc.).

Metabolites and Degradates.

The major metabolite of carbaryl degradation by both abiotic and microbially mediated processes is 1-naphthol. This degradate represented up to 67% of the applied carbaryl in degradation studies. It is also formed in the environment by degradation of naphthalene and other polyaromatic hydrocarbon compounds. EPA reports that only limited information on the environmental transport and fate of 1-naphthol is available and indicates this compound is less persistent and less mobile than the parent carbaryl (EPA 2003).

Carbofuran

Carbofuran is a NMC systemic pesticide first registered in the U.S. in 1969. The BE for carbofuran indicates it is registered as a restricted use broad spectrum insecticide, nematicide, and miticide for use on a wide variety of agricultural and non-agricultural crops (EPA 2004). Carbofuran is classified as a restricted use pesticide and is formulated into flowable, wettable powder, and granular forms. Through an agreement between EPA and the technical registrant in 1991, granular carbofuran has been limited to the sale of 2,500 lbs of a.i. per year in the U.S. since 1994, for use only on certain crops. Today granular carbofuran is limited to use on spinach grown for seed, pine seedlings, bananas (in Hawaii only), and cucurbits only (EPA 2006).

In the late 1990s, the technical registrant made a number of changes to labels in order to reduce human health (drinking water) and ecological risks of concern. These included reducing application rates and numbers of applications for alfalfa, cotton, corn, potatoes, soybeans,sugarcane, and sunflowers. Numbers of applications were also restricted on some soils to reduce groundwater concentrations (EPA 2006).

Several regulatory documents concerning carbofuran were issued after EPA's BE of the analysis of risk of carbofuran to threatened and endangered salmonids (EPA 2004). An IRED for carbofuran was published in August 2006. As previously indicated, EPA concluded the NMC cumulative risk assessment in September 2007. All tolerance reassessment and REDs for individual NMC pesticides were considered complete. The carbofuran IRED, therefore is considered a completed RED.

The carbofuran IRED (EPA 2006), and draft Notice of Intent to Cancel Carbofuran (January 2008) indicate EPA proposes cancellation of all uses of carbofuran, due to ecological, occupational, and human dietary risks of concern from some crops. Economic benefits are low to moderate for all of these uses, and do not outweigh the risks (EPA 2006). There are several uses for which residues do not pose dietary risks of concern *and* which have moderate benefits to growers [artichokes, chile peppers in the Southwestern U.S., cucurbits (granular formulation only), spinach grown for seed, sunflowers, and pine seedlings in the Southeastern U.S.]. For these uses, EPA is allowing a four-year phase-out in order to allow time for new alternatives to become available to growers (EPA 2006).

Although EPA determined that all uses of carbofuran are ineligible for reregistration, use of carbofuran will continue for an undetermined period of time. EPA has initiated cancellation procedures for product uses of low economic benefits. The remaining uses are subject to a four year phase-out. However, final cancellation of all carbofuran uses may take several years and the decision to cancel carbofuran registrations could be subject to legal challenges. Additionally, EPA indicated that FMC wishes to retain registrations for six uses: corn, potatoes, pumpkins, sunflowers, pine seedlings, and spinach grown for seed (Jones 2009). FMC also proposed to phase out use of artichokes over two years. EPA plans to consider FMC's proposal for the continued registration of carbofuran at a future date.

The IRED indicated there are currently one technical, two manufacturing-use, and six end-use products registered under Section 3 of FIFRA. There are also 77 active SLN registrations under Section 24(c) of FIFRA (EPA 2006). The NPIRS website suggests one registrant holds nine active registrations of pesticide products containing carbofuran (EPA registration numbers: 279-2712, 279-2862, 279-2874, 279-2876, 279-2922, 279-3023, 279-3038, 279-3060, and 279-3310). Many carbofuran uses and products were recently canceled through voluntary requests of the product registrant. Under the conditions of the cancelation order, existing stocks of these carbofuran products may be sold or used until they are depleted, opr until the effective date for the revocation of the associated tolerances (Federal Register/Vol. 74, No. 51/ March 18, 2009/11551-11553).

Usage Information

Nearly one million lbs a.i. are applied annually from the application of liquid carbofuran formulations (EPA 2006). The major use of liquid formulations of carbofuran is on corn, alfalfa, and potatoes. Under the existing terms and conditions of the registration, sale of the granular formulation is limited to 2,500 lbs a.i. per year, and use is limited to pine seedlings, cucurbits, bananas (in Hawaii only), and spinach grown for seed (EPA 2006). Carbofuran use has decreased significantly in California over the last decade. CDPR indicates agricultural uses of carbofuran exceeded 200,000 lbs in 1996 and use declined each preceding year with approximately 23,000 lbs of carbofuran applied in California in 2006 (CDPR 2007). Multiyear use statistics for describing temporal trends of carbofuran were unavailable for Idaho, Oregon, and Washington. It is anticipated that use of carbofuran will decrease across the action area as cancellation orders are implemented and existing stocks are depleted.

Examples of Registered Uses.

Food Crops. Alfalfa, artichoke, banana, barley, coffee, corn (field, pop, and sweet), cotton, cucurbits (cucumber, melons, and squash), grapes, oats, pepper, plantain, potato, sorghum, soybean, sugar beet, sugarcane, sunflower, and wheat (EPA 2006).

Non-food uses. Agricultural fallow land, cotton, ornamental and/or shade trees, ornamental herbaceous plants, ornamental non-flowering plants, ornamental woody shrubs and vines, pine, spinach grown for seed, and tobacco (EPA 2006).

Examples of Registered Formulation Types.

Carbofuran is formulated into flowable, wettable powder, and granular forms. The flowable formulation constitutes the vast majority of the carbofuran currently used (EPA 2004).

Examples of Approved Methods and Rates of Application.

Equipment. Carbofuran is applied by aerial equipment, chemigation systems, groundboom sprayers, airblast sprayers, tractor-drawn spreaders, push-type granular spreaders, and handheld equipment (EPA 2006).

Method and Rate. Carbofuran can be applied as a foliar or soil treatment. Maximum single and seasonal application rates range from 0.002 to 10 lbs a.i./acre, depending on the application scenario (EPA 2006).

Table 2. Registered uses and application rates for carbofuran in California, Idaho, Oregon, and Washington (EPA 2004)

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
Flowable Carbofuran – Section 3 registrations				
Alfalfa, corn, cotton	1	1	-	1
Ornamentals	0.06	Not specified	Not specified	0.06
Pine seedlings	0.05	1	-	Prepare slurry: Add 0.05 lbs a.i., 0.5 gallons water, and 2.0 lbs clay; Slurry sufficient to treat roots of 150 to 200 seedlings.
Potatoes	1	2	Not specified	2
Small grains, soybeans	0.25	2	Not specified	0.5
Sugarcane	0.75	2	Not specified	1.5
Sunflowers	1.4	1	-	1.4
Tobacco	6	Not specified	Not specified	6
Granular Carbofuran – Section 3 registrations				
Bananas	0.006	2	Not specified	0.012
Pine seedlings	0.002	Not specified	Not specified	0.002
Rice[1]	0.5	1	-	0.5
California- Flowable Carbofuran- Section 24C				
Artichokes	1	2	Not specified	2
Grapes	10	1	-	10
Ornamentals	10	Not specified	Not specified	10
Idaho- Flowable Carbofuran- Section 24C				
Potatoes	3	2	Not specified	6
Sugar beets	2	1	-	2
Oregon- Flowable Carbofuran- Section 24C				
Potatoes	3	2	Not specified	6
Nursery stock	10	Not specified	Not specified	10
Sugar beets	2	1	-	2
Oregon- Granular Carbofuran- Section 24C				
Watermelons	1	Not specified	Not specified	1
Washington- Flowable Carbofuran- Section 24C				
Potatoes	3	2	Not specified	6
Washington- Granular Carbofuran- Section 24C				
Spinach (grown for seed)	1	1	-	1

[1]The section 3 registration for rice was discontinued in 1997. Additional use of carbofuran on rice since that time has been from existing stock or in connection with emergency exemption requests (EPA 2004a).

Timing. Carbofuran is a contact insecticide applied at planting or post-planting (EPA 2006). The timing is variable among crops.

*Metabolites and Degradates***.** The major transformation product of carbofuran in water and aerobic aquatic metabolism is the hydrolysis product, carbofuran 7-phenol (EPA 2006). It also appears as the transformation endpoint prior to conversion to CO_2 and is shorter lived in water than the parent. Other major expected environmental transformation products in soils that have potential to reach the aquatic environment are 3-hydroxycarbofuran and 3-ketocarbofuran, which typically occur in small amounts (*i.e.*, < 5.0 % of applied) and are relatively short lived as compared to the parent (EPA 2006).

Methomyl

Methomyl was first registered for use in the U.S. in 1968. Methomyl is currently registered for use on a wide variety of sites including field, vegetable, and orchard crops; turf (sod farms only); livestock quarters; commercial premises; and refuse containers (EPA 2007). All methomyl products, except the 1% bait formulations, are classified as restricted use pesticides (EPA 2007). A Registration Standard issued in April 1989 required additional testing, modified tolerances. It also required label modifications related to applicator safety, re-entry intervals, and environmental hazards (EPA 2007). Additional label modifications were required with the publication of the methomyl RED in 1998 (EPA 1998).

EPA's BE of the analysis of risks of methomyl to threatened and endangered salmonids indicated there were 10 end-use products registered under Section 3 of FIFRA (EPA 2003). The NPIRS website suggests that there are currently six registrants with active registrations for nine products containing methomyl (EPA registration numbers: 270-255, 352-342, 352-361, 352-366, 352-384, 2724-274, 7319-6, 53871-3, 5742-2). Eighteen additional methomyl products are registered to individual states under SLN provisions in Section 24(c) of FIFRA (EPA 2003). California has seven SLN for use to control insects on ornamentals, beans, soybeans, radishes, sweet potatoes, Chinese broccoli, broccoli raab, and pumpkins (EPA 2003). Idaho, Oregon, and Washington do not have any SLNs for methomyl (EPA 2003). Methomyl also was previously registered as a molluscicide to control snails and slugs and as a fungicide for control of blights, rots, mildews and other fungal diseases. Those uses, as well as uses on ornamentals and in greenhouses, have been cancelled (EPA 2003).

Usage Information

The BE indicated EPA has no recent national data on the amount of methomyl applied annually (EPA 2003). According to the 1998 RED, an estimated 2.5 to 3.5 million lbs of methomyl a.i. were applied annually in the U.S. between 1987 and 1995. CDPR indicates approximately 262 – 554 thousand lbs of methomyl were applied annually in California between 2000 and 2006 based on agricultural and other “reportable uses” (CDPR 2007). Over 42,000 lbs of methomyl were applied in Oregon in 2007 (ODA 2008). Similar data on pesticide use were not found for Idaho and Washington.

Examples of Registered Uses

Agriculture. Methomyl is used for a variety of agricultural uses including alfalfa, anise, asparagus, barley, beans (succulent and dry), beets, Bermuda grass (pasture), blueberries, broccoli, broccoli raab, Brussels sprouts, cabbage, carrot, cauliflower, celery, chicory, Chinese broccoli, Chinese cabbage, collards (fresh market), corn (sweet), corn (field and popcorn), corn (seed), cotton, cucumber, eggplant, endive, garlic, horseradish, leafy green vegetables, lentils, lettuce (head and leaf), lupine, melons, mint, nonbearing nursery stock (field grown), oats, onions (dry and green), peas, peppers, potato, pumpkin, radishes, rye, sorghum, soybeans, spinach, sugar beet, summer squash, sweet potato, tomatillo, tomato, turf (sod farms only), wheat, and orchards including apple, avocado, grapes, grapefruit, lemon, nectarines, oranges, peaches, pomegranates, tangelo, and tangerine (EPA 2007).

Non-agriculture. Methomyl has several non-crop uses that are outside uses involving scatter bait or bait station formulations including the following use sites: bakeries, beverage plants, broiler houses, canneries, commercial dumpsters which are enclosed, commercial use sites (unspecified), commissaries, dairies, dumpsters, fast food establishments, feedlots, food processing establishments, hog houses, kennel, livestock barns, meat processing establishments, poultry houses, poultry processing establishments, restaurants, supermarkets, stables, and warehouses (EPA 2007).

Examples of Registered Formulations and Types.

End-use formulations of methomyl include soluble concentrate, wettable powder, granular, pelleted/tableted, and water soluble packaged. Products registered as fly baits also contain (Z)-9-tricosene (0.04 to 0.26% a.i.) as an a.i.; labels note that these products contain a sex attractant and feeding synergist (EPA 2003).

Examples of Approved Methods and Rates of Application.

Application Equipment. Methomyl can be applied by aircraft; bait box; brush; cup; duster; glove; granule applicator; ground; high volume ground sprayer; low volume ground sprayer; package applicator; scoop; shaker can; shaker jar; sprayer; and ultra low volume sprayer (EPA 1998).

Application Rates. The maximum single application rate allowed on the labels for agricultural uses in California, Idaho, Oregon, and Washington is 0.9 lb a.i./acre. Many agricultural uses allow repeated application of methomyl at relatively short intervals (1-5 days). For example, the application interval for methomyl for sweet corn is one day and methomyl can be applied 28 times within a single crop of sweet corn. Additionally, several crops of sweet corn may be grown per year in some locations within the action area (EPA 2007). The maximum seasonal labeled application rates (indicated on the label as maximum application rates per crop) for agricultural uses range from 0.9 lb a.i./acre/crop [*i.e.*, Bermuda grass (pasture), avocado, lentils, beans (interplanted with trees), sorghum, and soybeans (interplanted with trees)] to 7.2 lbs a.i./acre/crop [*i.e.*, cabbage, lettuce (head), cauliflower, broccoli raab, celery, and Chinese cabbage]. Several methomyl crops can be grown more than one time per year (*i.e.*, they have multiple crop cycles). Therefore, for those methomyl uses that have more than one crop cycle per year, the maximum allowable yearly application rate will be higher than the maximum seasonal application rate. For perennial crops (*e.g.*, alfalfa), the number of cuttings per year was used to determine the number of crop cycles per year. Based on the labeled application rates and information from EPA's OPP Benefits and Economic Analysis Division (BEAD) on the number of times each crop for which methomyl is registered for use can be grown in California, the maximum yearly application rates for methomyl are 32.4 lb a.i./acre/year (alfalfa) and 21.6 lb a.i./acre/year (broccoli raab, cabbage, and Chinese cabbage) for agricultural crops; 5.4 lb a.i./acre/year (peaches) for orchards; and 0.22 lb a.i./acre/application for nonagricultural uses (no maximum application/acre/year is provided on the nonagricultural use labels). All orchard and most agricultural uses involve foliar application. The only granular agricultural/orchard use is for corn which also has a foliar use (EPA 2007).

All non-agricultural outside uses for methomyl in California, Idaho, Oregon, and Washington are limited to scatter baits and bait stations around agricultural (*e.g.*, animal premises) and commercial structures and commercial dumpsters, where children or animals are not likely to contact the pesticide. The scatter bait can also be mixed with water to form a paste which can be brushed onto walls, window sills, and support beams. The maximum application rate for the scatter bait use is 0.22 lb a.i./acre (0.0025 lb a.i./500 ft^2). However, it is unlikely that applications would involve a full acre as the outside use of the scatter bait is limited to areas around structures and dumpsters. No minimum application interval or maximum application rate per year is provided on the scatter bait labels (EPA 2007).

Table 3. Examples of registered uses and application rates for methomyl in California, Idaho, Oregon, and Washington (EPA 2007).

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
Commercial dumpsters; poultry houses; unspecified commercial sites; outside commercial uses: feedlots, dairies, stables, broiler houses, hog houses, livestock barns, meat processing establishments, poultry processing establishments, beverage plants, canneries, food processing establishments, kennels, dumpsters, restaurants, supermarkets, commissaries, and bakeries.	0.22	Not specified	1-3	Not specified
Alfalfa	0.9	10 x 9 crops	5	32.4
Asparagus	0.9	5	7	4.5
Avocado	0.9	2	5	0.9
Barley	0.45	4	5	1.8
Beans	0.9	10	5	4.5
Broccoli	0.9	10 x 3 crops	5	21.6
Cabbage	0.9	15 x 3 crops	2	21.6
Corn	0.45	28 x 3 crops	1	18.9
Lettuce	0.9	8 x 2 crops	2	14.4

Use Site	Application			
	Maximum Single Application Rate (lb a.i./acre)	Maximum Number of Applications	Minimum Application Interval (days)	Maximum per year (lb a.i./acre)
Onions	0.9	8 x 3 crops	5	16.2
Spinach	0.9	8 x 3 crops	5	10.8
Turf	0.9	4 x 2 crops	5	7.2

Timing. The timing of application is dependent on use, but may occur throughout the year. In most cases multiple applications are allowed to maintain pest control. Applications occur on fruit crops during the bloom, petal fall, pre-bloom, and leaf stages and when pest pressure is highest on a "When Needed" basis. On corn, application may occur during the whorl/foliar stages. With other crops, application is during the foliar or leaf stages of the crop (EPA 1998).

Metabolites and Degradates

Several degradates and metabolites have been identified for methomyl. The major degradate in most metabolism studies was CO^2. Another degradate, S-methyl-N-hydroxythioacetamidate, which is highly mobile, appears primarily as a product of alkaline hydrolysis. In an aquatic metabolism study, methomyl degraded with estimated half-lives of four to five days. After seven days, acetonitrile comprised a maximum of 17% and acetamide up to 14% of the amount of methomyl applied. After 102 days, volatilized acetonitrile totaled up to 27% of the applied and CO_2 up to 46% of the applied material (EPA 2003).

Species Addressed in the BEs

EPA's BEs considered the effects of carbaryl, carbofuran, and methomyl to 26 species of listed Pacific salmonids and their designated critical habitat. EPA determined that carbaryl, carbofuran, and methomyl may affect most of these species. Exceptions follow:

EPA concluded that the registration of carbaryl products would have no effect on Northern California steelhead, SONCCal coho, Hood Canal summer-run chum, and Ozette Lake sockeye. EPA also concluded that the registration of carbaryl products would not likely adversely affect California Coastal Chinook and Puget Sound Chinook salmon.

EPA concluded that the registration of carbofuran products would have no effect on Northern California Steelhead, Central California Coast coho, and California Coastal Chinook salmon. EPA also concluded that the registration of carbofuran products would not likely adversely affect Central Valley Spring-run Chinook salmon, Lower Columbia River Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum, Hood Canal summer-run chum, Oregon Coast coho, SONCC coho, Ozette Lake sockeye, Snake River sockeye, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Snake River steelhead, South-Central California steelhead, Southern California steelhead, and Upper Willamette River steelhead.

EPA concluded that the registration of methomyl products would have no effect on Northern California steelhead and California Coastal Chinook salmon.

Although EPA has determined that its action in registering pesticides containing the three a.i.s is not likely to adversely affect certain ESUs/DPSs and will have no effect on others, EPA initiated formal consultation on its action because EPA concluded that its action may adversely affect other listed ESUs/DPSs. When an action agency concludes that its action will not affect any listed species or critical habitat, then no section 7 consultation is necessary (USFWS and NMFS 1998). If NMFS concurs with a federal agency that its action is not likely to adversely affect any listed species or critical habitat, then formal consultation is not required. Since formal consultation was triggered, NMFS evaluated the federal action and its impacts to all listed Pacific, anadromous salmonids and their designated critical habitat. In this Opinion, NMFS will analyze the impacts to all ESUs/DPSs of Pacific salmonids present in the action area, including those salmonid species identified by EPA as being unaffected or not likely to be adversely affected including two species of salmonid listed after EPA provided its BEs.

Approach to this Assessment

Overview of NMFS' Assessment Framework

NMFS uses a series of steps to assess the effects of federal actions on endangered and threatened species and designated critical habitat. The first step of our analysis identifies those physical, chemical, or biotic aspects of proposed actions that are likely to have individual, interactive, or cumulative direct and indirect effects on the environment (we use the term "potential stressors" for these aspects of an action). As part of this step, we identify the spatial extent of any potential stressors and recognize that the spatial extent of those stressors may change with time. The spatial extent of these stressors is the "action area" for a consultation.

The second step of our analyses identifies the listed resources (endangered and threatened species and designated critical habitat) that are likely to occur in the same space and at the same time as these potential stressors. If we conclude that such co-occurrence is likely, we then try to estimate the nature of co-occurrence (these represent our *Exposure Analyses*). In this step of our analysis, we try to identify the number, age (or life stage), gender, and life history of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent.

Once we identify which listed resources are likely to be exposed to potential stressors associated with an action and the nature of that exposure, in the third step of our analysis we examine the scientific and commercial data available to determine whether and how those listed resources are likely to respond given their exposure (these represent our *Response Analyses*). We integrate the exposure and response analyses to assess the risk to listed individuals and their habitat from the stressors of the action (these represent our *Risk Characterization*). NMFS' analysis is ultimately a qualitative assessment that draws on a variety of quantitative and qualitative tools and measures to address risk to listed resources.

In the final steps of our analyses, we establish the risks posed to listed species and to designated critical habitat. Our jeopardy determinations for listed species must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. Because the continued existence of listed species depends on the fate of the populations that comprise them, the viability (that is, the probability of extinction or probability of persistence) of listed species depends on the viability of the populations that comprise the species. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them; populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

The structure of our risk analyses reflects the relationships between listed species, the populations that comprise each species, and the individuals that comprise each population. Our risk analyses begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrates those individual-level effects to identify consequences to the populations those individuals represent. Our analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

We evaluate risks to listed individuals by measuring the individual's "fitness" defined as changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable response to an action's effect on the environment (which we identify in our *Response Analyses*) are likely to have consequences for the individual's fitness.

Reductions in abundance, reproduction rates, or growth rates (or increased variance in one or more of these rates) of the populations those individuals represent is a *necessary* condition for reductions in a population's viability, which is itself a *necessary* condition for reductions in a species' viability. On the other hand, when listed plants or animals exposed to an action's effects are *not* expected to experience reductions in fitness, we would not expect that action to have adverse consequences on the viability of the population those individuals represent or the species those populations comprise (Mills and Beatty 1979; Stearns 1982; Anderson, Phillips et al. 2006). If we conclude that listed species are *not* likely to experience reductions in their fitness, we would conclude our assessment because an action that is not likely to affect the fitness of individuals is not likely to jeopardize the continued existence of listed species.

If, however, we conclude that listed plants or animals are likely to experience reductions in their fitness, our assessment determines if those fitness reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population's extinction risks). In this step of our analyses, we use the population's base condition (established in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion) as our point of reference. Finally, our assessment determines if changes in population viability are likely to be sufficient to reduce the viability of the species those populations comprise.

Critical habitat analysis focuses on reductions in the quality or quantity of PCEs. The stressors of the action for this Opinion are chemicals and PCEs potentially affected are salmonid prey availability and degradation of water quality in freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas. Endpoints evaluated for the prey PCE include prey survival, prey growth, prey drift, prey reproduction, abundance of prey, health of invertebrate aquatic communities, and recovery of aquatic communities following pesticide exposure. Degradation of water quality was evaluated by considering the information available on the presence of constituents known to adversely affect aquatic organisms (*e.g.*, toxic chemicals, nutrients, sediments), whole effluent test or toxicity indicator evaluations, and/or instances of waterbodies not meeting local, state, or federal water quality criteria.

Evidence Available for the Consultation

We search, compile and use a variety of resources to conduct our analyses including:

- EPA's BEs, REDs, IREDS, other documents developed by EPA
- Peer-reviewed literature
- Gray literature
- Books
- Available pesticide labels
- Any correspondence (with EPA or others)
- Available monitoring data and other local, county, and state information
- Pesticide registrant generated data
- Online toxicity databases (PAN, EXTOXNET, ECOTOX, USGS, NPIC)
- Pesticide exposure models run by NMFS
- Population models run by NMFS
- Information and data provided by the registrants identified as applicants
- Comments on the draft Opinion from EPA and any applicants
- Incident reports

Collectively, this information provided the basis for our determination as to whether and to what degree listed resources under our jurisdiction are likely to be exposed to EPA's action and whether and to what degree the EPA can ensure that its authorization of pesticides is not likely to jeopardize the continued existence of threatened and endangered species or is not likely to result in the destruction or adverse modification of designated critical habitat.

Application of Approach in this Consultation

The EPA proposes to authorize the use of over 100 pesticide formulations (pesticide products) containing the a.i.s carbaryl, carbofuran, and methomyl through its authority to register pesticides under FIFRA. Registration by EPA authorizes the use of these formulations in the U.S. and its territories, documented by EPA's approval of registrant-derived pesticide labels. Pursuant to the court's 2002 order in Washington Toxics Coalition v. EPA, EPA initiated consultation on registration of carbaryl, carbofuran, and methomyl for 26 listed ESUs of Pacific salmonids. Since EPA initiated consultation, NMFS has listed one additional Pacific coho ESU and one additional Pacific steelhead DPS. This Opinion represents NMFS' evaluation of whether EPA's authorization of these labels satisfies EPA's obligations to listed salmonids pursuant to section 7(a)(2) of the ESA.

The NMFS evaluates whether endangered species, threatened species, and designated critical habitat are likely to be exposed to the direct and indirect effects of the proposed action. If those listed resources are not likely to be exposed to these activities, we would conclude that EPA's action is not likely to jeopardize the continued existence of threatened species, endangered species, or result in the destruction or adverse modification of designated critical habitat under NMFS' jurisdiction. If, however, listed individuals are likely to be exposed to these actions and individual fitness is reduced, then we evaluate the potential for population-level consequences.

A Viable Salmonid Population (VSP) is an independent population of any Pacific salmonid that has a negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a 100-year time frame (McElhaney, Ruckleshaus et al. 2000). The independent population is the fundamental unit of evaluation in determining the risk of extinction of salmon in an ESU. Attributes or metrics associated with a VSP include the abundance, productivity, spatial structure, and genetic diversity of the population. Abundance is defined as the size of the population and can be expressed in a number of ways, *e.g.*, the number of spawning adults, the number of adults surviving to recruit to fisheries, or the number of emigrating smolts. Abundance is a vital measure, as smaller populations run a greater risk of extinction. The second VSP measure is productivity, generally defined as the growth rate of a population. This Opinion discusses productivity in terms of lambda (λ). Appendix 1 contains a more detailed explanation of λ in the context of our population models. The spatial structure of a population is inherently dependant on the quantity and quality of available habitat. A limited spatial structure can hamper the ability of the ESU to respond to evolutionary pressures. Genetic variability within the ESU gives the species the ability to respond to short-term stochastic events, as well as to evolve to a changing environment in the long-term. These VSP parameters provide an indication of the population's capacity to adapt to various environmental conditions and ability to be self-sustaining in the natural environment (McElhaney, Ruckleshaus et al. 2000; McElhaney, Chilcote et al. 2007).

In determining the effect of an action to populations, we first address whether individual fitness level consequences are likely and whether those consequences affect populations. We evaluate whether identified VSP parameters of populations such as abundance and productivity are reduced by individual fitness effects. If populations are likely to be adversely affected by reductions in VSP parameters, we analyze the potential effects to the species as a whole. In parallel, if designated critical habitats are likely to be exposed and PCEs are adversely affected, then we evaluate the potential for reductions in the conservation value of the habitats. We devise risk hypotheses based on identified PCEs that are potentially affected by the stressors of the action. If the best available data indicate that PCE-specific risk hypotheses are supported, then we discuss whether critical habitat will remain functional to serve the intended conservation role for the species in the *Conclusion* section.

General conceptual framework for assessing risk of EPA's pesticide actions to listed resources.

We evaluate the risk to listed species and designated critical habitat in the *Effects of the Proposed Action* section by applying an ecological risk assessment framework that organizes the available information in three phases: problem formulation, analysis, and risk characterization (EPA 1998; McElhaney, Ruckleshaus et al. 2000). We adapted the EPA framework to address ESA-specific considerations (Figure 2). The framework follows a process for organizing, evaluating, and synthesizing the available information on listed resources and the stressors of the action. Below, we briefly describe each phase in the *Effects of the Proposed Action* section.

Problem Formulation

The first phase of the framework is problem formulation. In this phase, we generate conceptual models from our initial evaluation of the relationships between stressors of the action (pesticides and identified chemical stressors and potential receptors (listed species, habitat). We represent these relationships in conceptual models presented as diagrams and written risk hypotheses (EPA 1998). Conceptual model diagrams are constructed to illustrate potential pesticide exposure pathways and associated listed resources' responses. An example of a conceptual model is presented in Figure 3 for Pacific salmonids. In it, we illustrate where the pesticides generally reside in the environment following application, how pesticides may co-occur with listed species and their habitats, and how the individuals/habitat may respond upon exposure to them. In the case of Pacific salmonids, we ascribe exposure and response to specific life stages of individuals and then assess individual fitness endpoints sensitive to the action's stressors.

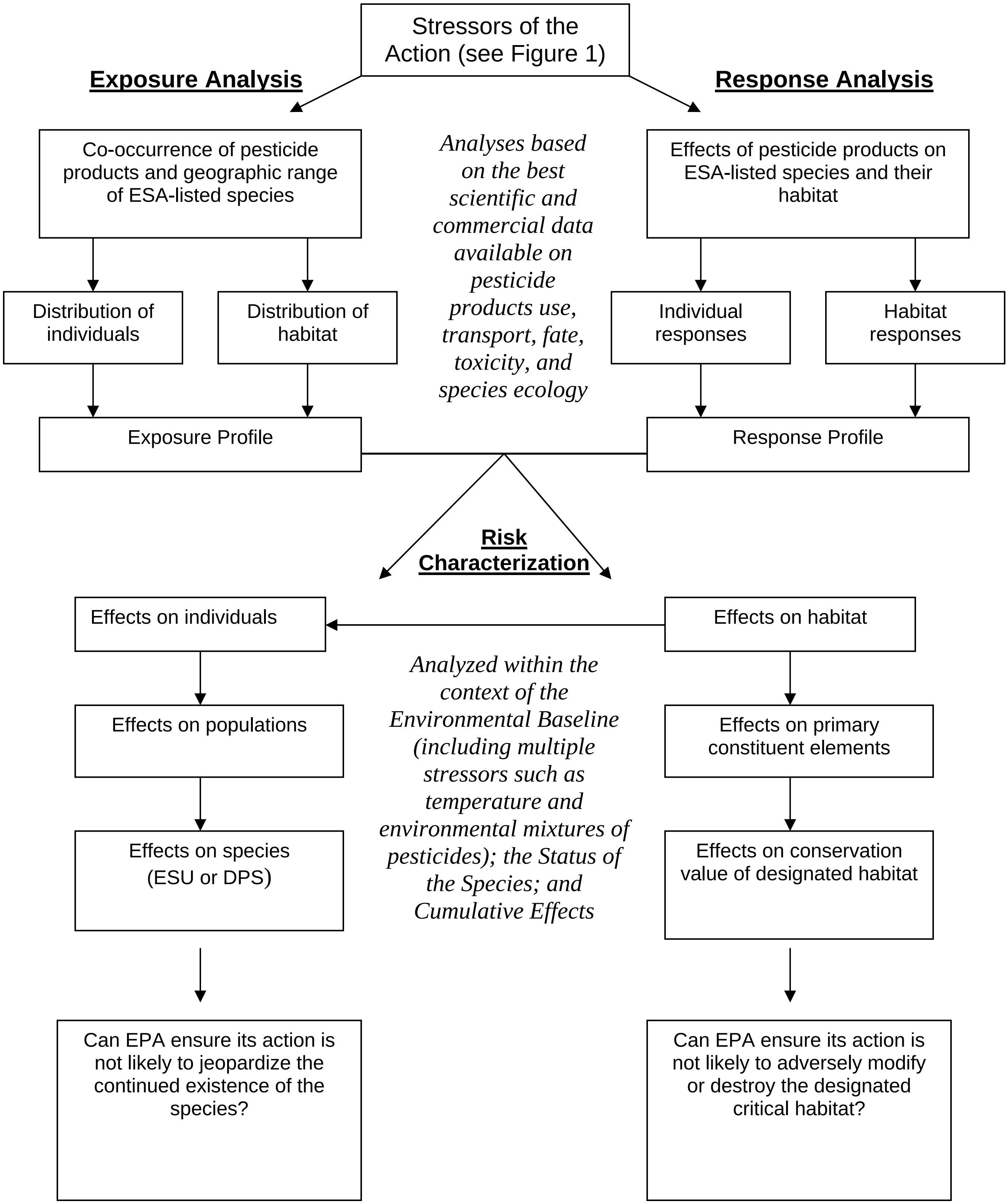


Figure 2. Conceptual framework for assessing risks of EPA's action to listed resources

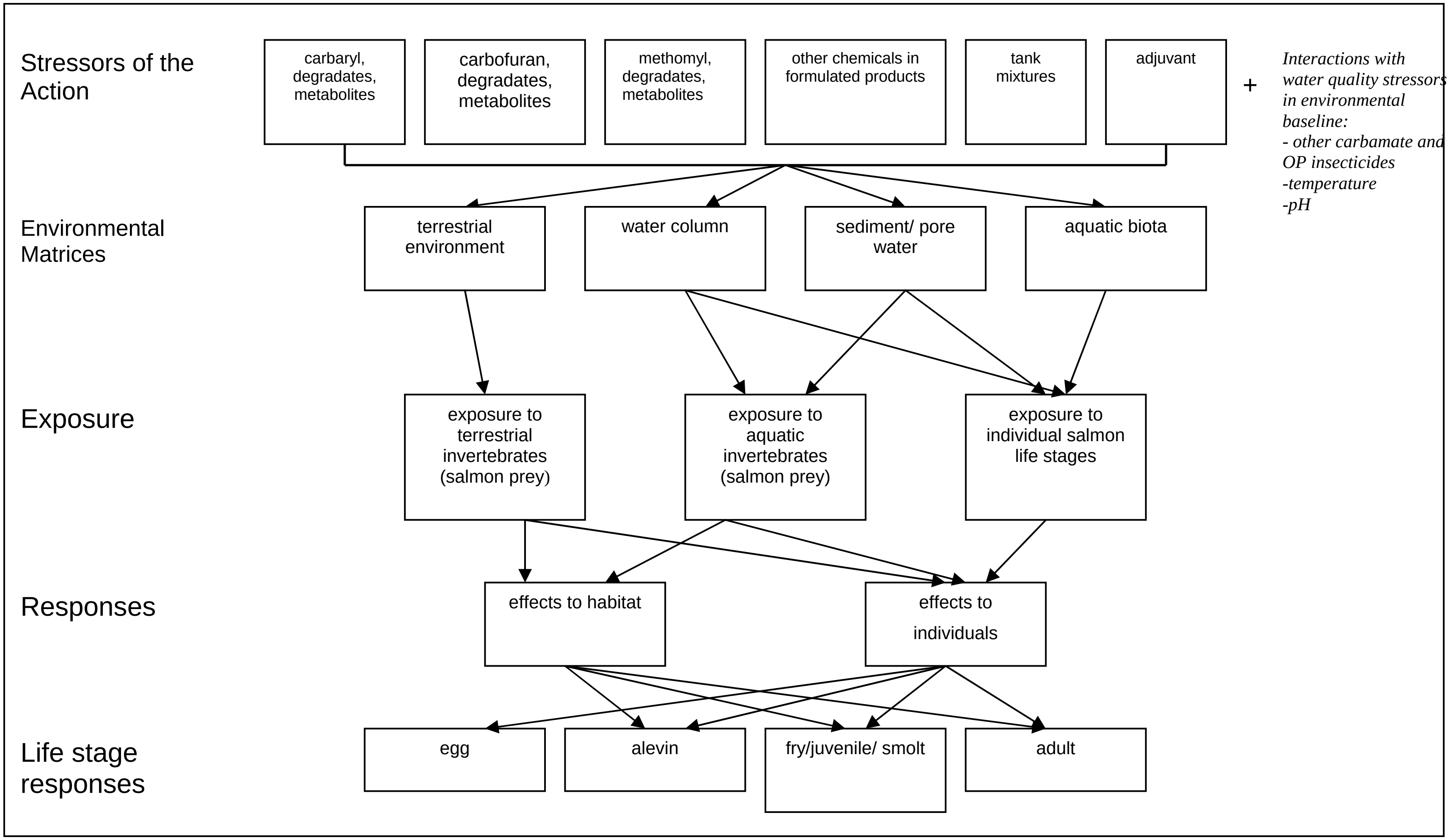


Figure 3. Exposure pathways to carbaryl, carbofuran, and methomyl and general responses of listed Pacific salmonids and habitat.

Species Risk Hypotheses

We construct risk hypotheses by identifying biological requirements or assessment endpoints (Table 4) for listed resources in the action area. We designate assessment endpoints as those biological properties of species and their habitat essential for successful completion of a species life cycle. We integrate the listed resources information with what is known about the stressors of the action, including their physical properties, use, presence in aquatic habitats, and their toxicity. We then evaluate how listed salmonids and their habitat are potentially affected by the stressors of the action and integrate this information with exposure information to develop risk hypotheses. Below are the risk hypotheses (written as affirmative statements) we evaluate in the *Effects of the Proposed Action* section:

1. Exposure to carbaryl, carbofuran, and methomyl is sufficient to:
    a. Kill salmonids from direct, acute exposure;
    b. Reduce salmonid survival through impacts to growth;
    c. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey;
    d. Impair swimming which leads to reduced growth (via reductions in feeding), delayed and interrupted migration patterns, survival (via reduced predator avoidance), and reproduction (reduced spawning success); and
    e. Reduce olfactory-mediated behaviors resulting in consequences to survival, migration, and reproduction.
2. Exposure to mixtures of carbaryl, carbofuran, and methomyl can act in combination to increase adverse effects to salmonids and salmonid habitat.
3. Exposure to other stressors of the action including degradates, adjuvants, tank mixtures, and other active and other ingredients in pesticide products containing carbaryl, carbofuran, and methomyl cause adverse effects to salmonids and their habitat.
4. Exposure to other pesticides present in the action area can act in combination with carbaryl, carbofuran, and methomyl to increase effects to salmonids and their habitat.
5. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

We discuss an example of one risk hypothesis to show the relationship between assessment endpoints and measures with species responses. In risk hypothesis 1 (d), aquatic exposure to carbaryl, carbofuran, and methomyl can impair a salmonid's nervous system and consequently affect swimming ability of fish. Behavioral modifications, such as changes in swimming performance, are regularly considered in NMFS' Opinions. Swimming performance therefore is an assessment endpoint. Measurable changes in swimming speed are the assessment measure used to evaluate this endpoint. Reductions in swimming performance could also affect other assessment endpoints such as migration and predator avoidance. We may or may not have empirical data that address these endpoints, resulting in a recognized data gap. This uncertainty would be identified during the problem formulation phase, and discussed in the risk characterization phase.

Critical Habitat Risk Hypotheses:

To determine potential effects to designated critical habitat, NMFS evaluates the effects of the action by first looking at the effects on PCEs of critical habitat. Effects to PCEs include changes to the functional condition of salmonid habitat caused by the action in the action area. Properly functioning salmonid PCEs are important to the conservation of the ESU/DPS. The stressors of the action for this Opinion are chemicals. As such, the key PCEs that are potentially affected are salmonid prey availability and degradation of water quality in freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas. We developed two risk hypotheses based on these PCEs:

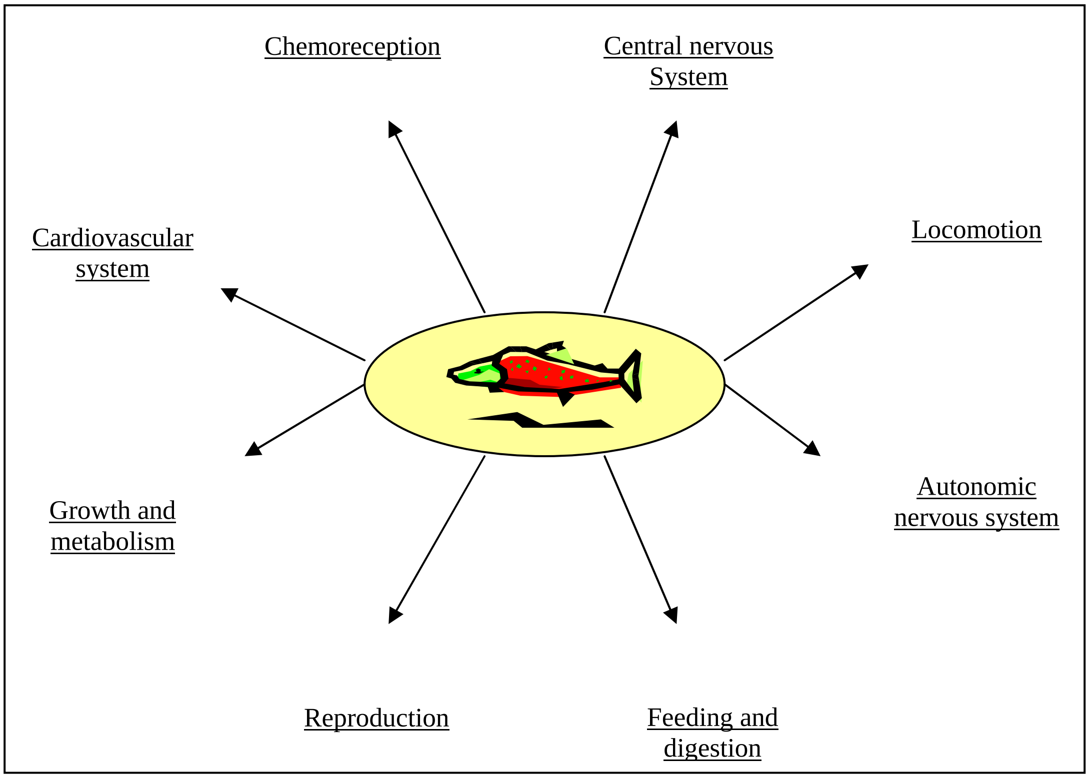
1. Exposure to the stressors of the action is sufficient to reduce abundance of aquatic prey items of salmonids; and
2. Exposure to the stressors of the action is sufficient to degrade water quality in designated critical habitat.

These hypotheses are evaluated using the best scientific and commercial data available presented in the *Response* section. Examples of assessment endpoints evaluated include prey survival, prey growth, prey drift, prey reproduction, abundance of prey, health of invertebrate aquatic communities, recovery of aquatic communities following pesticide exposure, etc. If the available evidence supports the risk hypotheses, then NMFS evaluates whether the potential reductions in PCEs are localized or widespread. The potential reduction of PCEs affect on the conservation value of designated critical habitats is then assessed. This portion of the analysis is conducted in the *Integration and Synthesis* section.

Below we discuss an example of one risk hypothesis to show the relationship between assessment endpoints and measures with species responses. In risk hypothesis 1 (d), aquatic exposure to carbaryl, carbofuran, and methomyl can impair a salmonid's nervous system and consequently affect swimming ability of fish. Behavioral modifications, such as changes in swimming performance, are regularly considered in NMFS' Opinions. Swimming performance therefore is an assessment endpoint. Measurable changes in swimming speed are the assessment measure used to evaluate this endpoint. Reductions in swimming performance could also affect other assessment endpoints such as migration and predator avoidance. We may or may not have empirical data that address these endpoints, resulting in a recognized data gap. This uncertainty would be identified during the problem formulation phase, and discussed in the risk characterization phase.

In the problem formulation phase, we also identify the toxic mode and mechanism of action of chemical stressors, particularly for the pesticide a.i.s. This information helps us understand what an organism's physiological consequences may be following exposure. It also helps us evaluate whether mixture toxicity occurs because we identify other pesticides that share similar modes of action and the likelihood for co-occurrence in listed species habitats. A similar mode of action with other pesticides is a key determinant of the likelihood of mixture toxicity. With vertebrates (fish and mammals) and invertebrates, the three a.i.s share a common mode and mechanism of action, acetylcholinesterase inhibition. Given this information, a range of potential adverse responses are possible (Figure 4). We then search, compile, and review the available toxicity information to ascertain which physiological systems are known to be affected and to what degree.

Figure 4. Physiological systems potentially affected by acetylcholinesterase inhibition



In Table 4, assessment endpoints and assessment measures are identified for particular life stages. We focused on the following physiological systems identified in Figure 4: chemoreception, locomotion, feeding, reproduction, and growth. We did not locate any information on the remaining systems in Figure 4. Thus, they were not specifically addressed in our analysis.

We assess the likelihood of these fitness level consequences occurring from exposure to the action. In the exposure analysis (Figure 2), we select exposure estimates for our listed resources derived from reviewing the available exposure data. Depending on the chemicals being evaluated, data may or may not be available for all endpoints and measures, and available data may vary in reliability. Thus, we use a weight-of-evidence approach in this Opinion.

The problem formulation phase as articulated in EPA's 1998 Guidelines for Conducting Ecological Risk Assessment concludes with the development of an analysis plan. In this Opinion, the *Approach to the Assessment* section is the general analysis plan. This section identifies how exposure will be assessed and which assessment endpoints will be evaluated. Therefore, the *Approach to the Assessment* is a road map for evaluating the effects of EPA's registration actions with carbaryl, carbofuran, and methomyl.

Table 4. Examples of salmonid lifestage assessment endpoints and measures

Salmonid Life Stage	Assessment Endpoint (individual fitness)	Assessment Measure (measures of changes in individual fitness)
Egg* * Is the egg permeable to pesticides (measured by pesticide concentrations in eggs)?	Development Survival	size, hatching success, morphological deformities viability (percent survival)
Alevin (yolk-sac fry)	Respiration Swimming: predator avoidance site fidelity Yolk-sac utilization: growth rate size at first feeding Development Survival	gas exchange, respiration rate swimming speed, orientation, burst speed predator avoidance assays rate of absorption, growth weight and length weight and length morphology, histology LC50 (dose-response slope). Percent dead at a given concentration
Fry, Juvenile, Smolt	First exogenous feeding (fry)– post yolk-sac absorption Survival Growth Feeding Swimming: predator avoidance behavior migration use of shelter Olfaction: kin recognition predator avoidance imprinting feeding Smoltification (smolt) Development	time to first feeding, starvation LC50 (dose-response slope). Percent dead at a given concentration weight, length stomach contents, weight, length, starvation, prey capture rates swimming speed, orientation, burst swimming speed predator avoidance assays swimming rate, downstream migration fish monitoring, bioassays electro-olfactogram measurements, behavioral assays behavioral assays behavioral assays behavioral assays Na/K ATPase activity, sea water challenge tests length, weight, malformations
Returning adult	Survival Feeding Swimming: predator avoidance migration spawning feeding Sexual development Olfaction: Predator avoidance Homing Spawning	LC50 (dose-response slope). Percent dead at a given concentration stomach contents behavioral assays numbers of adult returns, behavioral assays numbers of eggs fertilized stomach contents histological assessment of ovaries/testis electro-olfactogram measurements, measurements of intersex behavioral assays behavioral assays behavioral assays

Risk Characterization

We follow the framework presented in Figure 2 to conduct the analysis and risk characterization phases. First we conduct exposure and response analyses to estimate/determine the type, likelihood, magnitude, and frequency of adverse responses resulting from predicted exposure based on the best available information. We evaluate species information and pesticide information to determine when, where, and at what concentrations listed salmonids and their habitat may be exposed. We then correlate those exposure estimates with probable response based on available toxicity data. Once we have conducted the analysis phase, we move to the risk characterization phase (Figure 2).

In the risk characterization phase, we revisit the risk hypotheses and apply tools to address whether any individual fitness consequences assessed in the analysis phase would be expected to impact populations and ultimately species. One of the tools we employ is individual-based population models predicated on a juvenile salmonids' probability of survival in its first year of life. We also assess interactions between the stressors of the action and stressors in the *Environmental Baseline* (Figure 2). Some pesticides' toxicity profiles are influenced by environmental parameters such as pH and temperature. Temperature can affect pesticide metabolism in fish and is seasonally elevated in many salmonid supporting watersheds. As described earlier in this section we translate expected effects to identified PCEs by evaluating the available information to support risk hypotheses. If we expect PCEs to be reduced we discuss whether the expected reductions translate to reductions in the conservation value of designated critical habitat.

To conclude consultation, cumulative effects are described and the extent to which species and habitat are affected is documented. Cumulative effects as defined in 50 CFR §404.2 include the effects of future, state, tribal, local, or private actions that are reasonably certain to occur in the action area of this Opinion. Integrating the *Effects of the Proposed Action,* the *Status of Listed Resources,* and the *Environmental Baseline,* NMFS determines whether EPA's pesticide registration action jeopardizes the continued existence of the species. NMFS also determines whether the action results in the destruction or adverse modification of designated critical habitat.

Other Considerations

In this Opinion, we evaluated lines of evidence constructed as species-specific risk hypotheses to ensure relevant endpoints were addressed. Ultimately, our analysis weighs each line of evidence by evaluating the best commercial and scientific data available that pertain to a given risk hypothesis. Overall, the analysis is a qualitative approach that uses some quantitative tools to provide examples of potential risks to listed salmonids and their habitat. Multiple methods and tools currently exist for addressing contaminant-induced risk to the environment. Hazard-based assessments, probabilistic risk assessment techniques, combinations of the two, and deterministic approaches such as screening level assessments have been applied to questions of risk related to human health and the environment.

In recent pesticide risk assessments, probabilistic techniques have been used to evaluate the probability of exceeding a “toxic” threshold for aquatic organisms by combining pesticide monitoring data with species sensitivity distributions (Geisy, Solomon et al. 1999; Giddings 2009). There is utility in information generated by probabilistic approaches if supported by robust data. We compared the species sensitivity distributions presented in Giddings 2009 with the probability distributions of salmonid prey acute lethality values that we developed to highlight differences in outcomes. The assessment with carbaryl did not address many of the species-specific risk hypotheses. We found no other probabilistic assessments that addressed risk to salmonids affected by short-term sublethal exposures, mixtures, or affects on growth from reduced feeding ability and reduced abundances of prey.

NMFS considered the use of probabilistic risk assessment techniques for addressing risk at population and species (ESU and DPS) scales for the stressors of the action. However, we encountered significant limitations in available data that suggested the information was not sufficient to define exposure and/or response probabilities necessary to determine the probability of risk. In the *Risk Characterization* section, the distribution of the sensitivity of salmonid prey items was used to determine selection of a survival toxicity value used in population modeling exercises. Probabilistic techniques were not otherwise utilized in the Opinion due to issues with data collection, paucity of data, non-normal distributions of data, and quality assurance and quality control. For example, it was not deemed appropriate to pair the salmonid prey responses with exposure probabilities based on monitoring results given the limitations of that data set discussed in the *Effects of the Proposed Action*. To evaluate population consequences associated with potential lethality from pesticide exposure in salmon, NMFS selected the lowest reported salmonid LC50 from the available information to ensure risk was not underestimated. When we consider the data limitations coupled with the inherent complexity of EPA's proposed action (Figure 1) in California, Idaho, Oregon, and Washington, we find that probabilistic assessments at population and species scales introduce an unquantifiable amount of uncertainty that undermines confidence in derived risk estimates. These same studies do not factor the status of the existing health and baseline conditions of the environment into their assessment. At this time, the best available data do not support such an analysis and conclusions from such an analysis would be highly speculative.

Action Area

The action area is defined as all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR §402.02). Given EPA's nationwide authorization of these pesticides, the action area would encompass the entire U.S. and its territories. These same geographic areas would include all listed species and designated critical habitat under NMFS jurisdiction.

In this instance, as a result of the 2002 order in Washington Toxics Coalition v. EPA, EPA initiated consultation on its authorization of 37 pesticide a.i.s and their effects on listed Pacific salmonids under NMFS' jurisdiction and associated designated critical habitat in the states of California, Idaho, Oregon, and Washington. Consequently, for purposes of this Opinion, the action area consists of the entire range and most life history stages of listed salmon and steelhead and their designated critical habitat in California, Idaho, Oregon, and Washington. The action area encompasses all freshwater, estuarine, marsh, swamps, nearshore, and offshore marine surface waters of California, Oregon, and Washington. The action area also includes all freshwater surface waters in Idaho (Figure 5).

Carbaryl, carbofuran, and methomyl are the second set of three insecticides identified in the consultation schedule established in the settlement agreement and are analyzed in this Opinion. NMFS' analysis focuses only on the effects of EPA's action on listed Pacific salmonids in the above-mentioned states. It includes the effects of these pesticides on the recently listed Lower Columbia River coho salmon, Puget Sound steelhead, and Oregon Coast coho salmon. The Lower Columbia River coho salmon was listed as endangered in 2005. The Puget Sound steelhead and the Oregon Coast coho salmon were listed as threatened in 2007 and 2008, respectively.

EPA's consultation with NMFS remains incomplete until it analyzes the effects of its authorization of pesticide product labels with carbaryl, carbofuran, and methomyl for all remaining threatened and endangered species under NMFS' jurisdiction. EPA must ensure its action does not jeopardize the continued existence or result in the destruction or adverse modification of critical habitat for other listed species and designated critical habitat under NMFS' jurisdiction throughout the U.S. and its territories.

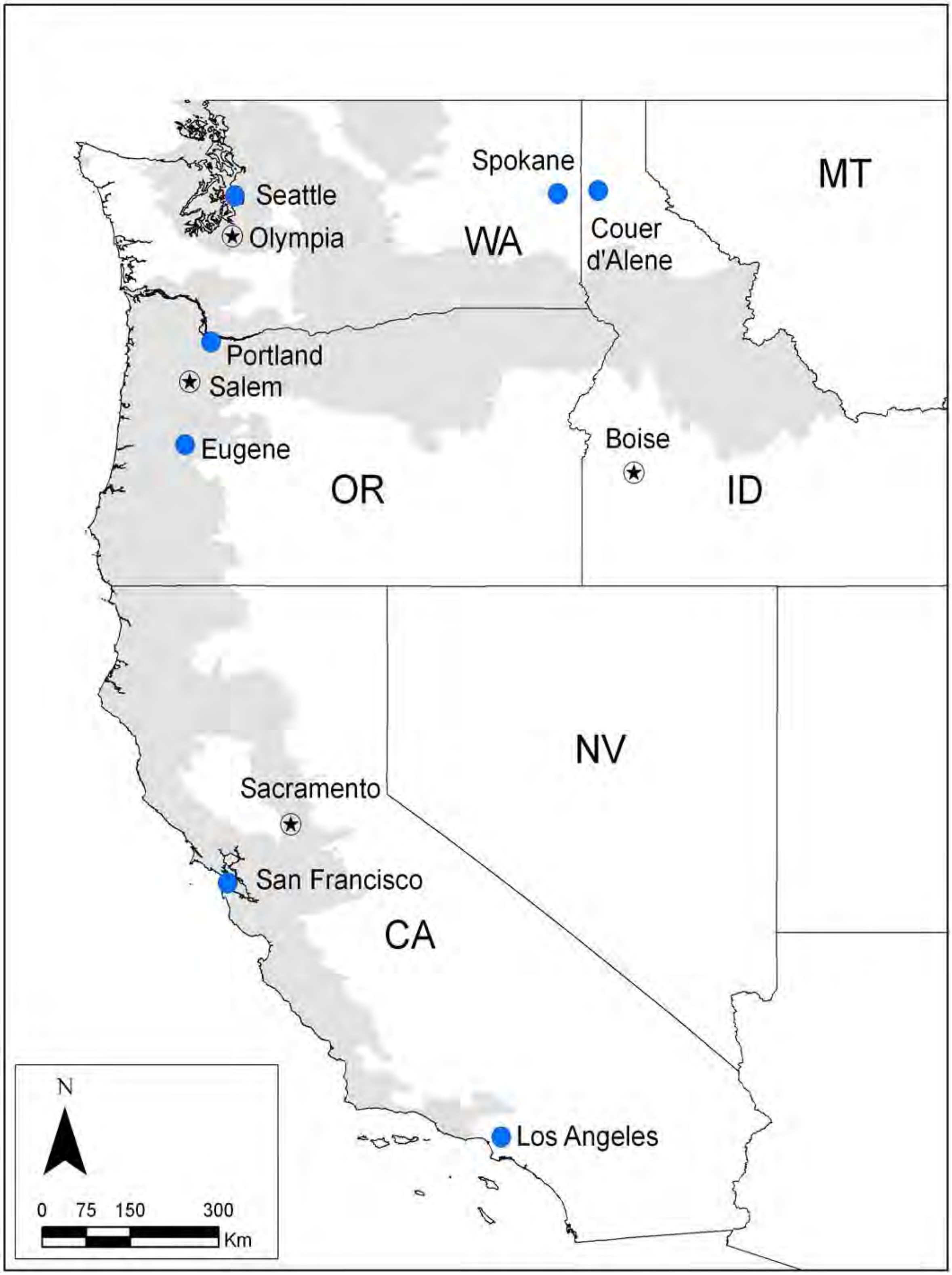


Figure 5. Map showing extent of inland action area with the range of all ESU and DPS boundaries for ESA listed salmonids highlighted in gray.

Status of Listed Resources

NMFS has determined that the following species and critical habitat designations may occur in this action area for EPA's registration of carbaryl, carbofuran, and methomyl - containing products (Table 5). More detailed information on the status of these species and critical habitat are found in a number of published documents including recent recovery plans, status reviews, stock assessment reports, and technical memorandums. Many are available on the Internet at http://www.nmfs.noaa.go/pr/species/.

Table 5. Listed Species and Critical Habitat (denoted by asterisk) in the Action Area

Common Name (Distinct Population Segment or Evolutionarily Significant Unit)	***Scientific Name***	***Status***
Chinook salmon (California Coastal*)	*Oncorhynchus tshawytscha*	Threatened
Chinook salmon (Central Valley Spring-run*)		Threatened
Chinook salmon (Lower Columbia River*)		Threatened
Chinook salmon (Upper Columbia River Spring-run*)		Endangered
Chinook salmon (Puget Sound*)		Threatened
Chinook salmon (Sacramento River Winter-run*)		Endangered
Chinook salmon (Snake River Fall-run*)		Threatened
Chinook salmon (Snake River Spring/Summer-run*)		Threatened
Chinook salmon (Upper Willamette River*)		Threatened
Chum salmon (Columbia River*)	*Oncorhynchus keta*	Threatened
Chum salmon (Hood Canal Summer-run*)		Threatened
Coho salmon (Central California Coast*)	*Oncorhynchus kisutch*	Endangered
Coho salmon (Lower Columbia River)		Threatened
Coho salmon (Southern Oregon & Northern California Coast*)		Threatened
Coho salmon (Oregon Coast*)		Threatened
Sockeye salmon (Ozette Lake*)	*Oncorhynchus nerka*	Threatened
Sockeye salmon (Snake River*)		Endangered
Steelhead (Central California Coast*)	*Oncorhynchus mykiss*	Threatened
Steelhead (California Central Valley*)		Threatened
Steelhead (Lower Columbia River*)		Threatened
Steelhead (Middle Columbia River*)		Threatened
Steelhead (Northern California*)		Threatened
Steelhead (Puget Sound)		Threatened
Steelhead (Snake River*)		Threatened
Steelhead (South-Central California Coast*)		Threatened
Steelhead (Southern California*)		Threatened
Steelhead (Upper Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened

The following brief narratives summarize the biology and ecology of threatened and endangered species in the action area that are relevant to the effects analysis in this

Opinion. Summaries of the status and trends [including (VSP) information] of each species are presented to provide a foundation for the analysis.

One of the important factors defining a viable population is the population's long- and short-term tendency to increase in abundance. In our status reviews of each listed salmonid species, we calculated the median annual population growth rate (denoted as lambda, λ) from available time series of abundance for individual populations. The lambda for each population is calculated using the rate at which four year running sums of available abundance estimates changes through time. Several publications provide a detailed description of the calculation of lambda (McClure, Holmes et al. 2003; Good, Waples et al. 2005). The lambda values for salmonid VSPs presented in these papers are summarized in Appendix 2. Unfortunately, reliable time series of abundance estimates are not available for most Pacific salmon and steelhead populations. In those cases, we made general inferences of long-term change based on what is known of historical and past abundances from snapshot surveys, surveys of a population segments, harvest by commercial and recreational fisheries, and professional judgment. We then compare these to similar information of current populations.

Below, each species narrative is followed by a description of its critical habitat with particular emphasis on any essential features of the habitat that may be exposed to the proposed action, and may warrant special attention.

Chinook Salmon

Description of the Species

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). In addition, Chinook salmon have been reported in the Canadian Beaufort Sea (McPhail and Lindsey 1970). We discuss the distribution, life history, diversity (when applicable), status, and critical habitat of the nine species of endangered and threatened Chinook salmon separately.

Of the Pacific salmon species, Chinook salmon exhibit one of the most diverse and complex life history strategies. Chinook salmon are generally described as one of two races, within which there is substantial variation. One form, the “stream-type” resides in freshwater for a year or more following emergence from gravel nests. Another form, the “ocean-type” migrates to the ocean within their first year. The ocean-type typifies populations north of 56^{o}N (Healey 1991). Within each race, there is often variation in age at seaward migration, age of maturity, timing of spawning migrations, male precocity, and female fecundity.

Status and Trends

Over the past few decades, the size and distribution of Chinook salmon populations have declined because of natural phenomena and human activity. Geographic features, such as waterfalls, pose natural barriers to salmon migrating to spawning habitat. Flooding can eliminate salmon runs and significantly alter large regions of salmon habitat. However, these threats are not considered as serious as several anthropogenic threats. Of the various natural phenomena that affect most populations of Pacific salmon, changes in ocean productivity are generally considered most important. Natural variations in freshwater and marine environments have substantial effects on the abundance of salmon populations.

Salmon along the U.S. west coast are prey for a variety of predators, including marine mammals, birds, sharks, and other fishes. In general, Chinook salmon are prey for pelagic fishes, birds, and marine mammals, including harbor seals, sea lions, and killer whales. Chinook salmon are also exposed to high rates of natural predation, during freshwater rearing and migration stages, as well as during ocean migration. There have been recent concerns that the increasing size of tern, seal, and sea lion populations in the Pacific Northwest may have reduced the survival of some salmon ESUs. Human activities include the operation of hydropower systems, over-harvest, hatcheries, and habitat degradation including poor water quality from chemical contamination.

Chinook salmon are dependent on the quantity and quality of aquatic habitats. Juvenile salmonids rely on a variety of non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle. Examples of off-channel habitat include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, and braids (Anderson 1999; Swift III 1979). Chinook salmon, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the hydrogeomorphology of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile Chinook salmon; and land use practices (logging, agriculture, urbanization) that destroy or alter wetland and riparian ecosystems. These activities and features introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Salmonids along the west coast of the U.S. share common threats. Therefore, anthropogenic threats for all species and stocks are summarized here (see (NMFS 2005) for a review). Population declines have resulted from several human-mediated causes. However, the greatest negative influence has been the establishment of waterway obstructions such as dams, power plants, and sluiceways for hydropower, agriculture, flood control, and water storage. These structures have blocked salmon migration to spawning habitat or resulted in direct mortality and have eliminated entire salmon runs. Presently, many of these structures have been re-engineered, renovated, or removed to allow for surviving runs to access former habitat. However, success has been limited. Remaining freshwater habitats are threatened from development along waterways as well as sedimentation, pollution run-off, habitat modification, and erosion. These factors can directly cause mortality, affect salmonid health, or modify spawning habitat so as to reduce reproductive success. Immature salmonids remain in freshwater systems and may be exposed to these modifications for years. These conditions reduce juvenile survival.

Salmonids are also a popular commercial resource and have faced significant pressure from fishing. Although currently protected, illegal oceanic driftnet gear is suspected of hindering salmon survival and recovery. Despite the protection of weaker salmonid stocks from fishing, exploitation of more populous stocks may actually harm weaker stocks. Hatchery-reared salmon have been and are still being introduced to bolster stocks. However, the broader effects of this action are unknown.

California Coastal Chinook Salmon

Distribution

California Coastal (CC) Chinook salmon includes all naturally-spawned coastal Chinook salmon spawning from Redwood Creek south through the Russian River as shown in (Figure 6).

CC Chinook salmon are a fall-run, ocean-type fish. Although a spring-run (river-type) component existed historically, it is now considered extinct (Bjorkstedt, Spence et al. 2005). Table 6 identifies populations within the CC Chinook salmon ESU, their abundances, and hatchery input.

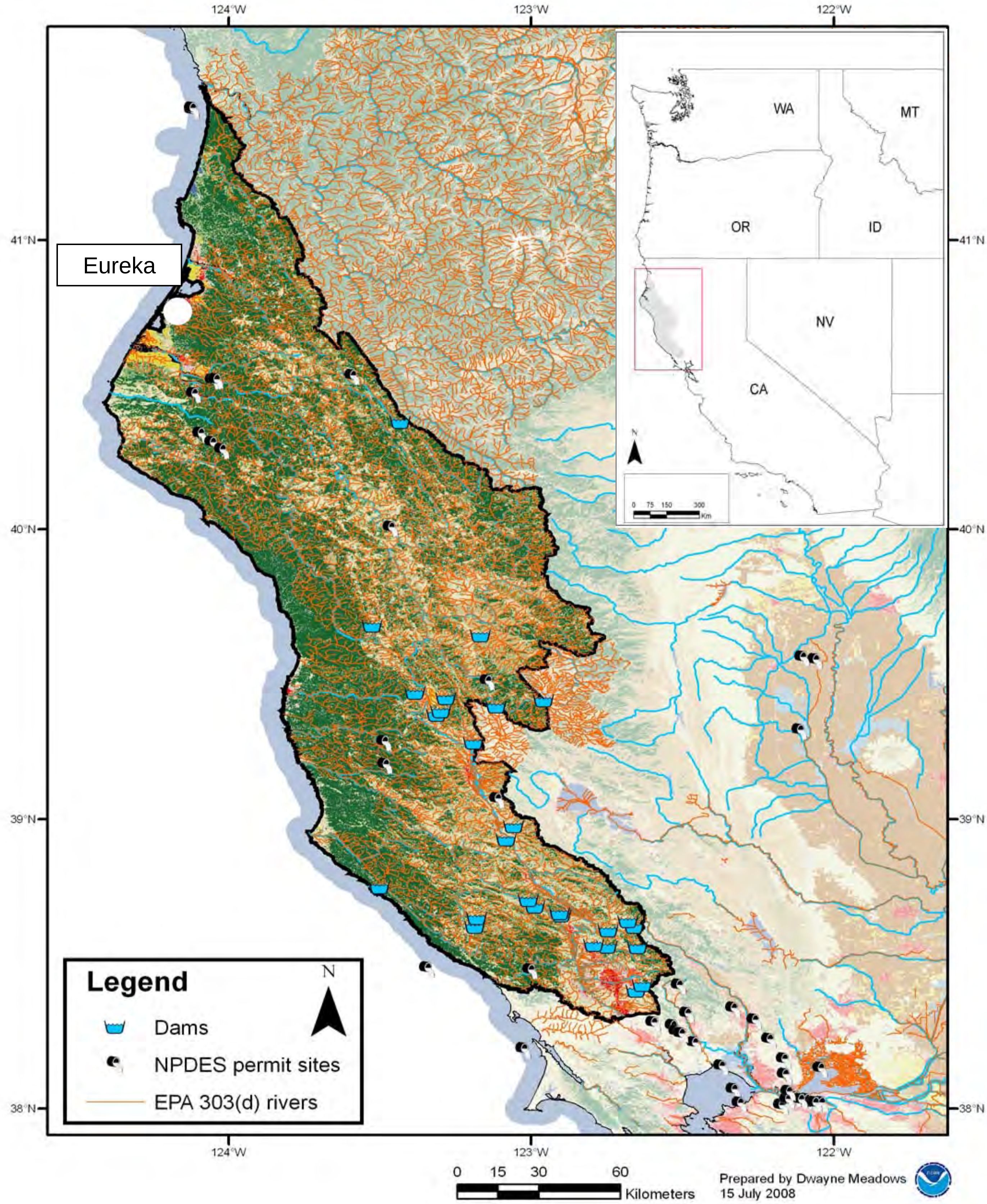


Figure 6. CC Chinook salmon distribution. Land Cover Class Legend in Figure 7.

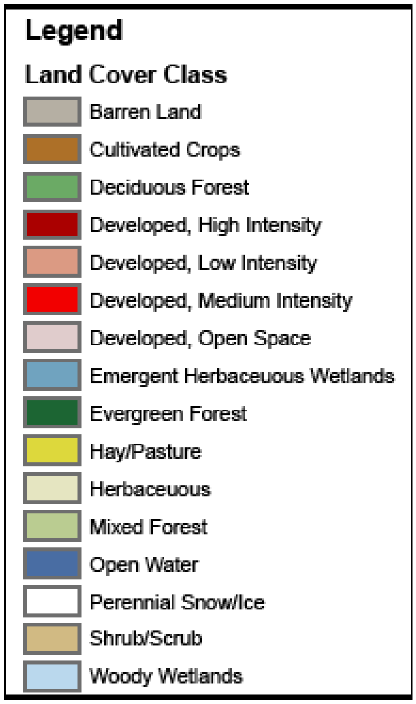


Figure 7. Legend for the Land Cover Class categories found in species distribution maps. Land cover is based on the 2001 National Land Cover Data and classifications. http://www.mrlc.gov/index.php.

Table 6. CC Chinook salmon--preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Eel River (includes * tributaries below)	17,000-55,000	156-2,730	~30%
Mainstem Eel River*	13,000	Inc. in Eel River	Unknown
Van Duzen River*	2,500	Inc. in Eel River	Unknown
Middle Fork Eel River*	13,000	Inc. in Eel River	Unknown
South Fork Eel River*	27,000	Inc. in Eel River	Unknown
North Fork Eel River*	Unknown	Inc. in Eel River	Unknown
Upper Eel River*	Unknown	Inc. in Eel River	Unknown
Redwood Creek	1,000-5,000	Unknown	0
Mad River	1,000-5,000	19-103	Unknown
Bear River	100	Unknown	0
Mattole River	1,000-5,000	Unknown	Unknown
Russian River	50-500	200,000	~0%
Humbolt Bay tributaries	40	120	40 (33%)
Tenmile to Gualala coastal effluents	Unknown	Unknown	0
Small Humboldt County rivers	1,500	Unknown	0
Rivers north of Mattole River	600	Unknown	0
Noyo River	50	Unknown	0
Total	**20,750-72,550**	**200,175 (min)**	

Status and Trends

CC Chinook salmon were listed as threatened on September 16, 1999 (64 FR 50393). Their classification was reaffirmed following a status review on June 28, 2005 (70 FR 37160). The outcome was based on the combined effect of dams that prevent individuals from reaching spawning habitat, logging, agricultural activities, urbanization, and water withdrawals in the river drainages that support CC Chinook salmon. Historical estimates of escapement, based on professional opinion and evaluation of habitat conditions, suggest abundance was roughly 73,000 in the early 1960s with the majority of fish spawning in the Eel River [see CDFG 1965 *in* (Good, Waples et al. 2005)]. The species exists as small populations with highly variable cohort sizes and discussion is underway to split Eel River salmon into as many as five separate populations (see Table 3). The Russian River probably contains some natural production. However, the origin of those fish is unclear as a number of introductions of hatchery fish occurred over the last century. The Eel River contains a substantial fraction of the remaining Chinook salmon spawning habitat for this species.

Since the original listing and status review, little new data are available or suitable for analyzing trends or estimating changes in the Eel River population's growth rate (Good, Waples et al. 2005). Historical and current abundance information indicates that independent populations of Chinook salmon are depressed in many of those basins where they have been monitored.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Critical habitat in this ESU consists of limited quantity and quality summer and winter rearing habitat, as well as marginal spawning habitat. Compared to historical conditions, there are fewer pools, limited cover, and reduced habitat complexity. The limited instream cover that does exist is provided mainly by large cobble and overhanging vegetation. Instream large woody debris, needed for foraging sites, cover, and velocity refuges is especially lacking in most of the streams throughout the basin. NMFS has determined that these degraded habitat conditions are, in part, the result of many human-induced factors affecting critical habitat. They include dam construction, agricultural and mining activities, urbanization, stream channelization, water diversion, and logging.

Central Valley Spring-Run Chinook Salmon

Distribution

The Central Valley spring-run (CV) Chinook salmon includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California (Figure 8).

Table 7 identifies populations within the CV spring-run Chinook salmon ESU, their abundances, and hatchery input**.**

Table 7. CV Chinook salmon--preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Butte Creek Spring-run Chinook		67-4,513	Unknown
Deer Creek Spring-run Chinook		243-1,076	Unknown
Mill Creek Spring-run Chinook		203-491	Unknown
Total	**~700,000 for all populations**	**513-6,080**	Unknown

Life History

CV Chinook salmon enter the Sacramento River from March to July and spawn from late August through early October, with a peak in September. Spring-run fish in the Sacramento River exhibit an ocean-type life history, emigrating as fry and sub-yearlings. Chinook salmon require cool freshwater while they mature over the summer. This species tends to take advantage of high flows. Adult upstream migration may be blocked by temperatures above 21ºC (McCullough 1999). Temperatures below 21ºC can stress fish by increasing their susceptibility to disease (Berman 1990) and elevating their metabolism (Brett 1979).

Status and Trends

CV Chinook salmon were listed as threatened on September 16, 1999 (64 FR 50393). This classification was retained following a status review on June 28, 2005 (70 FR 37160). The species was listed because dams isolated individuals from most of their historic spawning habitat and the remaining habitat is degraded. Historically, spring-run Chinook salmon were predominant throughout the CV. This species occupied the upper and middle reaches (1,000 to 6,000 ft) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud and Pit Rivers. Smaller populations occurred in most tributaries with sufficient habitat for over-summering adults (Stone 1874; Rutter 1904; Clarke 1929).

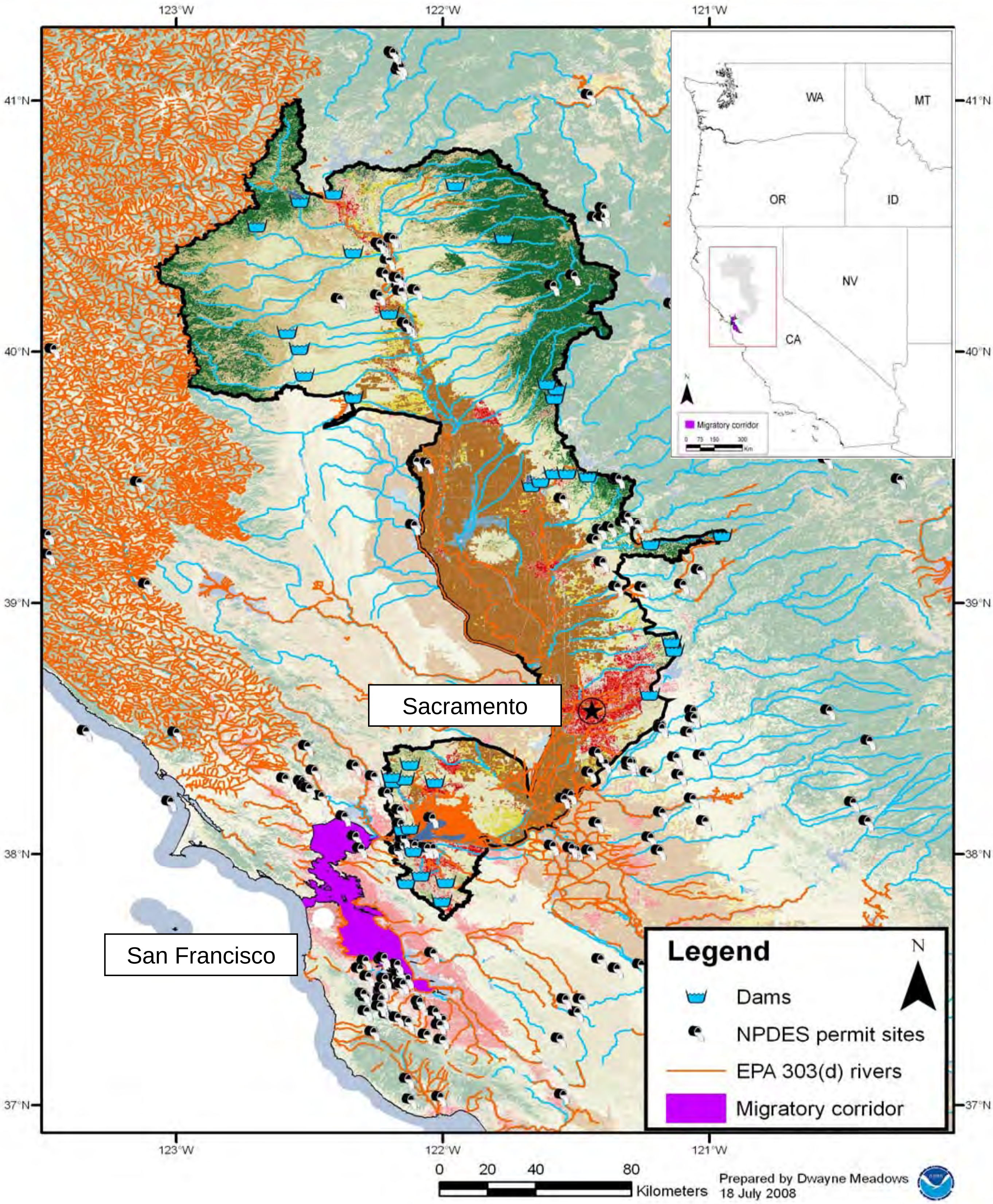


Figure 8. CV Chinook salmon distribution. Land Cover Class Legend in Figure 7.

The CV drainage as a whole is estimated to have supported spring-run Chinook salmon runs as large as 700,000 fish between the late 1880s and the 1940s (Brown, Moyle et al. 1994). Before construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone (Fry 1961). Following the completion of Friant Dam, the native population from the San Joaquin River and its tributaries (*i.e.*, the Stanislaus and Mokelumne Rivers) was extirpated. Spring-run Chinook salmon no longer exist in the American River due to the operation of Folsom Dam. Naturally spawning populations of CV Chinook salmon currently are restricted to accessible reaches of the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Feather River, Mill Creek, and Yuba River (CDFG 1998). Since 1969, the CV Chinook salmon ESU (excluding Feather River fish) has displayed broad fluctuations in abundance ranging from 25,890 in 1982 to 1,403 in 1993 (CDFG unpublished data).

The average abundance for the ESU was 12,499 for the period of 1969 to 1979, 12,981 for the period of 1980 to 1990, and 6,542 for the period of 1991 to 2001. In 2003 and 2004, total run size for the ESU was 8,775 and 9,872 adults, respectively. These averages are well above the 1991 to 2001 average.

Evaluating the ESU as a whole, however, masks significant changes that are occurring among populations that comprise the ESU. For example, the mainstem Sacramento River population has undergone a significant decline while the abundance of many tributary populations increased. Average abundance of Sacramento River mainstem spring-run Chinook salmon recently declined from a high of 12,107 for the period 1980 to 1990, to a low of 609 for the period 1991 to 2001 (Good, Waples et al. 2005). Meanwhile, the average abundance of Sacramento River tributary populations increased from a low of 1,227 to a high of 5,925 over the same periods.

According to Good et al. (2005), abundance time series data for Mill, Deer, and Butte creeks spring-run Chinook salmon (updated through 2001) confirm that population increases in the tributary populations seen in the 1990s have continued. During this period, habitat improvements included the removal of several small dams and increases in summer flows in the watersheds, a reduced ocean fisheries, and a favorable terrestrial and marine climate. All three spring-run Chinook populations in the Sacramento River tributaries have long-and short-term lambdas >1, indicating population growth. However, population sizes are relatively small compared to fall-run Chinook salmon populations, and there have been some extreme fluctuations in population size, which is often indicative of an impending collapse in small populations. Additionally, Feather River hatchery and Feather River spring-run Chinook salmon are not closely related to the Mill, Deer, and Butte creek spring-run Chinook salmon populations. This group represents a distinct genetic legacy. Although protective measures and critical habitat restoration likely have contributed to recent increases in spring-run Chinook salmon abundance, the ESU is still below levels observed from the 1960s through 1990. Threats from hatchery production (*i.e.*, competition for food between naturally spawned and hatchery fish, and run hybridization and homogenization), climatic variation, reduced stream flow, high water temperatures, predation, and large scale unscreened water diversions persist.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Factors contributing to the downward trends in this ESU include: loss of most historical spawning habitat, reduced access to spawning/rearing habitat behind impassable dams, climatic variation, water management activities, hybridization with fall-run Chinook salmon, predation, and harvest. Additional factors include the degradation and modification of remaining rearing and migration habitats in the natal stream, the Sacramento River, and the Sacramento delta. The natal tributaries have many small hydropower dams and water diversions that in some years have greatly reduced or eliminated in-stream flows during spring-run migration periods. Problems in the migration corridor include unscreened or inadequately screened water diversions, predation by nonnative species, and excessively high water temperatures. Collectively, these factors have impacted spring-run Chinook salmon critical habitat and population numbers (CDFG 1998). Several actions have been taken to improve and increase the PCEs of critical habitat for spring-run Chinook salmon, including improved management of CV water (*e.g.*, through use of CALFED EWA and CV Project Improvement Act (b)(2) water accounts), implementing new and improved screen and ladder designs at major water diversions along the mainstem Sacramento River and tributaries, removal of several small dams on important spring-run Chinook salmon spawning streams, and changes in ocean and inland fishing regulations to minimize harvest.

Lower Columbia River Chinook Salmon

Distribution

Lower Columbia River (LCR) Chinook salmon includes all naturally-spawned populations of Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Oregon and Washington, east of the Hood River and the White Salmon River (Figure 7). Naturally spawned populations also occur along the Willamette River to Willamette Falls, Oregon, exclusive of spring-run Chinook salmon in the Clackamas River (Table 5). The Cowlitz, Kalama, Lewis, White Salmon, and Klickitat Rivers are the major river systems on the Washington side, and the lower Willamette and Sandy Rivers are foremost on the Oregon side. The eastern boundary for this species occurs at Celilo Falls, which corresponds to the edge of the drier Columbia Basin Ecosystem. Historically, Celilo Falls may have been a barrier to salmon migration at certain times of the year. Table 8 identifies populations within the LCR Chinook salmon ESU, their abundances, and hatchery input

Table 8. LCR Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay	Unknown	Unknown	Unknown
Grays River	2,477	99	38%
Big Creek	Unknown	Unknown	Unknown
Elochoman River	Unknown	676	68%
Clatskanie River	Unknown	Unknown	Unknown
Mill, Abernathy, and German Creeks	Unknown	734	47%
Scappoose Creek	Unknown	Unknown	Unknown
Coweeman River	Unknown	274	0%
Lower Cowlitz River	4,971	1,562	62%
Upper Cowlitz River (fall run)	Unknown	5,682	Unknown
Toutle River (fall run)	53,956	Unknown	Unknown
Kalama River (fall run)	25,392	2,931	67%
Salmon Creek and Lewis River	47,591	256	0%
Clackamas River	Unknown	40	Unknown
Washougal River	7,518	3,254	58%
Sandy River (fall run)	Unknown	183	Unknown
Columbia Gorge-lower tributaries	Unknown	Unknown	Unknown
Columbia Gorge-upper tributaries	Unknown	Unknown	Unknown
Hood River (fall run)	Unknown	18	Unknown
Big White Salmon River	Unknown	334	21%
Sandy River (late fall run)	Unknown	504	3%
Lewis River-North Fork	Unknown	7,841	13%
Upper Cowlitz River (spring run)	Unknown	Unknown	Unknown
Cispus River	Unknown	1,787	Unknown
Tilton River	Unknown	Unknown	Unknown
Toutle River (spring run)	2,901	Unknown	Unknown
Kalama River (spring run)	4,178	98	Unknown
Lewis River	Unknown	347	Unknown
Sandy River (spring run)	Unknown	Unknown	Unknown
Big White Salmon River (spring run)	Unknown	Unknown	Unknown
Hood River (spring run)	Unknown	51	Unknown
Total	**148,984 (min)**	**26,273 (min)**	

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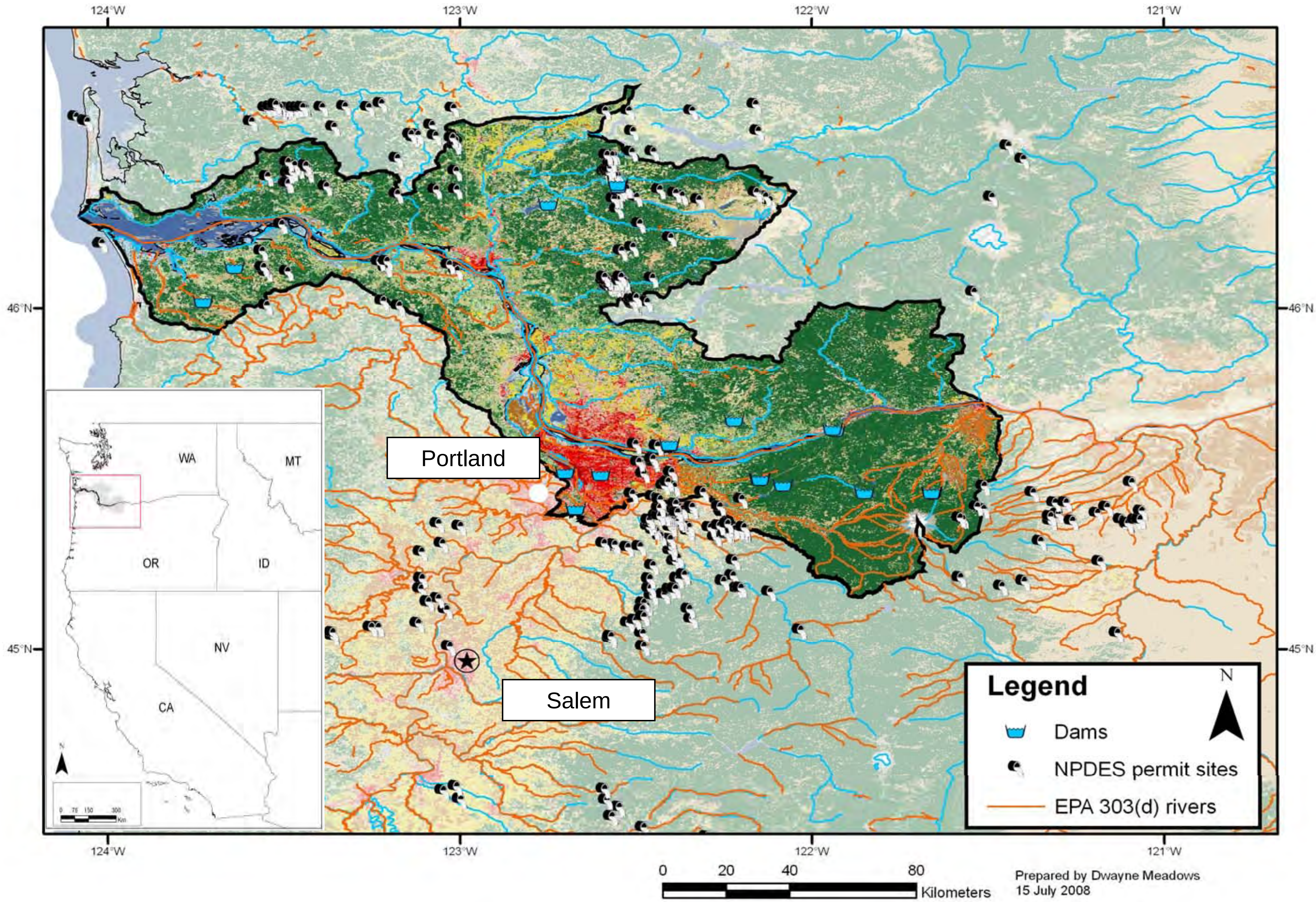

Figure 9. LCR Chinook salmon distribution. Land Cover Class Legend in Figure 7.

Life History

LCR Chinook salmon display three life history types including early fall runs, late fall runs, and spring-runs. Spring and fall runs have been designated as part of a LCR Chinook salmon ESU. The predominant life history type for this species is the fall-run. Fall Chinook salmon enter freshwater typically in August through October to spawn in large river mainstems. The juvenile life history stage emigrates from freshwater as sub-yearling (ocean-type). Spring Chinook salmon enter freshwater in March through June to spawn in upstream tributaries and generally emigrate from freshwater as yearlings (stream-type).

Status and Trends

LCR Chinook salmon were originally listed as threatened on March 24, 1999 (64 FR 14308). This status was reaffirmed on June 28, 2005 (70 FR 37160). Historical records of Chinook salmon abundance are sparse. However, cannery records suggest a peak run of 4.6 million fish [43 million lbs see (Lichatowich 1999)] in 1883. Although fall-run Chinook salmon occur throughout much of their historical range, they remain vulnerable to large-scale hatchery production, relatively high harvest, and extensive habitat degradation. The Lewis River late fall Chinook salmon population is the healthiest and has a reasonable probability of being self-sustaining. Abundances largely declined during 1998 to 2000. Trend indicators for most populations are negative, especially if hatchery fish are assumed to have a reproductive success equivalent to that of natural-origin fish.

New data acquired for the Good et al. (2005) report includes spawner abundance estimates through 2001, new estimates of the fraction of hatchery spawners, and harvest estimates. In addition, estimates of historical abundance have been provided by the Washington Department of Fish and Wildlife (WDFW). The Willamette/Lower Columbia River Technical Review Team (W/LCRTRT) has estimated that 8-10 historic populations have been extirpated, most of them spring-run populations. Almost all of the spring-run Chinook of LCR Chinook are at very high risk of extinction. Near loss of that important life history type remains an important concern. Although some natural production currently occurs in 20 or so populations, only one exceeds 1,000 spawners. Most LCR Chinook salmon populations have not seen increases in recent years as pronounced as those that have occurred in many other geographic areas.

According to Good et al. (2005), the majority of populations for which data are available have a long-term trend of <1; indicating the population is in decline. Currently, the spatial structures of populations in the Coastal and Cascade Fall Run major population groups (MPGs) are similar to their respective historical conditions. The genetic diversity of the Coastal, Cascade, and Gorge Fall Run MPGs (*i.e.*, all except the Late Fall Run Chinook salmon MPG) has been eroded by large hatchery influences and periodically by low effective population sizes. Hatchery programs for spring Chinook salmon are preserving the genetic legacy from populations that were extirpated from blocked areas. High hatchery production also poses genetic and ecological risks to natural populations and masks their performance.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Hood Rivers as well as specific stream reaches in a number of tributary subbasins. The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 52 subbasins reviewed in NMFS' assessment of critical habitat for the LCR Chinook salmon ESU, 13 subbasins were rated as having a medium conservation value, four were rated as low, and the remaining subbasins (35), were rated as having a high conservation value to LCR Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes resulting from hydropower development, loss of tidal marsh and swamp habitat, and degraded freshwater and marine habitat from industrial harbor and port development, and urban development. Limiting factors identified for this species include: (1) Habitat degradation and loss due to extensive hydropower development projects, urbanization, logging, and agriculture on Chinook spawning and rearing habitat in the LCR, (2) reduced access to spawning/rearing habitat in tributaries, (3) hatchery impacts, (4) loss of habitat diversity and channel stability in tributaries, (5) excessive fine sediment in spawning gravels, (6) elevated water temperature in tributaries, (7) harvest impacts, and (8) poor water quality.

Upper Columbia River Spring-run Chinook Salmon

Distribution

Endangered Upper Columbia River (UCR) spring-run Chinook salmon includes stream-type Chinook salmon that inhabit tributaries upstream from the Yakima River to Chief Joseph Dam (Figure 10). The UCR spring-run Chinook salmon is composed of three major population groups (MPGs): the Wenatchee River population, the Entiat River population, and the Methow River population. These same populations currently spawn in only three river basins above Rock Island Dam: the Wenatchee, Entiat, and Methow Rivers. Several hatchery populations are also listed including those from the Chiwawa, Methow, Twisp, Chewuch, and White rivers, and Nason Creek (Table 9). Table 9 identifies populations within the UCR Chinook salmon ESU, their abundances, and hatchery input.

Table 9. UCR Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Methow River	~2,100	79-9,904	59%
Twisp River	Unknown	10-369	54%
Chewuch River	Unknown	6-1,105	41%
Lost/Early River	Unknown	3-164	54%
Entiat River	~380	53-444	42%
Wenatchee River	~2,400	119-4,446	42%
Chiwawa River	Unknown	34-1,046	47%
Nason Creek	Unknown	8-374	39%
Upper Wenatchee River	Unknown	0-215	66%
White River	Unknown	1-104	8%
Little Wenatchee River	Unknown	3-74	21%
Total	**~4,880 (min)**		

Life History

UCR spring Chinook salmon begin returning from the ocean in the early spring. They enter the upper Columbia tributaries from April through July, with the run into the Columbia River peaking in mid-May. After migration, UCR spring Chinook salmon hold in freshwater tributaries until spawning occurs in the late summer, peaking in mid- to late August. Juvenile spring Chinook salmon spend a year in freshwater before emigrating to salt water in the spring of their second year.

Status and Trends

UCR spring-run Chinook salmon were listed as endangered on March 24, 1999 (64 FR 14308). This listing was reaffirmed on June 28, 2005 (70 FR 37160) based on a reduction of UCR spring-run Chinook salmon to small populations in three watersheds. Based on redd count data series, spawning escapements for the Wenatchee, Entiat, and Methow rivers have declined an average of 5.6%, 4.8%, and 6.3% per year, respectively, since 1958.

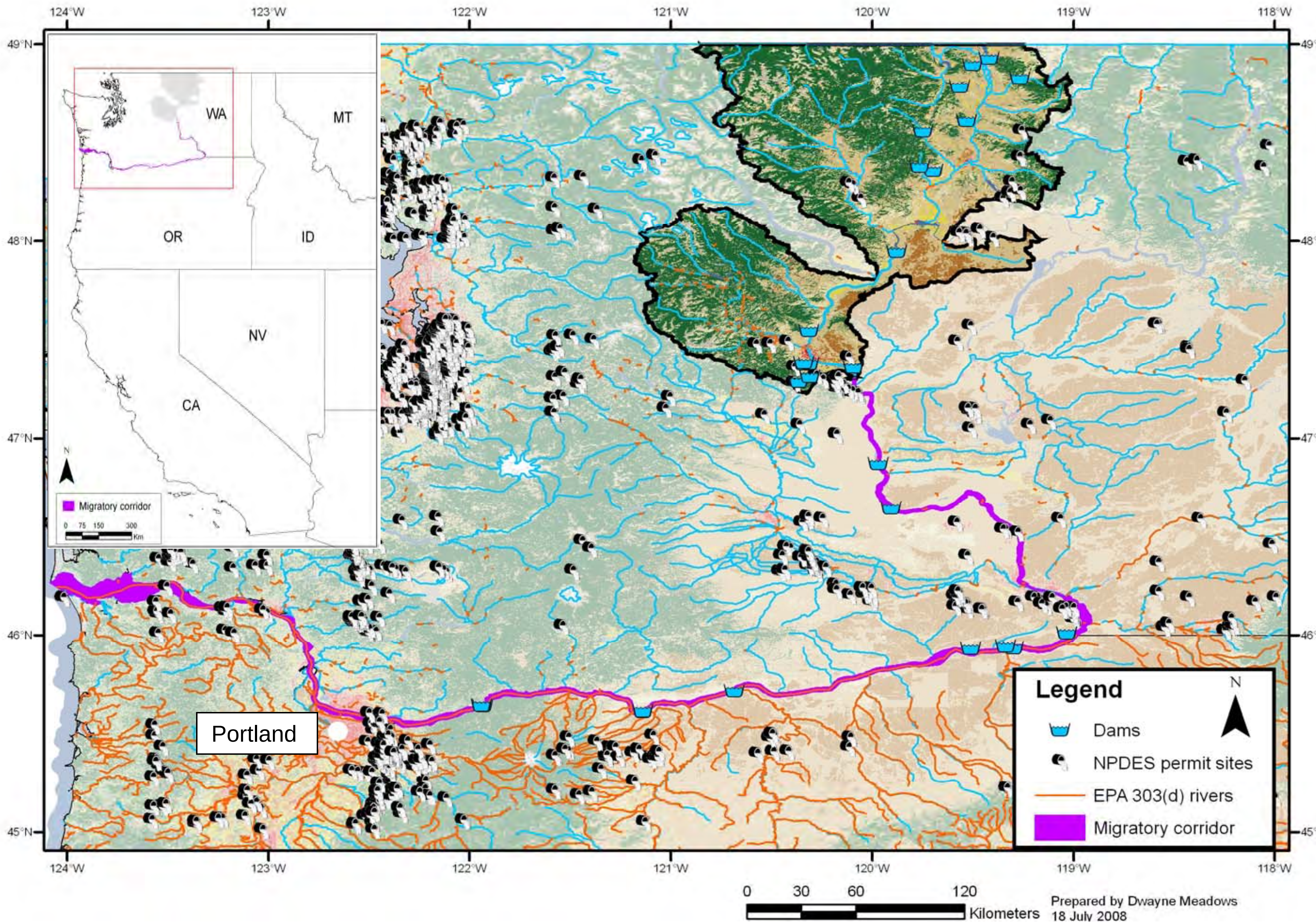

Figure 10. UCR Spring-run Chinook salmon distribution. Land Cover Class Legend in Figure 7.

In the most recent five-year geometric mean (1997 to 2001), spawning escapements were 273 for the Wenatchee population, 65 for the Entiat population, and 282 for the Methow population. These numbers represent only 8% to 15% of the minimum abundance thresholds. However, escapement increased substantially in 2000 and 2001 in all three river systems. Based on 1980-2004 returns, the average annual population growth rate, lambda, for this ESU is estimated at 0.93 (meaning the population is not replacing itself) (Fisher and Hinrichsen 2006). Assuming that population growth rates were to continue at 1980-2004 levels, UCR spring-run Chinook salmon populations are projected to have very high probabilities of decline within 50 years. Population viability analyses for this species suggest that these Chinook salmon face a significant risk of extinction: a 75 to 100% probability of extinction within 100 years (given return rates for 1980 to present). Finally, the Interior Columbia Basin Technical Recovery Team (ICBTRT) characterizes the diversity risk to all UCR spring Chinook populations as "high". The high risk is a result of reduced genetic diversity from homogenization of populations that occurred under the Grand Coulee Fish Maintenance Project in 1939-1943. Straying hatchery fish, and a low proportion of natural-origin fish in some broodstocks and a high proportion of hatchery fish on the spawning grounds have also contributed to the high genetic diversity risk.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to Chief Joseph Dam and several tributary subbasins. The critical habitat designation for this ESU also identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The UCR spring-run Chinook salmon ESU has 31 watersheds within its range. Five watersheds received a medium rating and 26 received a high rating of conservation value to the ESU. The Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Factors contributing to the downward trends in this ESU include: (1) Mainstem Columbia River hydropower system mortality, (2) tributary riparian degradation and loss of in-river wood, (3) altered tributary floodplain and channel morphology, (4) reduced tributary stream flow and impaired passage, (5) harvest impacts, and (6) degraded water quality.

Puget Sound Chinook Salmon

Distribution

The boundaries of the Puget Sound ESU correspond generally with the boundaries of the Puget Lowland Ecoregion (Figure 11). The Puget Lowland Ecoregion begins in Washington at approximately the Dungeness River near the eastern end of the Strait of Juan de Fuca and extends through Puget Sound to the British Columbia border and up to the Cascade foothills. The Puget Sound ESU includes all runs of Chinook salmon in the Puget Sound region from the North Fork Nooksack River to the Elwha River on the Olympic Peninsula. This ESU is comprised of 31 historical populations. Of these, 22 populations are believed to be extant. Thirty-six hatchery populations were included as part of the ESU and five were considered essential for recovery and listed. They include spring Chinook salmon from Kendall Creek, the North Fork Stillaguamish River, White River, and Dungeness River, and fall run fish from the Elwha River (Table 10). Table 10 identifies populations within the Puget Sound Chinook salmon ESU, their abundances, and hatchery input.

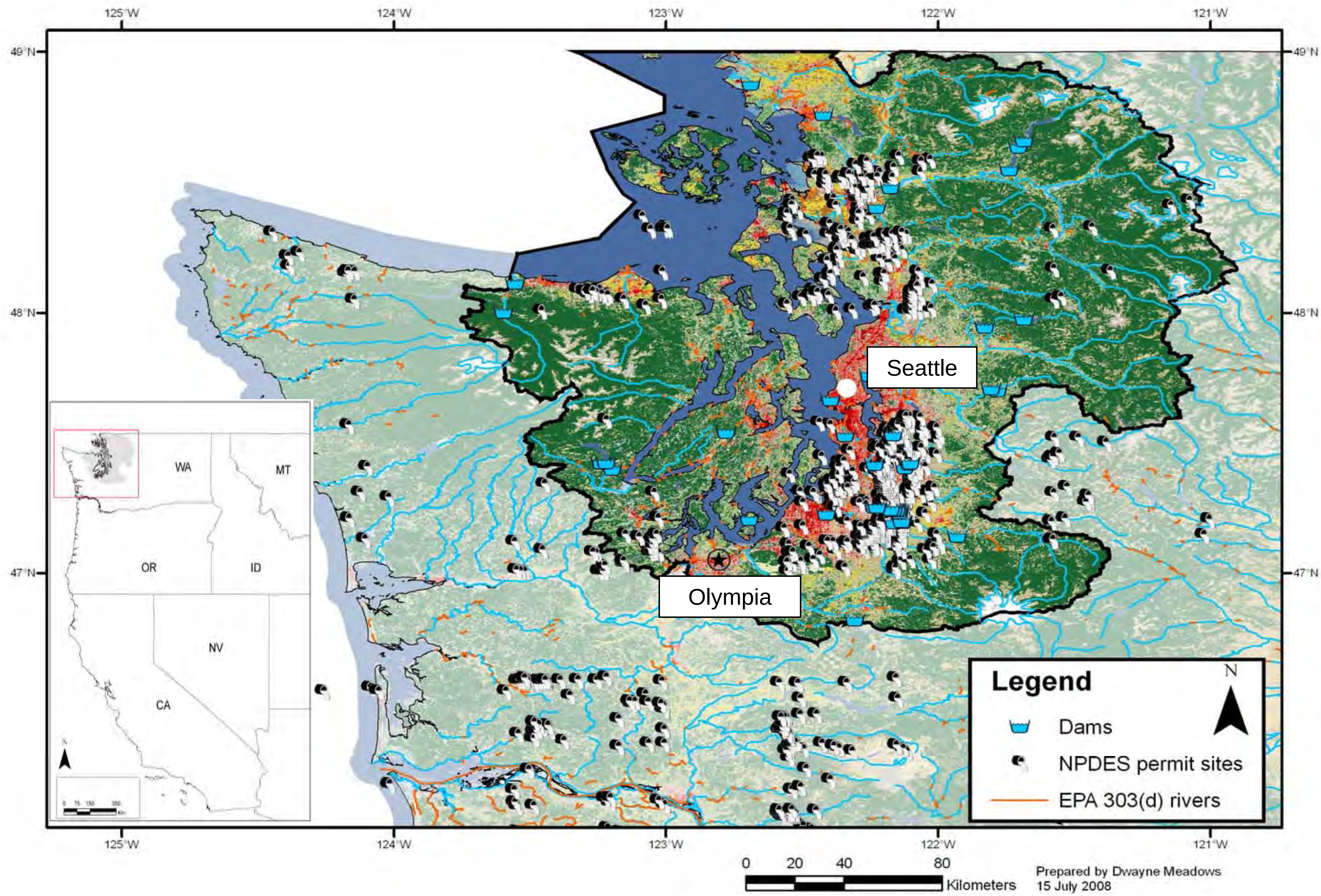

Figure 11. Puget Sound Chinook distribution. Land Cover Class Legend in Figure 7..

Table 10. Puget Sound Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Nooksack-North Fork	26,000	1,538	91%
Nooksack-South Fork	13,000	338	40%
Lower Skagit	22,000	2,527	0.2%
Upper Skagit	35,000	9,489	2%
Upper Cascade	1,700	274	0.3%
Lower Sauk	7,800	601	0%
Upper Sauk	4,200	324	0%
Suiattle	830	365	0%
Stillaguamish-North Fork	24,000	1,154	40%
Stillaguamish-South Fork	20,000	270	Unknown
Skykomish	51,000	4,262	40%
Snoqualmie	33,000	2,067	16%
North Lake Washington	Unknown	331	Unknown
Cedar	Unknown	327	Unknown
Green	Unknown	8,884	83%
White	Unknown	844	Unknown
Puyallup	33,000	1,653	Unknown
Nisqually	18,000	1,195	Unknown
Skokomish	Unknown	1,392	Unknown
Dosewallips	4,700	48	Unknown
Duckabush	Unknown	43	Unknown
Hamma Hamma	Unknown	196	Unknown
Mid Hood Canal	Unknown	311	Unknown
Dungeness	8,100	222	Unknown
Elwha	Unknown	688	Unknown
Total	**~690,000**	**39,343**	

Life History

Chinook salmon in this area generally have an "ocean-type" life history. Puget Sound populations exhibit both the early-returning and late-returning Chinook salmon spawners described by Healey (1997). However, within these two generalized behavioral forms, substantial variation occurs in juvenile behavior and residence time in fresh water and estuarine environments. Hayman et al. (1996) described three juvenile life histories for Chinook salmon with varying freshwater and estuarine residency times in the Skagit River system in northern Puget Sound. Chinook salmon use the nearshore area of Puget Sound during all seasons of the year and can be found long distances from their natal river systems (Brennan, Higgins et al. 2004).

Status and Trends

Puget Sound Chinook salmon were listed as threatened in 1999 (64 FR 14308). This status was re-affirmed on June 28, 2005 (70 FR 37160). This ESU has lost 15 spawning aggregations that were either demographically independent historical populations or major components of the life history diversity of the remaining 22 existing independent historical populations identified (Good, Waples et al. 2005). Nine of the 15 extinct spawning aggregations were early-run type Chinook salmon (Good, Waples et al. 2005). The disproportionate loss of early-run life history diversity represents a significant loss of the evolutionary legacy of the historical ESU.

The estimated total run size of Chinook salmon in Puget Sound in the early 1990s was 240,000 fish, representing a loss of nearly 450,000 fish from historic numbers. During a recent five-year period, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over 9,489 fish. Most populations had natural spawners numbering in the hundreds (median recent natural escapement is 766). Of the six populations with greater than 1,000 natural spawners, only two have a low fraction of hatchery fish. Estimates of the historical equilibrium abundance, based on pre-European settlement habitat conditions, range from 1,700 to 51,000 potential Puget Sound Chinook salmon spawners per population. The historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good, Waples et al. 2005).

Long-term trends in abundance and median population growth rates for naturally spawning populations of Puget Sound Chinook salmon indicate that approximately half of the populations are declining and the other half are increasing in abundance over the length of available time series. Eight of 22 populations are declining over the short-term, compared to 11 or 12 populations that have long-term declines (Good, Waples et al. 2005). Widespread declines and extirpations of spring- and summer-run Puget Sound Chinook salmon populations represent a significant reduction in the life history diversity of this ESU (Meyers, Kope et al. 1998). The median overall populations of long-term trend in abundance is 1, indicating that most populations are just replacing themselves.

Populations with the greatest long-term population growth rate are the North Fork Nooksack and White rivers.

Regarding spatial structure, the populations (22) presumed to be extinct are mostly early returning fish. Most of these are in the mid- to southern Puget Sound or Hood Canal and the Strait of Juan de Fuca. The ESU populations with the greatest estimated fractions of hatchery fish tend to be in the mid-to southern Puget Sound, Hood Canal, and the Strait of Juan de Fuca. Finally, all but one of the nine extinct Chinook salmon stocks is an early run population (or component of a population).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 49 subbasins (5th field Hydrological Units) reviewed in NMFS' assessment of critical habitat for the Puget Sound ESUs, nine subbasins were rated as having a medium conservation value, 12 were rated as low, and the remaining subbasins (40), where the bulk of Federal lands occur in this ESU, were rated as having a high conservation value to Puget Sound Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes (such as diking, revetments, loss of secondary channels in floodplains, widespread blockages of streams, and changes in peak flows), degraded freshwater and marine habitat affected by agricultural activities and urbanization, and upper river tributaries widely affected by poor forest practices, and lower tributaries. Hydroelectric development and flood control also impact Puget Sound Chinook salmon in several basins. Changes in habitat quantity, availability, diversity, flow, temperature, sediment load, water quality, and channel stability are common limiting factors in areas of critical habitat.

Sacramento River Winter-Run Chinook Salmon

Distribution

Sacramento River winter-run Chinook salmon consists of a single spawning population that enters the Sacramento River and its tributaries in California from November to June and spawns from late April to mid-August, with a peak from May to June (Figure 12). Sacramento River winter Chinook salmon historically occupied cold, headwater streams, such as the upper reaches of the Little Sacramento, McCloud, and lower Pit Rivers.

Life History

Winter-run fish spawn mainly in May and June in the upper mainstem of the Sacramento River. Winter-run fish have characteristics of both stream- and ocean-type races. They enter the river and migrate far upstream. Spawning is delayed for some time after river entry. Young winter-run Chinook salmon, however migrate to sea in November and December, after only four to seven months of river life (Burgner 1991).

Status and Trends

Sacramento River winter-run Chinook salmon were listed as endangered on January 4, 1994 (59 FR 440), and were reaffirmed as endangered on June 28, 2005 (70 FR 37160). This was based on restricted access from dams to a small fraction of salmon historic spawning habitat and the degraded conditions of remaining habitat. Sacramento River winter-run Chinook salmon consist of a single self-sustaining population which is entirely dependent upon the provision of suitably cool water from Shasta Reservoir during periods of spawning, incubation, and rearing.

Construction of Shasta Dams in the 1940s eliminated access to historic spawning habitat for winter-run Chinook salmon in the basin. Winter-run Chinook salmon were not expected to survive this habitat alteration (Moffett 1949). However, cold water releases from Shasta Dam have created conditions suitable for winter Chinook salmon for roughly

60 miles downstream from the dam. As a result the ESU has been reduced to a single spawning population confined to the mainstem Sacramento River below Keswick Dam. Some adult winter-run Chinook salmon were recently observed in Battle Creek, a tributary to the upper Sacramento River.

Quantitative estimates of run-size are not available for the period before 1996, the completion of Red Bluff Diversion Dam. However, winter-runs may have been as large as 200,000 fish based upon commercial fishery records from the 1870s (Brown, Moyle et al. 1994).

The CDFG estimated spawning escapement of Sacramento River winter-run Chinook salmon at 61,300 (60,000 mainstem, 1,000 Battle Creek, and 300 in Mill Creek) in the early 1960s. During the first three years of operation of the county facility at the Red Bluff Diversion Dam (1967 to 1969), the spawning run of winter-run Chinook salmon averaged 86,500 fish. From 1967 through the mid-1990s, the population declined at an average rate of 18% per year, or roughly 50% per generation. The population reached critically low levels during the drought of 1987 to 1992. The three-year average run size for the period of 1989 to 1991 was 388 fish.

Based on the Red Bluff Diversion Dam counts, the population has been growing rapidly since the 1990s. Mean run size from 1995-2000 has been 2,191, but have ranged from 364 to 65,683 (Good, Waples et al. 2005). Most recent estimates indicate that the short-term trend is 0.26, and the population growth rate is less than one.

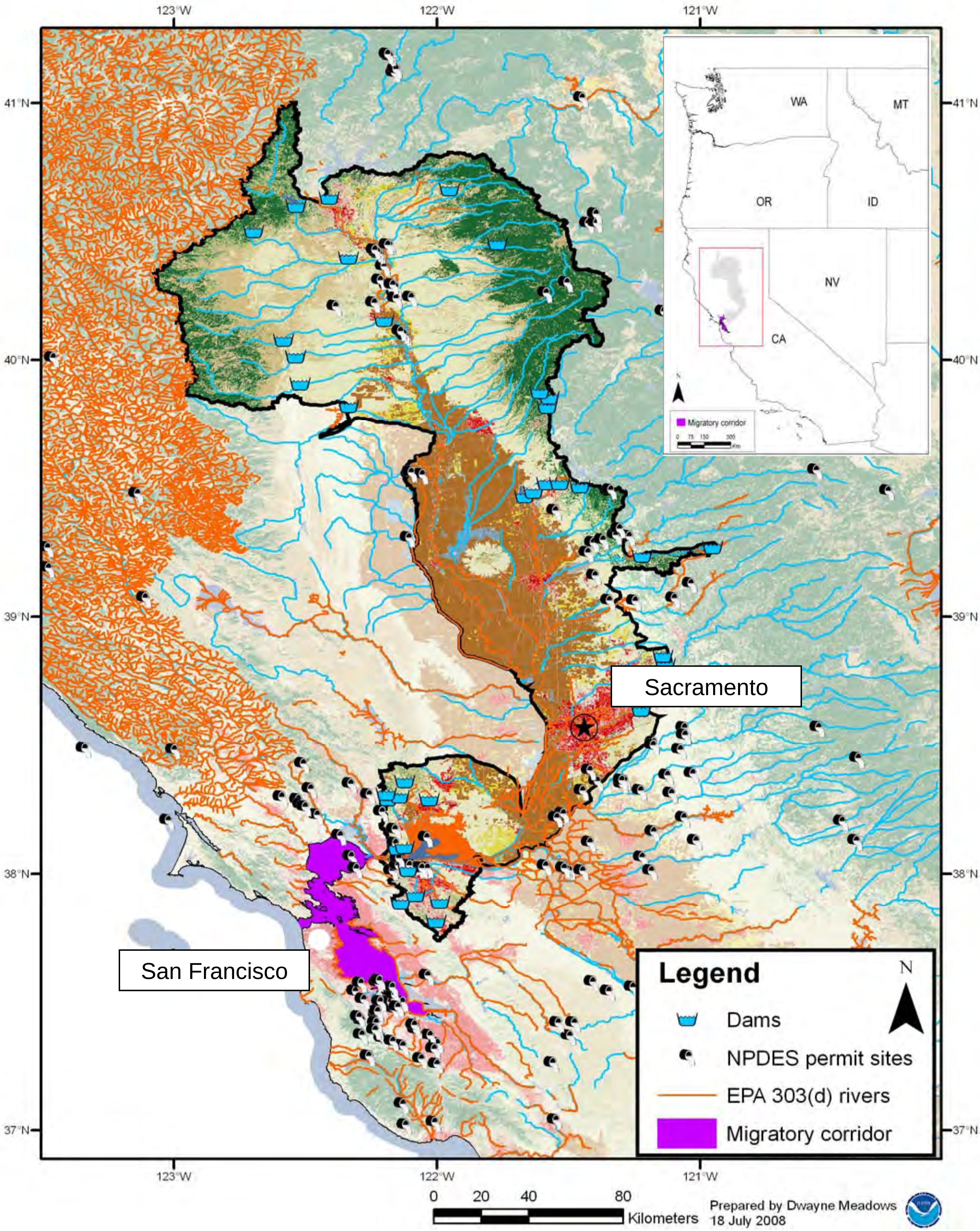

Figure 12. Sacramento River Winter-run Chinook salmon distribution. Land Cover Class Legend in Figure 7.

Critical Habitat

Critical habitat was designated for this species on June 16, 1993 (58 FR 33212). The following areas consist of the water, waterway bottom, and adjacent riparian zones: the Sacramento River from Keswick Dam, Shasta County (river mile 302) to Chipps Island (river mile 0) at the westward margin of the Sacramento-San Joaquin Delta, and other specified estuarine waters. Factors contributing to the downward trends in this ESU include: (1) Reduced access to spawning/rearing habitat, (2) possible loss of genetic integrity through population bottlenecks, (3) inadequately screened diversions, (4) predation at artificial structures and by nonnative species, (5) pollution from Iron Mountain Mine and other sources, (6) adverse flow conditions, (7) high summer water temperatures, (8) degraded water quality, (9) unsustainable harvest rates, (10) passage problems at various structures, and (11) vulnerability to drought (Good, Waples et al. 2005).

Snake River Fall-Run Chinook Salmon

Distribution

Historically, the primary fall-run Chinook salmon spawning areas occurred on the upper mainstem Snake River (SR) (Connor, Sneva et al. 2005). A series of SR mainstem dams blocks access to the upper SR, which significantly reduced spawning and rearing habitat for SR fall-run Chinook salmon (Figure 13).

The present range of spawning and rearing habitat for naturally-spawned SR fall-run Chinook salmon is limited to the SR below Hells Canyon Dam and the lower reaches of the Clearwater River. SR fall-run Chinook salmon spawn above Lower Granite Dam in the mainstem SR and in the lower reaches of the larger tributaries.

As a consequence of lost access to historic spawning and rearing sites in the Upper SR, fall-run Chinook salmon now reside in waters that are generally cooler than the majority of historic spawning areas. Additionally, alteration of the Lower SR by hydroelectric dams has created a series of low-velocity pools in the SR that did not exist historically.

Life History

Prior to alteration of the SR basin by dams, fall Chinook salmon exhibited a largely ocean-type life history, where they migrated downstream and reared in the mainstem SR during their first year. Today, fall Chinook salmon in the SR Basin exhibit one of two life histories: ocean-type and reservoir-type (Connor, Sneva et al. 2005). The reservoir-type life history is one where juveniles overwinter in the pools created by the dams, prior to migrating out of the SR. The reservoir-type life history is likely a response to early development in cooler temperatures which prevents juveniles from reaching suitable size to migrate out of the SR.

Adult SR fall-run Chinook salmon enter the Columbia River in July and August. Spawning occurs from October through November. Juveniles emerge from gravels in March and April of the following year, moving downstream from natal spawning and early rearing areas from June through early fall.

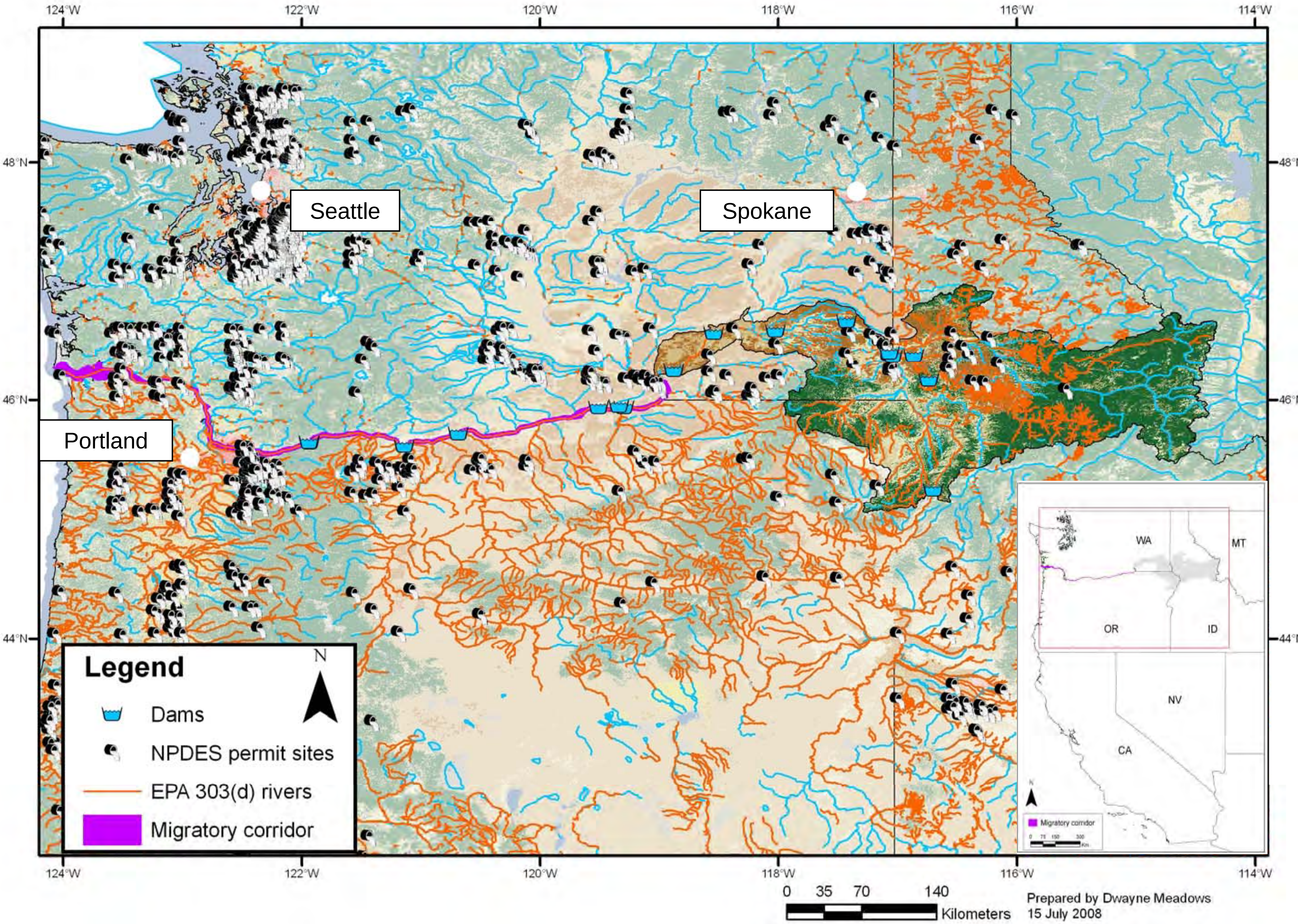

Figure 13. SR fall-run Chinook salmon distribution. Land Cover Class Legend in Figure 7..

Status and Trends

SR fall-run Chinook salmon were originally listed as threatened in 1992 (57 FR 14653). Their classification was reaffirmed following a status review on June 28, 2005 (70 FR 37160). Estimated annual returns for the period 1938 to 1949 was 72,000 fish. By the 1950s, numbers had declined to an annual average of 29,000 fish (Bjornn and Horner 1980). Numbers of SR fall-run Chinook salmon continued to decline during the 1960s and 1970s as approximately 80% of their historic habitat was eliminated or severely degraded by the construction of the Hells Canyon complex (1958 to 1967) and the lower SR dams (1961 to 1975). Counts of natural-origin adult SR fall-run Chinook salmon at Lower Granite Dam were 1,000 fish in 1975, and ranged from 78 to 905 fish (with an average of 489 fish) over the ensuing 25-year period (Good, Waples et al. 2005). Numbers of natural-origin SR fall-run Chinook salmon have increased over the last few years, with estimates at Lower Granite Dam of 2,652 fish in 2001, 2,095 fish in 2002, and 3,895 fish in 2003.

SR fall-run Chinook salmon have exhibited an upward trend in returns over Lower Granite Dam since the mid-1990s. Returns classified as natural-origin exceeded 2,600 fish in 2001, compared to a 1997-2001 geometric mean natural-origin count of 871. Long- and short-term trends in natural returns are positive. Harvest impacts on SR fall-run Chinook salmon declined after listing and have remained relatively constant in recent years. There have been major reductions in fisheries impacting this stock. Mainstem conditions for subyearling Chinook salmon migrants from the SR have generally improved since the early 1990s. The hatchery component, derived from outside the basin, has decreased as a percentage of the run at Lower Granite Dam from the 1998/99 status reviews (five year average of 26.2%) to 2001 (8%). This reflects an increase in the Lyons Ferry component, systematic removal of marked hatchery fish at the Lower Granite trap, and modifications to the Umatilla supplementation program to increase homing of fall Chinook release groups.

Overall abundance for SR fall-run Chinook salmon is relatively low, but has been increasing in the last decade (Good, Waples et al. 2005). The 1997 to 2001 geometric mean natural-origin count over Lower Granite Dam approximate 35% of the proposed delisting abundance criteria of 2,500 natural spawners averaged over 8 years. The recent abundance is approaching the delisting criteria. However, hatchery fish are faring better than wild fish.

Regarding productivity [population growth rate (lambda)], the long-term trend in total returns is >1; indicating the population size is growing. Although total abundance has dropped sharply in the past two years, it still remains at levels higher than previous decades. Productivity is likely sustained largely by a system of small artificial rearing facilities in the Lower SR Basin. The growth trend for natural-origin fish is close to 1, and could either be higher or lower, depending on the number of hatchery fish that spawn naturally.

The historic spatial structure has been reduced to one single remnant population. The ESU occupies a relatively small amount of marginal habitat, with the vast majority of historic habitat inaccessible. Genetic diversity is likely reduced from historic levels. Hatcheries affect ESU genetics due to three major components: natural-origin fish (which may be progeny of hatchery fish), returns of SR fall-run fish from the Lyons Ferry Hatchery program, and strays from hatchery programs outside the SR. Nevertheless, the SR fall-run Chinook salmon remains genetically distinct from similar fish in other basins. Phenotypic characteristics have shifted in apparent response to environmental changes from hydroelectric dams (Connor, Sneva et al. 2005).

The ICBTRT has defined only one extant population for the SR fall-run Chinook salmon, the lower SR mainstem population. This population occupies the SR from its confluence with the Columbia River to Hells Canyon Dam, and the lower reaches of the Clearwater, Imnaha, Grande Rhonde, Salmon, and Tucannonh Rivers (ICBTRT 2003).

Critical Habitat

Critical habitat for these salmon was designated on December 28, 1993 (58 FR 68543). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed SR fall-run salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 ft from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side), all river reaches from the estuary upstream to the confluence of the SR, and all SR reaches upstream to Hells Canyon Dam. Critical habitat also includes several river reaches presently or historically accessible to SR fall-run Chinook salmon. Limiting factors identified for SR fall-run Chinook salmon include: (1) Mainstem lower Snake and Columbia hydrosystem mortality, (2) degraded water quality, (3) reduced spawning and rearing habitat due to mainstem lower SR hydropower system, (4) harvest impacts, (5) impaired stream flows, barriers to fish passage in tributaries, excessive sediment, and (6) altered floodplain and channel morphology (NMFS 2005). The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Snake River Spring/Summer-Run Chinook Salmon

Distribution

SR spring/summer-run Chinook salmon are primarily limited to the Salmon, Grande Ronde, Imnaha, and Tucannon Rivers in the SR basin (Figure 14). The SR basin drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. Environmental conditions are generally drier and warmer in these areas than in areas occupied by other Chinook salmon species. The ICBTRT has identified 32 populations in five MPGs (Upper Salmon River, South Fork Salmon River, Middle Fork , Salmon

River, Grande Ronde/Imnaha, Lower Snake Mainstem Tributaries) for this species. Historic populations above Hells Canyon Dam are considered extinct (ICBTRT 2003). This ESU includes production areas that are characterized by spring-timed returns, summer-timed returns, and combinations from the two adult timing patterns. Historically, the Salmon River system may have supported more than 40% of the total run of spring and summer Chinook salmon to the Columbia system (Fulton 1968).

Some or all of the fish returning to several of the hatchery programs are also listed, including those returning to the Tucannon River, Imnaha River, and Grande Ronde River hatcheries, and to the Sawtooth, Pahsimeroi, and McCall hatcheries on the Salmon River. The Salmon River system contains a range of habitats used by spring/summer Chinook. The South Fork and Middle Fork Salmon Rivers currently support the bulk of natural production in the drainage. Returns into the upper Salmon River tributaries have reestablished following the opening of passage around Sunbeam Dam on the mainstem Salmon River downstream of Stanley, Idaho. The dam was impassable to anadromous fish from 1910 until the 1930s. Table 11 identifies populations within the SR spring/summer Chinook salmon ESU, their abundances, and hatchery input.

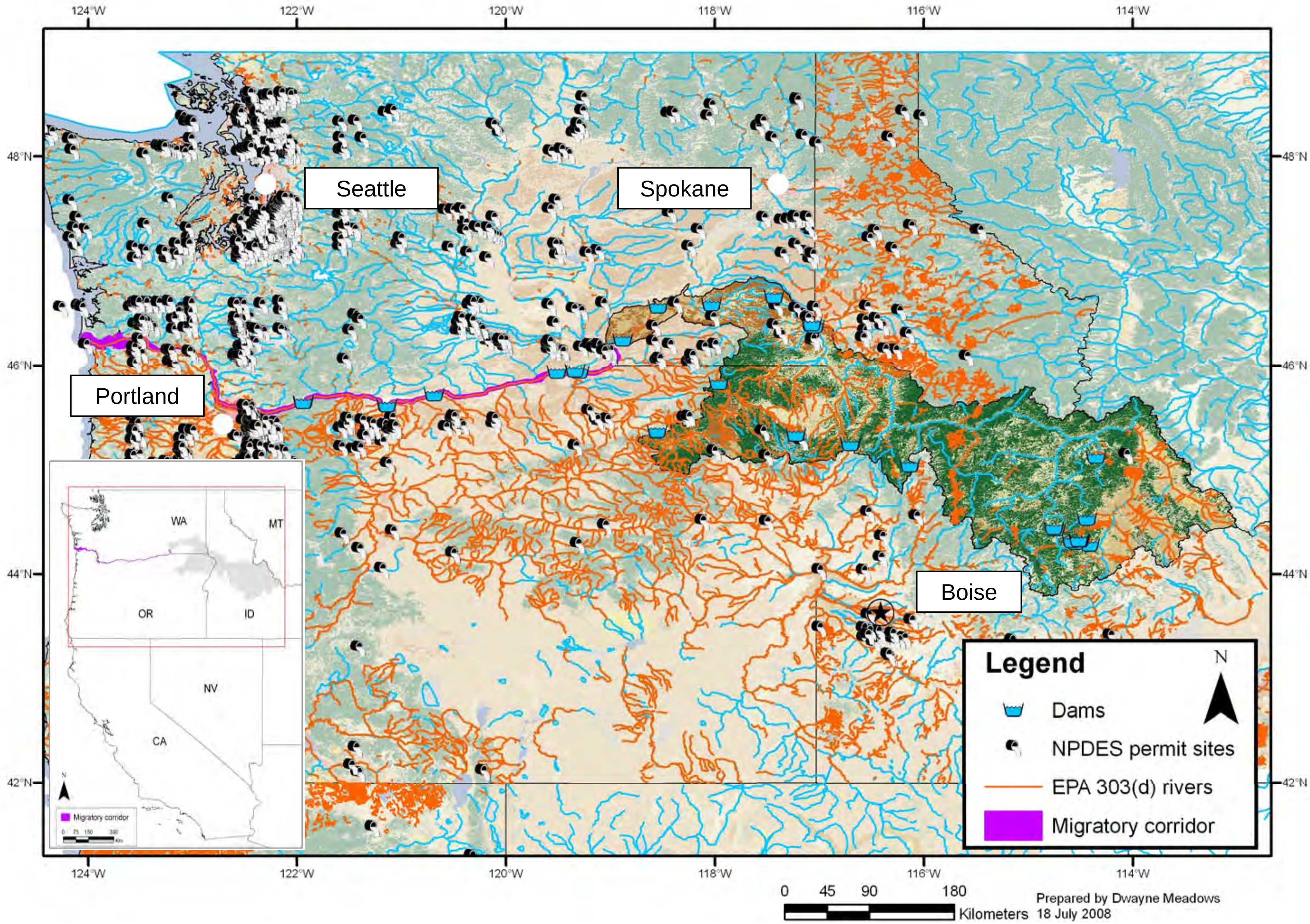

Figure 14. SR Spring/Summer-run Chinook salmon distribution. Land Cover Class Legend in Figure 7.

Table 11. SR Spring/Summer Chinook salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile.

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Tucannon River	Unknown	128-1,012	76%
Wenaha River	Unknown	67-586	64%
Wallowa River	Unknown	0-29 redds	5%
Lostine River	Unknown	9-131 redds	5%
Minam River	Unknown	96-573	5%
Catherine Creek	Unknown	13-262	56%
Upper Grande Ronde River	Unknown	3-336	58%
South Fork Salmon River	Unknown	277-679 redds	9%
Secesh River	Unknown	38-444 redds	4%
Johnson Creek	Unknown	49-444 redds	0%
Big Creek spring run	Unknown	21-296	0%
Big Creek summer run	Unknown	2-58 redds	Unknown
Loon Creek	Unknown	6-255 redds	0%
Marsh Creek	Unknown	0-164	0%
Bear Valley/Elk Creek	Unknown	72-712	0%
North Fork Salmon River	Unknown	2-19 redds	Unknown
Lemhi River	Unknown	35-216 redds	0%
Pahsimeroi River	Unknown	72-1,097	Unknown
East Fork Salmon spring run	Unknown	0.27 rpm	Unknown
East Fork Salmon summer run	Unknown	1.22 rpm	0%
Yankee Fork spring run	Unknown	0	Unknown
Yankee Fork summer run	Unknown	1-18 redds	0%
Valley Creek spring run	Unknown	2-28 redds	0%
Valley Creek summer run	Unknown	2.14 rpm	Unknown
Upper Salmon spring run	Unknown	25-357 redds	Unknown
Upper Salmon summer run	Unknown	0.24 rpm	Unknown
Alturas Lake Creek	Unknown	0-18 redds	Unknown
Imnaha River	Unknown	194-3,041 redds	62%
Big Sheep Creek	Unknown	0.25 redds	97%
Lick Creek	Unknown	0-29 redds	59%
Total	**~1.5 million**	**~9,700**	

Life History

SR spring/summer-run Chinook salmon exhibit a stream-type life history. Eggs are deposited in late summer and early fall, incubate over the following winter, and hatch in late winter and early spring of the following year. Juvenile fish mature in fresh water for one year before they migrate to the ocean in the spring of their second year of life. Depending on the tributary and the specific habitat conditions, juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. SR spring/summer-run Chinook salmon return from the ocean to spawn primarily as four and five-year old fish, after two to three years in the ocean. A small fraction of the fish return as three year-old "jacks", heavily predominated by males.

Status and Trends

SR spring/summer-run Chinook salmon were originally listed as threatened on April 22, 1992 (57 FR 14653). Their classification was reaffirmed following a review on June 28, 2005 (70 FR 37160). Although direct estimates of historical annual SR spring/summer Chinook salmon returns are not available, returns may have declined by as much as 97% between the late 1800s and 2000. According to Matthews and Waples (1997), total annual SR spring/summer Chinook salmon production may have exceeded 1.5 million adult fish in the late 1800s. Total (natural plus hatchery origin) returns fell to roughly 100,000 spawners by the late 1960s (Fulton 1968) and were below 10,000 by 1980. Between 1981 and 2000, total returns fluctuated between extremes of 1,800 and 44,000 fish. The 2001 and 2002 total returns increased to over 185,000 and 97,184 adults, respectively. The 1997 to 2001 geometric mean total return for the summer run component at Lower Granite Dam was slightly more than 6,000 fish, compared to the geometric mean of 3,076 fish for the years 1987 to 1996. The 2002 to 2006 geometric mean of the combined Chinook salmon runs at Lower Granite Dam was over 18,000 fish. However, over 80% of the 2001 return and over 60% of the 2002 return originated in hatcheries (Good, Waples et al. 2005). Good et al. (2005) reported that risks to individual populations within the ESU may be greater than the extinction risk for the entire ESU due to low levels of annual abundance and the extensive production areas within the SR basin. Year-to-year abundance has high variability and is most pronounced in natural-origin fish. Although the average abundance in the most recent decade is more abundant than the previous decade, there is no obvious long-term trend. Additionally, hatchery fish are faring better than wild fish, which comprise roughly 40% of the total returns in the past decade. Overall, most populations are far below their respective interim recovery targets.

Regarding population growth rate (lambda), long-term trends are <1; indicating the population size is shrinking. However, recent trends, buoyed by last 5 years, are approaching 1. Nevertheless, many spawning aggregates have been extirpated, which has increased the spatial separation of some populations. Populations are widely distributed in a diversity of habitats although roughly one-half of historic habitats are inaccessible. There is no evidence of wide-scale genetic introgression by hatchery populations. The high variability in life history traits indicates sufficient genetic variability within the DPS to maintain distinct subpopulations adapted to local environments. Despite the recent increases in total spring/summer-run Chinook salmon returns to the basin, natural-origin abundance and productivity remain below their targets. SR spring/summer Chinook salmon remains likely to become endangered (Good, Waples et al. 2005).

Critical Habitat

Critical habitat for these salmon was designated on October 25, 1999 (64 FR 57399). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed SR salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 ft from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side). Critical habitat also includes all river reaches from the estuary upstream to the confluence of the SR, and all SR reaches upstream to Hells Canyon Dam; the Palouse River from its confluence with the SR upstream to Palouse Falls, the Clearwater River from its confluence with the SR upstream to its confluence with Lolo Creek; the North Fork Clearwater River from its confluence with the Clearwater river upstream to Dworshak Dam.

Limiting factors identified for this species include: (1) Hydrosystem mortality, (2) reduced stream flow, (3) altered channel morphology and floodplain, (4) excessive fine sediment, and (5) degraded water quality (Myers, Kope et al. 1998). The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Upper Willamette River Chinook Salmon

Distribution

Upper Willamette River (UWR) Chinook salmon occupy the Willamette River and tributaries upstream of Willamette Falls (Figure 15). In the past, this ESU included sizable numbers of spawning salmon in the Santiam River, the middle fork of the Willamette River, and the McKenzie River, as well as smaller numbers in the Molalla River, Calapooia River, and Albiqua Creek. Historically, access above Willamette Falls was restricted to the spring when flows were high. In autumn, low flows prevented fish from ascending past the falls. The UWR Chinook salmon are one of the most genetically distinct Chinook salmon groups in the Columbia River Basin. Fall-run Chinook salmon spawn in the Upper Willamette but are not considered part of the species because they are not native. None of the hatchery populations in the Willamette River were listed although five spring-run hatchery stocks were included in the species' listing. UWR Chinook salmon migrate far north and are caught incidentally in ocean fisheries, particularly off southeast Alaska and northern Canada, and in spring season fisheries in the mainstem Columbia and Willamette Rivers. Table 12 identifies populations within the UWR Chinook salmon ESU, their abundances, and hatchery input

Table 12. UWR Chinook salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Clackamas River	Unknown	2,910	64%
Molalla River	Unknown	52 redds	>93%
North Santiam River	Unknown	~ 7.1 rpm	>95%
South Santiam River	Unknown	982 redds	>84%
Calapooia River	Unknown	16 redds	100%
McKenzie River	Unknown	~2,470	26%
Middle Fork Willamette River	Unknown	235 redds	>39%
Upper Fork Willamette River	Unknown	Unknown	Unknown
Total	**>70,000**	**~9,700**	Mostly hatchery

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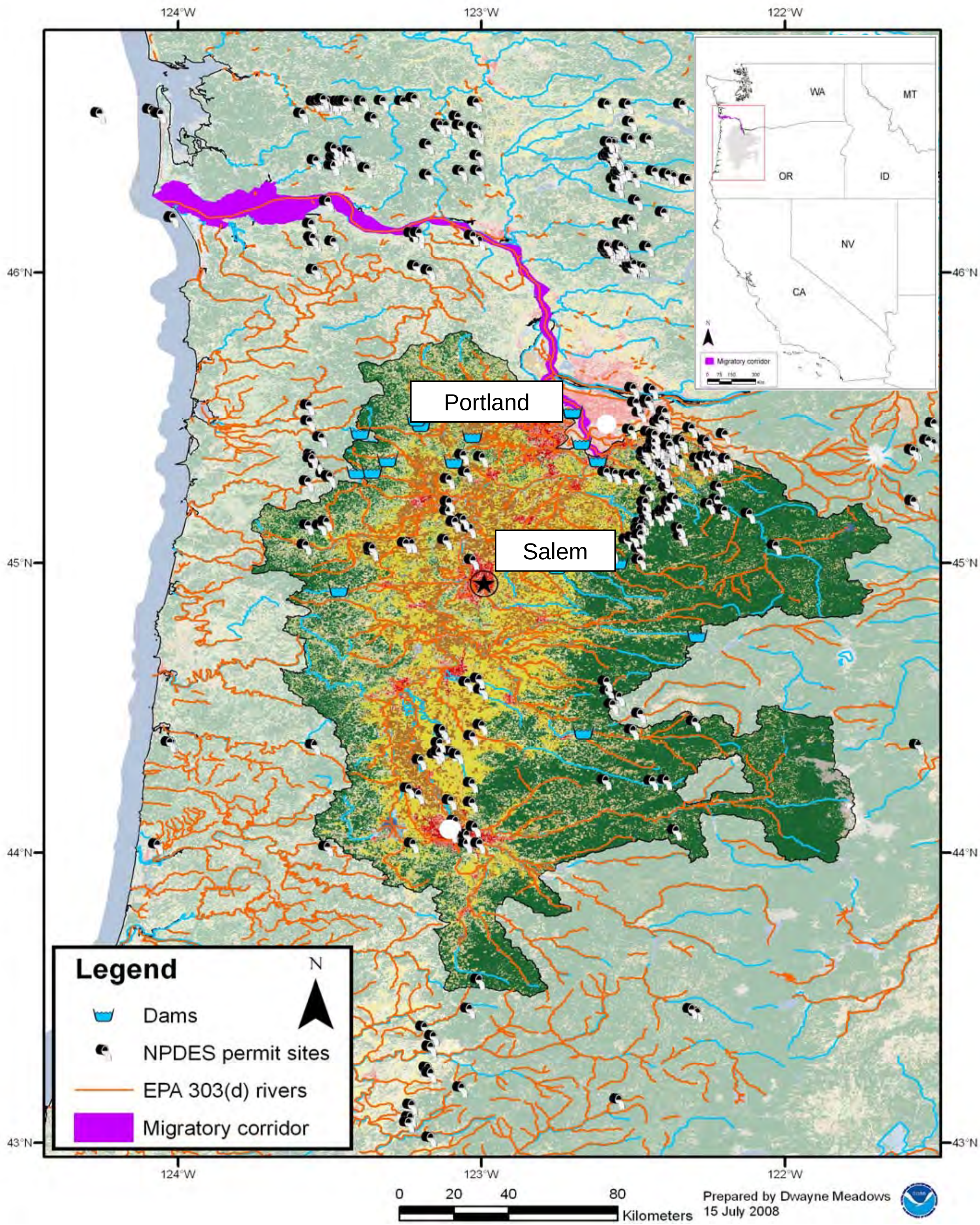


Figure 15. UWR Chinook salmon distribution. Land Cover Class Legend in Figure 7.

Life History

UWR Chinook salmon exhibit an earlier time of entry into the Columbia River and estuary than other spring Chinook salmon ESUs (Meyers, Kope et al. 1998). Although juveniles from interior spring Chinook salmon populations reach the mainstem migration corridor as yearling, some juvenile Chinook salmon in the lower Willamette River are subyearlings (Friesen, Vile et al. 2004).

Status and Trends

UWR Chinook salmon were listed as threatened on March 24, 1999 (64 FR 14308), and reaffirmed as threatened on June 28, 2005 (70 FR 37160). The total abundance of adult spring-run Chinook salmon (hatchery-origin + natural-origin fish) passing Willamette Falls has remained relatively steady over the past 50 years (ranging from approximately 20,000 to 70,000 fish). However, it is an order of magnitude below the peak abundance levels observed in the 1920s (approximately 300,000 adults). Until recent years, interpretation of abundance levels has been confounded by a high but uncertain fraction of hatchery-produced fish.

Most natural spring Chinook salmon populations is likely extirpated or nearly so. Only one remaining naturally reproducing population is identified in this ESU: the spring Chinook salmon in the McKenzie River. Unfortunately, recent short-term declines in abundance suggest that this population may not be self-sustaining (Meyers, Kope et al. 1998; Good, Waples et al. 2005). Most of the natural-origin populations in this ESU have very low current abundances (less than a few hundred fish) and many largely have been replaced by hatchery production. Long- and short-term trends for population growth rate are approximately 1 or are negative, depending on the metric examined (*i.e.*, long-term trend [regression of log-transformed spawner abundance] or lambda [median population growth rate]). Although the population increased substantially in 2000-2003, it was probably due to increased survival in the ocean. Future survival rates in the ocean are unpredictable, and the likelihood of long-term sustainability for this population has not been determined. Although the number of adult spring-run Chinook salmon crossing Willamette Falls is in the same range (about 20,000 to 70,000 adults) it has been for the last 50 years, a large fraction of these are hatchery produced. Of concern is that a majority of the spawning habitat and approximately 30 to 40% of total historical habitat are no longer accessible because of dams (Good, Waples et al. 2005).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Willamette River as well as specific stream reaches in a number of subbasins. The critical habitat designation for this ESU also identifies PCEs that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning and rearing sites, freshwater migration corridors. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 65 subbasins reviewed in NMFS' assessment of critical habitat for the UWR Chinook salmon ESU, 19 subbasins were rated as having a medium conservation value, 19 were rated as low, and the remaining subbasins (27), were rated as having a high conservation value to UWR Chinook salmon. Federal lands were generally rated as having high conservation value to the species' spawning and rearing. Factors contributing to the downward trends in this ESU include: (1) Reduced access to spawning/rearing habitat in tributaries, (2) hatchery impacts, (3) altered water quality and temperature in tributaries, (4) altered stream flow in tributaries, and (5) lost/degraded floodplain connectivity and lowland stream habitat.

Chum Salmon

Description of the Species

Chum salmon has the widest natural geographic and spawning distribution of any Pacific salmonid because its range extends farther along the shores of the Arctic Ocean than other salmonids. Chum salmon have been documented to spawn from Korea and the Japanese island of Honshu, east around the rim of the North Pacific Ocean to Monterey

Bay, California. Historically, chum salmon were distributed throughout the coastal regions of western Canada and the U.S. Presently, major spawning populations are found only as far south as Tillamook Bay on the northern Oregon coast. We discuss the distribution, life history diversity, status, and critical habitat of the two species of threatened chum salmon separately.

Chum salmon are semelparous, spawn primarily in freshwater, and exhibit obligatory anadromy (there are no recorded landlocked or naturalized freshwater populations). Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea. North American chum salmon (as opposed to chum salmon originating in Asia) rarely occur west of 175° E longitude.

North American chum salmon migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. However, some data suggest that Puget Sound chum, including Hood Canal summer run chum, may not make extended migrations into northern British Columbian and Alaskan waters. Instead, they may travel directly offshore into the north Pacific Ocean.

Chum salmon, like pink salmon, usually spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to nearly 100 km from the sea. Juveniles outmigrate to seawater almost immediately after emerging from the gravel that covers their redds (Salo 1991). The immature salmon distribute themselves widely over the North Pacific Ocean. The maturing adults return to the home streams at various ages, usually at two through five years, and at some cases up to seven years (Bigler 1985). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus *Oncorhynchus* (*e.g.*, coastal cutthroat trout, steelhead, coho salmon, and most types of Chinook and sockeye salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. This means that survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions. Another behavioral difference between chum salmon and species that rear extensively in freshwater is that chum salmon form schools. Presumably, this behavior reduces predation (Pitcher 1986), especially if fish movements are synchronized to swamp predators (Miller and Brannon 1982).

The duration of estuarine residence for chum salmon juveniles are known for only a few estuaries. Observed residence times range from 4 to 32 days; with a period of about 24 days being the most common (Johnson, Grant et al. 1997). Juvenile salmonids rely on a variety of non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle. Examples of off-channel habitat include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, and braids (Anderson 1999; Swift III 1979).

Status and Trends

Chum salmon have been threatened by overharvests in commercial and recreational fisheries, adult and juvenile mortalities associated with hydropower systems, habitat degradation from forestry and urban expansion, and shifts in climatic conditions that changed patterns and intensity of precipitation.

Chum salmon, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy or alter wetland and riparian ecosystems. The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Columbia River Chum Salmon

Distribution

Columbia River chum salmon includes all natural-origin chum salmon in the Columbia River and its tributaries in Oregon and Washington. The species consists of three populations: Grays River, Hardy, and Hamilton Creek in Washington State (Figure 16).

This ESU also includes three artificial hatchery programs. There were 16 historical populations in three MPGs in Oregon and Washington between the mouth of the Columbia River and the Cascade crest. Significant spawning now occurs for two of the historical populations. About 88% of the historical populations are extirpated. Table 10 identifies populations within the Columbia River Chum salmon ESU, their abundances, and hatchery input.

Table 13. Columbia River Chum salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay	Unknown	0	0
Grays River	7,511	331-704	Unknown
Big Creek	Unknown	0	0
Elochoman River	Unknown	0	0
Clatskanie River	Unknown	0	0
Mill, Abernathy, and German Creeks	Unknown	0	0
Scappoose Creek	Unknown	0	0
Cowlitz River	141,582	0	0
Kalama River	9,953	0	0
Lewis River	89,671	0	0
Salmon Creek	Unknown	0	0
Clackamus River	Unknown	0	0
Sandy River	Unknown	0	0
Washougal River	15,140	0	0
Lower gorge tributaries	>3,141	425	0
Upper gorge tributaries	>8,912	0	0
Total	**>283,421**	**756-1,129**	

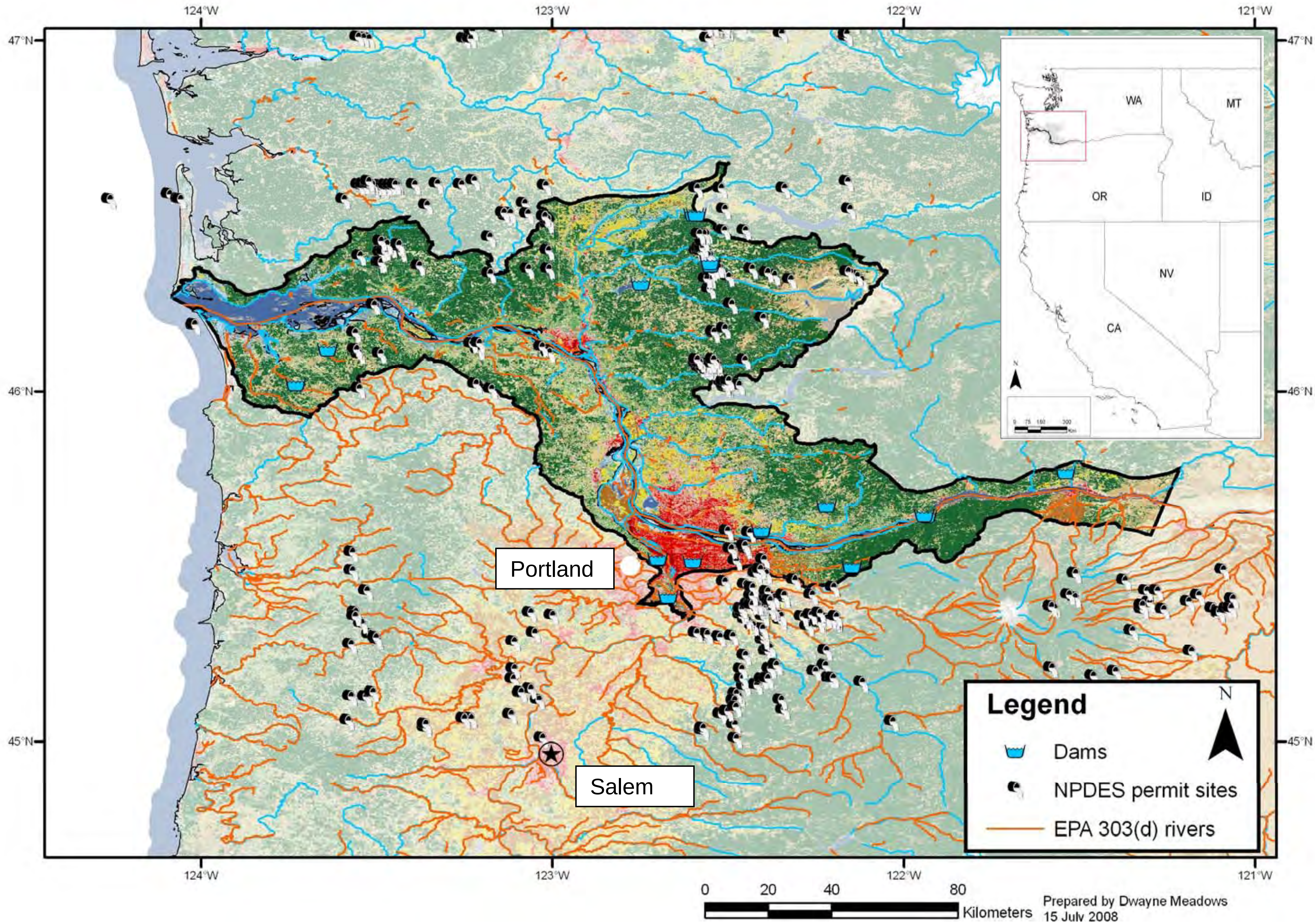


Figure 16. Columbia River Chum salmon distribution. Land Cover Class Legend in Figure 7.

Life History

Chum salmon return to the Columbia River in late fall (mid-October to December). They primarily spawn in the lower reaches of rivers, digging redds along the edges of the mainstem and in tributaries or side channels. Some spawning sites are located in areas where geothermally-warmed groundwater or mainstem flow upwells through the gravel. Chum salmon fry emigrate from March through May shortly after emergence in contrast to other salmonids (*e.g.*, steelhead, coho salmon, and most Chinook salmon), which usually migrate to sea at a larger size after months or years of freshwater rearing. Juvenile chum salmon reside in estuaries to feed before beginning a long-distance oceanic migration. Chum salmon may choose either the upper or lower estuaries depending on the relative productivity of each. The timing of entry of juvenile chum salmon into sea water is commonly correlated with the warming of the nearshore waters and the accompanying plankton blooms (Burgner 1991). The movement offshore generally coincides with the decline of inshore prey resources and is normally at the time when the fish has grown to a size that allows them to feed upon neritic organisms and and avoid predators (Burgner 1991).

Although most juvenile chum salmon migrate rapidly from freshwater to shallow nearshore marine habitats after emergence from gravel beds, some may remain up to a year in fresh water in large northern rivers. The period of estuarine residence appears to be a critical life history phase and may play a major role in determining the size of the subsequent adult run back to freshwater.

Status and Trends

Columbia River chum salmon were listed as threatened on March 25, 1999, and their threatened status was reaffirmed on June 28, 2005 (71 FR 37160). Chum salmon in the Columbia River once numbered in the hundreds of thousands of adults and were reported in almost every river in the LCR basin. However, by the 1950s most runs disappeared (Rich 1942; Marr 1943; Fulton 1968). The total number of chum salmon returning to the Columbia River in the last 50 years has averaged a few thousand per year, with returns limited to a very restricted portion of the historical range. Significant spawning occurs in only two of the 16 historical populations. Nearly 88% of the historical populations are extirpated. The two remaining populations are the Grays River and the Lower Gorge (Good, Waples et al. 2005). Chum salmon appear to be extirpated from the Oregon portion of this ESU. In 2000, the Oregon Department of Fish and Wildlife (ODFW) conducted surveys to determine the abundance and distribution of chum salmon in the Columbia River. Of 30 sites surveyed, only one chum salmon was observed.

Historically, the Columbia River chum salmon supported a large commercial fishery in the first half of this century which landed more than 500,000 fish per year as recently as 1942. Commercial catches declined beginning in the mid-1950s, and in later years rarely exceeded 2,000 per year. During the 1980s and 1990s, the combined abundance of natural spawners for the Lower Gorge, Washougal, and Grays River populations was below 4,000 adults. In 2002, however, the abundance of natural spawners exhibited a substantial increase at several locations (estimate of natural spawners is approximately 20,000 adults). The cause of this dramatic increase in abundance is unknown. Estimates of abundance and trends are available only for the Grays River and Lower Gorge populations. The 10-year trend was negative for the Grays River population and just over 1.0 for the Lower Gorge. The Upper Gorge population, and all four of the populations on the Oregon side of the river in the Coastal MPG, are extirpated or nearly so (McElhaney, Chilcote et al. 2007). However, long- and short-term productivity trends for populations are at or below replacement. Regarding spatial structure, few Columbia River chum salmon have been observed in tributaries between The Dalles and Bonneville dams. Surveys of the White Salmon River in 2002 found one male and one female carcass and the latter had not spawned (Ehlke and Keller 2003). Chum salmon were not observed in any of the upper gorge tributaries, including the White Salmon River, during the 2003 and 2004 spawning ground surveys. Finally, most Columbia River chum populations have been functionally extirpated or are presently at very low abundance levels. The loss of off-channel habitat and the extirpation of approximately 17 historical populations increase this species' vulnerability to environmental variability and catastrophic events. Overall, the populations that remain have low abundance, limited distribution, and poor connectivity (Good, Waples et al. 2005).

Critical Habitat

Critical habitat was originally designated for this species on February 16, 2000 (65 FR 7764) and was re-designated on September 2, 2005 (70 FR 52630). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more chum salmon life stages. Columbia River chum salmon have PCEs of: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 21 subbasins reviewed in NMFS' assessment of critical habitat for the Columbia River chum salmon ESU, three subbasins were rated as having a medium conservation value, no subbasins were rated as low, and the majority of subbasins (18), were rated as having a high conservation value to Columbia River chum salmon. Washington's federal lands were rated as having high conservation value to the species. The major factors limiting recovery for Columbia River chum salmon are altered channel form and stability in tributaries, excessive sediment in tributary spawning gravels, altered stream flow in tributaries and the mainstem Columbia River, loss of some tributary habitat types, and harassment of spawners in the tributaries and mainstem. The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Hood Canal Summer-Run Chum Salmon

Distribution

This ESU includes all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries as well as populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington (64 FR 14508, Figure 17). Eight artificial propagation programs are considered as part of the ESU: the Quilcene National Fish Hatchery, Hamma Hamma Fish Hatchery, Lilliwaup Creek Fish Hatchery, Union River/Tahuya, Big Beef Creek Fish Hatchery, Salmon Creek Fish Hatchery, Chimacum Creek Fish Hatchery, and the Jimmycomelately Creek Fish Hatchery summer-run chum hatchery programs. NMFS determined that these artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the species. Table 14 identifies populations within the Hood Canal summer-run Chum salmon ESU, their abundances, and hatchery input.

Table 14. Hood Canal summer-run Chum salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Jimmycomelately Creek	Unknown	~60	Unknown
Salmon/Snow creeks	Unknown	~2,200	0-69%
Big/Little Quilcene rivers	Unknown	~4,240	5-51%
Lilliwaup Creek	Unknown	~164	Unknown
Hamma Hamma River	Unknown	~758	Unknown
Duckabush River	Unknown	Unknown	Unknown
Dosewallips River	Unknown	~900	Unknown
Union River	Unknown	~690	Unknown
Chimacum Creek	Unknown	0	100
Big Beef Creek	Unknown	0	100
Dewetto Creek	Unknown	0	Unknown
Total	**Unknown**	**~9,012**	

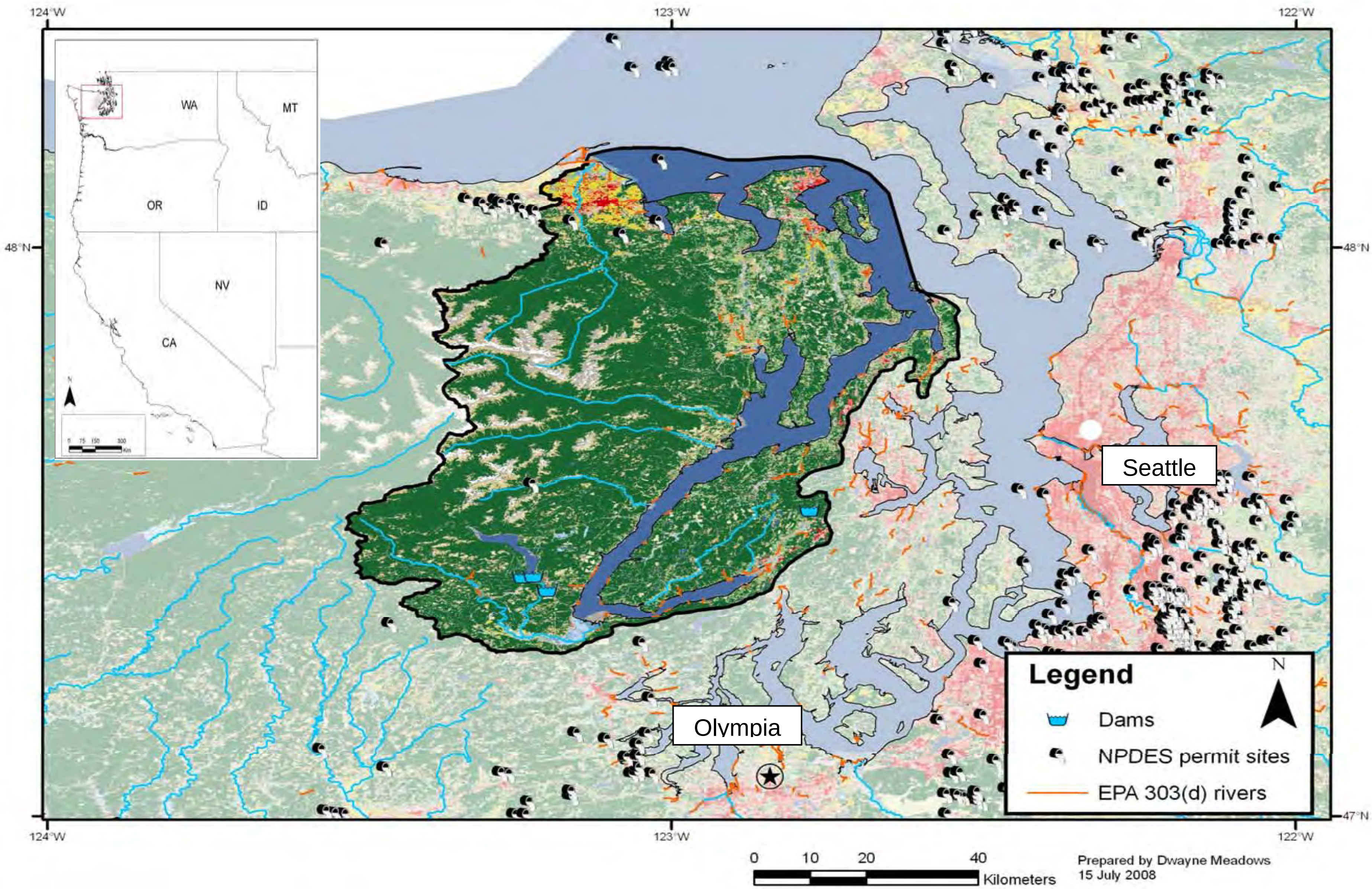


Figure 17. Hood Canal Summer-run Chum salmon distribution. Land Cover Class Legend in Figure 7.

Life History

The Hood Canal summer-run Chum salmon are defined in the Salmon and Steelhead Stock Inventory (WDF, WDW et al. 1993) as fish that spawn from mid-September to mid-October. However, summer chum have been known to enter natal rivers in late August. Fall-run chum salmon are defined as fish that spawn from November through December or January. Run-timing data for as early as 1913 indicated temporal separation between summer and fall chum salmon in Hood Canal (Johnson, Grant et al. 1997). Hood Canal summer Chum salmon are genetically distinct from healthy populations of Hood Canal fall Chum salmon originating within this area. Hood Canal summer Chum salmon return to natal rivers to spawn during the August through early October period. The fall Chum salmon spawn between November and December, when streams are higher and water temperature is lower.

The time to hatching varies among populations and among individuals within a population (Salo 1991). Fry tend to emerge when they had their best chances of surviving in streams and estuaries (Koski 1975). A variety of factors may influence the time to hatching, emergence from the gravel, or both. They include dissolved oxygen, gravel size, salinity, nutritional conditions, behaviour of alevins in the gravel and incubation temperature [reviewed in (Bakkala 1970; Schroder, Koski et al. 1974; Schroder 1977; Salo 1991)]. The average residence time in estuaries for Hood canal chum salmon is 23 days. Fry in Hood Canal have not been observed to display daily tidal migrations (Bax 1983). Fry movement is associated with prey availability. Summer-run chum salmon migrate up the Hood Canal and into the main body of Puget Sound. Fish may emerge from streams over an extended period or juveniles may also remain in Quilcene Bay for several weeks.

Status and Trends

Hood Canal summer-run Chum salmon were listed as threatened on March 25, 1999, and reaffirmed as threatened on June 28, 2005 (70 FR 37160). Adult returns for some populations in the Hood Canal summer-run Chum salmon species showed modest improvements in 2000, with upward trends continuing in 2001 and 2002. The recent five-year mean abundance is variable among populations in the species, ranging from one fish to nearly 4,500 fish. Hood Canal summer-run chum salmon are the focus of an extensive rebuilding program developed and implemented since 1992 by the state and tribal co-managers. Two populations (the combined Quilcene and Union River populations) are above the conservation thresholds established by the rebuilding plan. However, most populations remain depressed. Estimates of the fraction of naturally spawning hatchery fish exceed 60% for some populations. This indicates that reintroduction programs are supplementing the numbers of total fish spawning naturally in streams. Long-term trends in productivity are above replacement for only the Quilcene and Union River populations. Buoyed by recent increases, seven populations are exhibiting short-term productivity trends above replacement.

Of an estimated 16 historical populations in the ESU, seven populations are believed to have been extirpated or nearly extirpated. Most of these extirpations have occurred in populations on the eastern side of Hood Canal, generating additional concern for ESU spatial structure. The widespread loss of estuary and lower floodplain habitat was noted by the BRT as a continuing threat to ESU spatial structure and connectivity. There is some concern that the Quilcene hatchery stock is exhibiting high rates of straying, and may represent a risk to historical population structure and diversity. However, with the extirpation of many local populations, much of this historical structure has been lost, and the use of Quilcene hatchery fish may represent one of a few remaining options for Hood Canal summer-run Chum salmon conservation.

Of the eight programs releasing summer chum salmon that are considered to be part of this ESU, six of the programs are supplementation programs implemented to preserve and increase the abundance of native populations in their natal watersheds. NMFS' assessment of the effects of artificial propagation on ESU extinction risk concluded that these hatchery programs collectively do not substantially reduce the extinction risk of the ESU. The hatchery programs are reducing risks to ESU abundance by increasing total ESU abundance as well as the number of naturally spawning summer-run chum salmon.

Several of the programs have likely prevented further population extirpations in the ESU. The contribution of ESU hatchery programs to the productivity of the ESU in-total is uncertain. The hatchery programs are benefiting ESU spatial structure by increasing the spawning area utilized in several watersheds and by increasing the geographic range of the ESU through reintroductions. These programs also provide benefits to ESU diversity. By bolstering total population sizes, the hatchery programs have likely stemmed adverse genetic effects for populations at critically low levels. Additionally, measures have been implemented to maintain current genetic diversity, including the use of native broodstock and the termination of the programs after 12 years of operation to guard against long-term domestication effects. Collectively, artificial propagation programs in the ESU presently provide a slight beneficial effect to ESU abundance, spatial structure, and diversity. However, artificial propagation programs also provide uncertain effects to ESU productivity.

Critical Habitat

Critical habitat for this species was designated on September 2, 2005 (70 FR 52630). Hood Canal summer-run chum salmon have PCEs of: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 17 subbasins reviewed in NMFS' assessment of critical habitat for the Hood Canal chum salmon ESU, 14 subbasins were rated as having a high conservation value, while only three were rated as having a medium value to the conservation. Limiting factors identified for this species include: (1) Degraded floodplain and mainstem river channel structure, (2) degraded estuarine water quality conditions and loss of estuarine habitat, (3) riparian area degradation and loss of in-river wood in mainstem, (4) excessive sediment in spawning gravels, and (5) reduced stream flow in migration areas. These conditions also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Coho Salmon

Description of the Species

Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle, Pauley et al. 1986). We discuss the distribution, life history diversity, status, and critical habitat of the four endangered and threatened coho species separately.

After entering the ocean, immature coho salmon initially remain in nearshore waters close to the parent stream. Most coho salmon adults are three-year-olds, having spent approximately 18 months rearing in freshwater and 18 months in salt water. Most coho salmon enter rivers between September and February. However, entry is influenced by discharge and other factors. In many systems, coho salmon and other Pacific salmon are unable to enter the rivers until sufficiently strong flows open passages and provide sufficient depth. Wild female coho salmon return to spawn almost exclusively at age three. Coho salmon spawn from November to January, and occasionally into February and March. Spawning occurs in a few third-order streams. Most spawning activity occurs in fourth- and fifth-order streams. Spawning generally occurs in tributaries with gradients of 3% or less.

Eggs incubate for about 35 to 50 days, and start emerging from the gravel within two to three weeks after hatching. Following emergence, fry move to shallow areas near the stream banks. As fry grow, they disperse upstream and downstream to establish and defend territories. Juvenile rearing usually occurs in tributaries with gradients of 3% or less, although they may move to streams with gradients of 4 to 5%. Juvenile coho salmon are often found in small streams less than five ft wide, and may migrate considerable distances to rear in lakes and off-channel ponds. During the summer, fry prefer pools featuring adequate cover such as large woody debris, undercut banks, and overhanging vegetation. Overwintering tends to occur in larger pools and backwater areas.

North American coho salmon will migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. During this migration, juvenile coho salmon tend to occur in both coastal and offshore waters. During spring and summer, coho salmon will forage in waters between 46ºN, the Gulf of Alaska, and along Alaska's Aleutian Islands.

Status and Trends

Coho salmon survive only in aquatic ecosystems and depend on the quantity and quality of those aquatic systems. Coho salmon, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems. The above activities and features introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Central California Coast Coho Salmon

Distribution

The CCC coho salmon ESU extends from Punta Gorda in northern California south to and including the San Lorenzo River in central California (Weitkamp, Wainwright et al. 1995). Table 15 identifies populations within the CCC Coho salmon ESU, their abundances, and hatchery input (Figure 18).

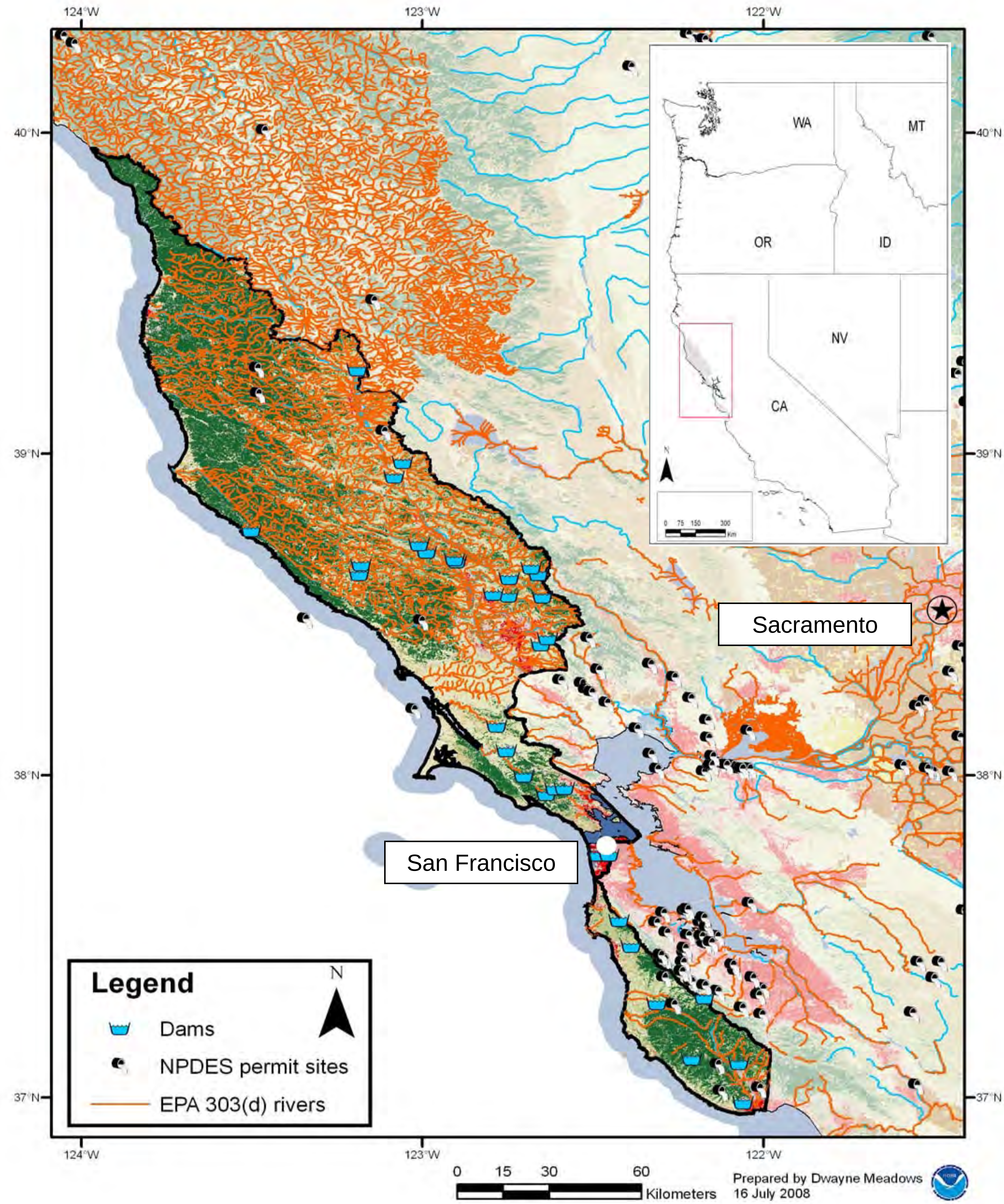


Figure 18. CCC Coho salmon distribution. Land Cover Class Legend in Figure 7.

Table 15. CCC Coho salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River/Region	Historical Escapement (1963)	1987-1991 Escapement Abundance	Hatchery Abundance Contributions
Ten Mile River	6,000	160	Unknown
Noyo River	6,000	3,740	Unknown
Big River	6,000	280	Unknown
Navarro River	7,000	300	Unknown
Garcia River	2,000	500 (1984-1985)	Unknown
Other Mendacino County rivers	10,000	470	Unknown
Gualala River	4,000	200	Unknown
Russian River	5,000	255	Unknown
Other Sonoma County rivers	1,000	180	Unknown
Marin County	5,000	435	Unknown
San Mateo County	1,000	Unknown	Unknown
Santa Cruz County	1,500	50 (1984-1985)	Unknown
San Lorenzo River	1,600	Unknown	Unknown
Total	**200,000-500,000**	**6,570 (min)**	

Life History

Both run and spawn timing of coho salmon in this region are very late (both peaking in January), with little time spent in freshwater between river entry and spawning. This compressed adult freshwater residency appears to coincide with the single, brief peak of river flow characteristic of this area.

Status and Trends

The CCC coho salmon ESU was originally listed as threatened under the ESA on October 31, 1996 (61 FR 56138) and later revised to endangered status on June 28, 2005 (70 FR 37160). The ESU includes all naturally spawned populations of coho salmon from Punta Gorda in northern California south to and including the San Lorenzo River in central California, as well as populations in tributaries to San Francisco Bay, excluding the Sacramento-San Joaquin River system. The ESU also includes four artificial propagation programs: the Don Clausen Fish Hatchery Captive Broodstock Program, Scott Creek/King Fisher Flats Conservation Program, Scott Creek Captive Broodstock Program, and the Noyo River Fish Station egg-take Program coho hatchery programs.

Information on the abundance and productivity trends for the naturally spawning component of the CCC coho salmon ESU is extremely limited. There are no long-term time series of spawner abundance for individual river systems. Analyses of juvenile coho presence-absence information, juvenile density surveys, and irregular adult counts for the South Fork Noyo River indicate low abundance and long-term downward trends for the naturally spawning populations throughout the ESU. Improved ocean conditions coupled with favorable stream flows and harvest restrictions have contributed to increased returns in 2001 in streams in the northern portion of the ESU, as indicated by an increase in the observed presence of fish in historically occupied streams. Data are lacking for many river basins in the southern two thirds of the ESU where naturally spawning populations are considered at the greatest risk. The extirpation or near extirpation of natural coho salmon populations in several major river basins, and across most of the southern historical range of the ESU, represents a significant risk to ESU spatial structure and diversity. Artificial propagation of coho salmon within the CCC ESU has declined since the ESU was listed in 1996 though it continues at the Noyo River and Scott Creek facilities, and two captive broodstock populations have recently been established. Genetic diversity risk associated with out-of-basin transfers appears to be minimal. However, diversity risk from domestication selection and low effective population sizes in the remaining hatchery programs remains a concern. An out-of-ESU artificial propagation program for coho was operated at the Don Clausen hatchery on the Russian River through the mid-1990s. However, the program was terminated in 1996. Termination of this program was considered by the Biological Review Team (BRT) as a positive development for naturally produced coho salmon in this ESU.

CCC coho salmon populations continue to be depressed relative to historical numbers. Strong indications show that breeding groups have been lost from a significant percentage of streams in their historical range. A number of coho salmon populations in the southern portion of the range appear to be either extinct or nearly so. They include those in Gualala, Garcia, and Russian rivers, as well as smaller coastal streams in and south of San Francisco Bay (Good, Waples et al. 2005). For the naturally spawning component of the ESU, the BRT found very high risk (of extinction) for the abundance, productivity, and spatial structure VSP parameters and comparatively moderate risk with respect to the diversity VSP parameter. The lack of direct estimates of the performance of the naturally spawned populations in this ESU, and the associated uncertainty this generates, was of specific concern to the BRT, as the naturally spawned component of the CCC coho salmon ESU was "in danger of extinction".

Critical Habitat

Critical habitat for the CCC coho salmon ESU was designated on May 5, 1999 (64 FR 24049). Designated critical habitat encompasses accessible reaches of all rivers (including estuarine areas and tributaries) between Punta Gorda and the San Lorenzo River (inclusive) in California. Critical habitat for this species also includes two streams entering San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creek.

Lower Columbia River Coho Salmon

Distribution

LCR coho salmon include all naturally spawned populations of coho salmon in the Columbia River and its tributaries in Oregon and Washington, from the mouth of the Columbia up to and including the Big White Salmon and Hood Rivers, and includes the Willamette River to Willamette Falls, Oregon (Figure 19). This ESU also includes 25 artificial propagation programs: the Grays River, Sea Resources Hatchery, Peterson Coho Project, Big Creek Hatchery, Astoria High School Coho Program, Warrenton High School Coho Program, Elochoman Type-S Coho Program, Elochoman Type-N Coho Program, Cathlamet High School FFA Type-N Coho Program, Cowlitz Type-N Coho Program in the Upper and Lower Cowlitz Rivers, Cowlitz Game and Anglers Coho Program, Friends of the Cowlitz Coho Program, North Fork Toutle River Hatchery, Kalama River Type-N Coho Program, Kalama River Type-S Coho Program, Washougal Hatchery Type-N Coho Program, Lewis River Type-N Coho Program, Lewis River Type-S Coho Program, Fish First Wild Coho Program, Fish First Type-N Coho Program, Syverson Project Type-N Coho Program, Eagle Creek National Fish Hatchery, Sandy Hatchery, and the Bonneville/Cascade/Oxbow complex coho hatchery programs.

Table 16 identifies populations within the LCR Coho salmon ESU, their abundances, and hatchery input.

Table 16. LCR Coho salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River/Region	Historical Abundance	2002 Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay and Big Creek	Unknown	4,473	91%
Grays River	Unknown	Unknown	Unknown
Elochoman River	Unknown	Unknown	Unknown
Clatskanie River	Unknown	229	60%
Mill, Germany, and Abernathy creeks	Unknown	Unknown	Unknown
Scappoose Rivers	Unknown	458	0%
Cispus River	Unknown	Unknown	Unknown
Tilton River	Unknown	Unknown	Unknown
Upper Cowlitz River	Unknown	Unknown	Unknown
Lower Cowlitz River	Unknown	Unknown	Unknown
North Fork Toutle River	Unknown	Unknown	Unknown
South Fork Toutle River	Unknown	Unknown	Unknown
Coweeman River	Unknown	Unknown	Unknown
Kalama River	Unknown	Unknown	Unknown
North Fork Lewis River	Unknown	Unknown	Unknown
East Fork Lewis River	Unknown	Unknown	Unknown
Upper Clackamas River	Unknown	1,001	12%
Lower Clackamas River	Unknown	2,402	78%
Salmon Creek	Unknown	Unknown	Unknown
Upper Sandy River	Unknown	310	0%
Lower Sandy River	Unknown	271	97%
Washougal River	Unknown	Unknown	Unknown
LCR gorge tributaries	Unknown	Unknown	Unknown
White Salmon	Unknown	Unknown	Unknown
Upper Columbia River gorge tributaries	Unknown	1,317	>65%
Hood River	Unknown	Unknown	Unknown
Total	**Unknown**	**10,461 (min)**	

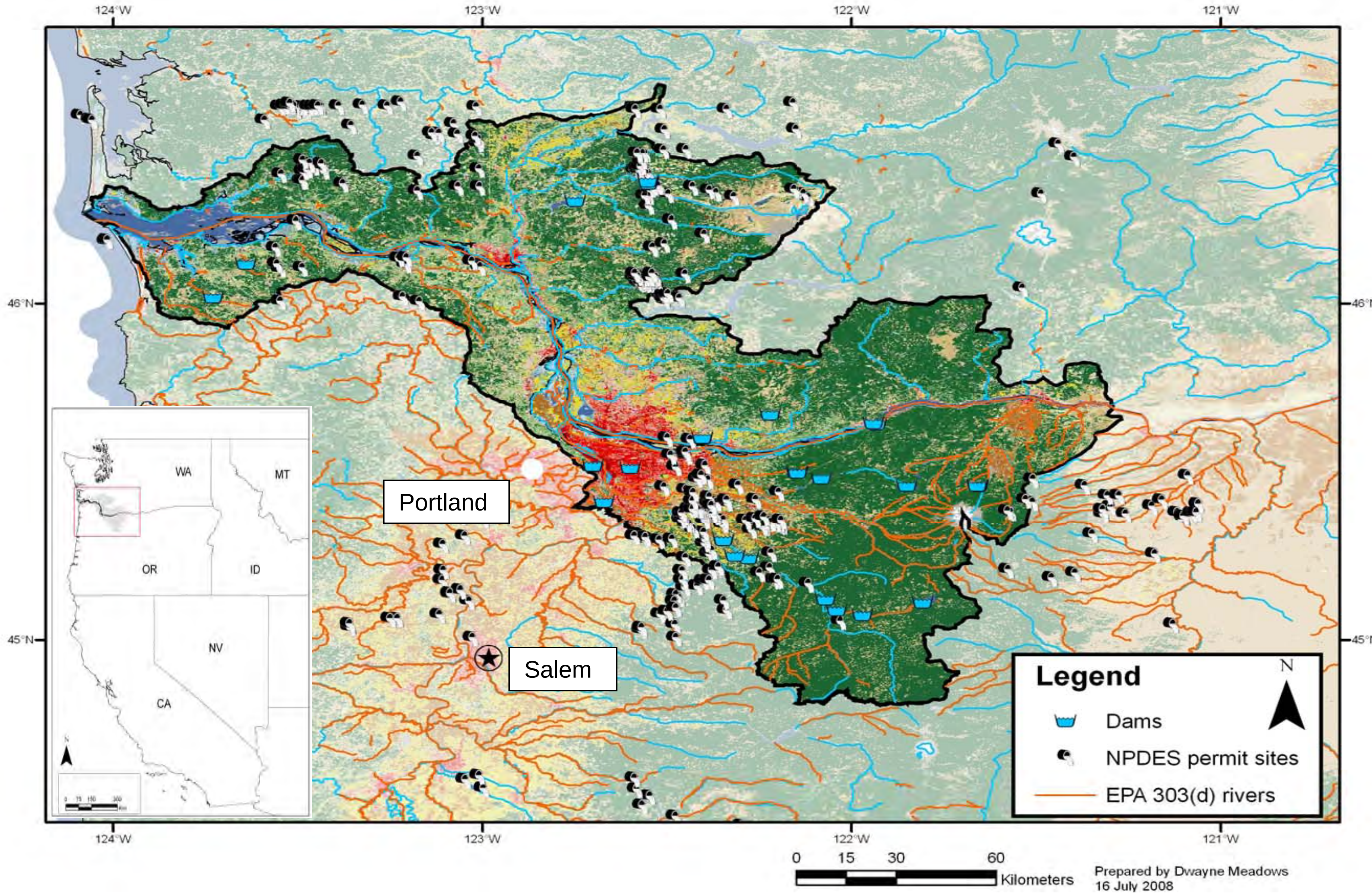


Figure 19 . LCR coho salmon distribution. Land Cover Class Legend in Figure 7.

Life History

Although run time variation is inherent to coho salmon life history, the ESU includes two distinct runs: early returning (Type S) and late returning (Type N). Type S coho salmon generally migrate south of the Columbia once they reach the ocean, returning to freshwater in mid-August and to the spawning tributaries in early September. Spawning peaks from mid-October to early November. Type N coho salmon have a northern distribution in the ocean, return to the Columbia River from late September through December and enter the tributaries from October through January. Most Type N spawning occurs from November through January. However some spawning occurs in February and as late as March (LCFRB 2004). Almost all LCR ESU coho salmon females and most males spawn at three years of age.

Status and Trends

LCR coho salmon were listed as endangered on June 28, 2005 (70 FR 37160). The vast majority (over 90%) of the historic population in the LCR coho salmon ESU appear to be either extirpated or nearly so. The two populations with any significant natural production (Sandy and Clackamas) are at appreciable risk because of low abundance, declining trends, and failure to respond after a dramatic reduction in harvest. Most of the other populations are believed to have very little, if any, natural production.

The Sandy population had a recent mean abundance of 342 spawners and a very low fraction of hatchery-origin spawners. Trends in the Sandy are similar to the Clackamas. The long-term trends and growth rate estimates over the period 1977 to 2001 have been slightly positive and the short-term trends have been slightly negative. Other populations in this ESU are dominated by hatchery production. There is very little, if any, natural production in Oregon beyond the Clackamas and Sandy rivers. The Washington side of the ESU is also dominated by hatchery production. There are no populations with appreciable natural production. The most serious threat facing this ESU is the scarcity of naturally-produced spawners, with attendant risks associated with small population, loss of diversity, and fragmentation and isolation of the remaining naturally-produced fish. In the only two populations with significant natural production (Sandy and Clackamas), short- and long-term trends are negative and productivity (as gauged by pre-harvest recruits) is down sharply from recent (1980s) levels.

The Federal Columbia River Power System Opinion (FCRPS) (2008) describes this ESU as consisting of three MPGs. Each is comprised of three to 14 populations. In many cases, populations have low abundance and natural runs have been extensively replaced by hatchery production. Abundance estimates are available for only five populations and trend estimates for only two. Time series are not available for Washington coho populations. The 100-year risk of extinction was derived qualitatively, based on risk categories and criteria identified by the W/LCTRT in 2004. Most of the population of LCR had high or very high extinction risk probabilities. Spatial structure has been substantially reduced by the loss of access to the upper portions of some basins from tributary hydro development (*i.e.*, Condit Dam on the Big White Salmon River and Powerdale Dam on the Hood River). Finally, the diversity of populations in all three MPGs has been eroded by large hatchery influences and periodically, low effective population sizes. Nevertheless, the genetic legacy of the Lewis and Cowlitz River coho salmon populations is preserved in ongoing hatchery programs.

Critical Habitat

NMFS has not designated critical habitat for LCR coho salmon.

SONCC *Coho Salmon*

Distribution

SONCC coho salmon consists of all naturally spawning populations of coho salmon that reside below long-term, naturally impassible barriers in streams between Punta Gorda, California and Cape Blanco, Oregon (Figure 20).

This ESU also includes three artificial propagation programs: the Cole Rivers Hatchery (ODFW stock #52), Trinity River Hatchery, and Iron Gate Hatchery coho hatchery programs. The three major river systems supporting Southern Oregon / Northern Coastal California coast coho are the Rogue, Klamath (including the Trinity), and Eel rivers.

Life History

SONCC coho salmon enter rivers in September or October. River entry is much later south of the Klamath River Basin, occurring in November and December, in basins south of the Klamath River to the Mattole River, California. River entry occurs from mid-Decmeber to mid-February in rivers farther south. Because coho salmon enter rivers late and spawn late south of the Mattole River, they spend much less time in the river prior to spawning. Coho salmon adults spawn at age three, spending just over a year in freshwater and a year and a half in the ocean.

Status and Trends

SONCC coho salmon were listed as threatened on May 7, 1997 (62 FR 24588). This species retained its original classification when its status was reviewed on June 28, 2005 (70 FR 37160). The status of coho salmon coast wide, including the SONCC coho salmon ESU, was formally assessed in 1995 (Weitkamp, Wainwright et al. 1995). Two subsequent status review updates have been published by NMFS. One review update addressed all West Coast coho salmon ESUs (Busby, Wainwright et al. 1996). The second update specifically addressed the Oregon Coast and SONCC coho salmon ESUs (Gustafson, Wainwright et al. 1997). In the 1997 status update, estimates of natural population abundance were based on very limited information. New data on presence/absence in northern California streams that historically supported coho salmon were even more disturbing than earlier results. Data indicated that a smaller percentage of streams contained coho salmon compared to the percentage presence in an earlier study. However, it was unclear whether these new data represented actual trends in local extinctions, or were biased by sampling effort.

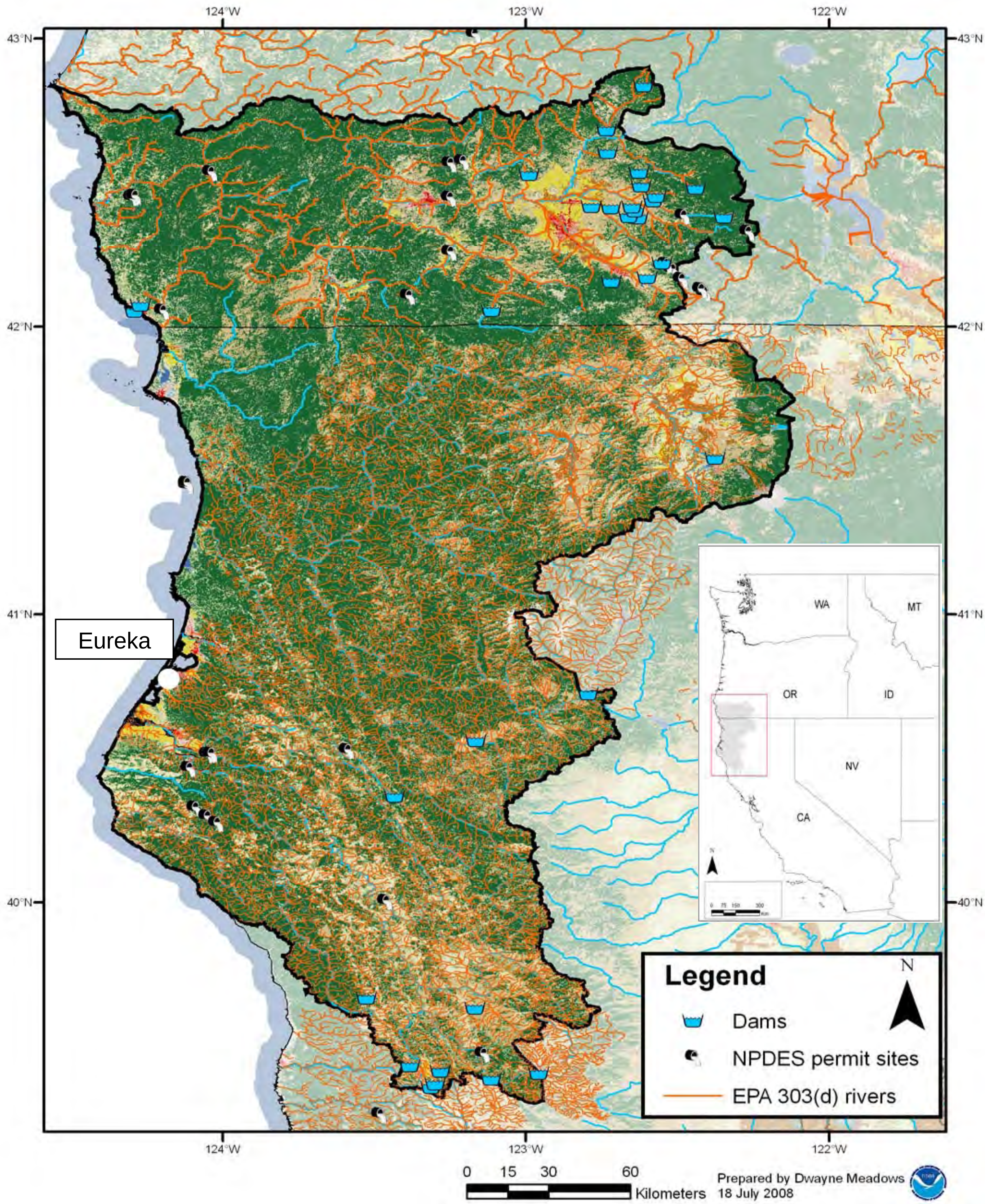


Figure 20. SONCC coho salmon distribution. Legend for Land Cover Class in Figure 7.

Data on population abundance and trends are limited for the California portion of this ESU. No regular estimates of natural spawner escapement are available. Historical point estimates of coho salmon abundance for the early 1960s and mid-1980s suggest that statewide coho spawning escapement in the 1940s ranged between 200,000 and 500,000 fish. Numbers declined to about 100,000 fish by the mid-1960s with about 43% originating from this ESU. Brown et al. (1994) estimated that the California portion of this ESU was represented by about 7,000 wild and naturalized coho salmon (Good, Waples et al. 2005). In the Klamath River, the estimated escapement has dropped from approximately 15,400 in the mid-1960s to about 3,000 in the mid-1980s, and more recently to about 2,000 (Good, Waples et al. 2005). The second largest producing river in this ESU, the Eel River, dropped from 14,000, to 4,000 to about 2,000 during the same period. Historical estimates are considered "best guesses" made using a combination of limited catch statistics, hatchery records, and the personal observations of biologists and managers.

Most recently, Williams et al. (2006) described the structure of historic populations of SONCC coho salmon. They described three categories of populations: functionally independent populations, potentially independent populations, and dependent populations. Functionally independent populations are populations capable of existing in isolation with a minimal risk of extinction. Potentially independent populations are similar but rely on some interchange with adjacent populations to maintain a low probability of extinction. Dependent populations have a high risk of extinction in isolation over a 100-year timeframe and rely on exchange of individuals from adjacent populations to maintain themselves.

Critical Habitat

Critical habitat was designated for the SONCC coho salmon on November 25, 1997, and re-designated on May 5, 1999. Species critical habitat encompasses all accessible river reaches between Cape Blanco, Oregon, and Punta Gorda, California and consists of the water, substrate, and river reaches (including off-channel habitats) in specified areas. Accessible reaches are those within the historical range of the ESU that can still be occupied by any life stage of coho salmon. Of 155 historical streams for which data are available, 63% likely still support coho salmon. Limiting factors identified for this species include: (1) Loss of channel complexity, connectivity and sinuosity, (2) loss of floodplain and estuarine habitats, (3) loss of riparian habitats and large in-river wood, (4) reduced streamflow, (5) poor water quality, temperature and excessive sedimentation, and (6) unscreened diversions and fish passage structures.

Oregon Coast Coho Salmon

Distribution

The Oregon Coast (OC) coho salmon ESU includes all naturally spawned populations of coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco (63 FR 42587; August 10, 1998; Figure 21). One hatchery stock, the Cow Creek (ODFW stock # 37) hatchery coho, is considered part of the ESU. Table 17 identifies populations within the OC coho salmon ESU, their abundances, and hatchery input.

Table 17. Oregon Coast Coho salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Basin	Historical Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
Necanicum	Unknown	1,889	35-40%
Nehalem	Unknown	18,741	40-75%
Tillamook	Unknown	3,949	30-35%
Nestucca	Unknown	3,846	~5%
Siletz	Unknown	2,295	~50%
Yaquima	Unknown	3,665	~25%
Alsea	Unknown	3,621	~40%
Siuslaw	Unknown	16,213	~40%
Umpqua	Unknown	24,351	<10%
Coos	Unknown	20,136	<5%
Coquille	Unknown	8,847	<5%
Total	**924,000**	**107,553**	

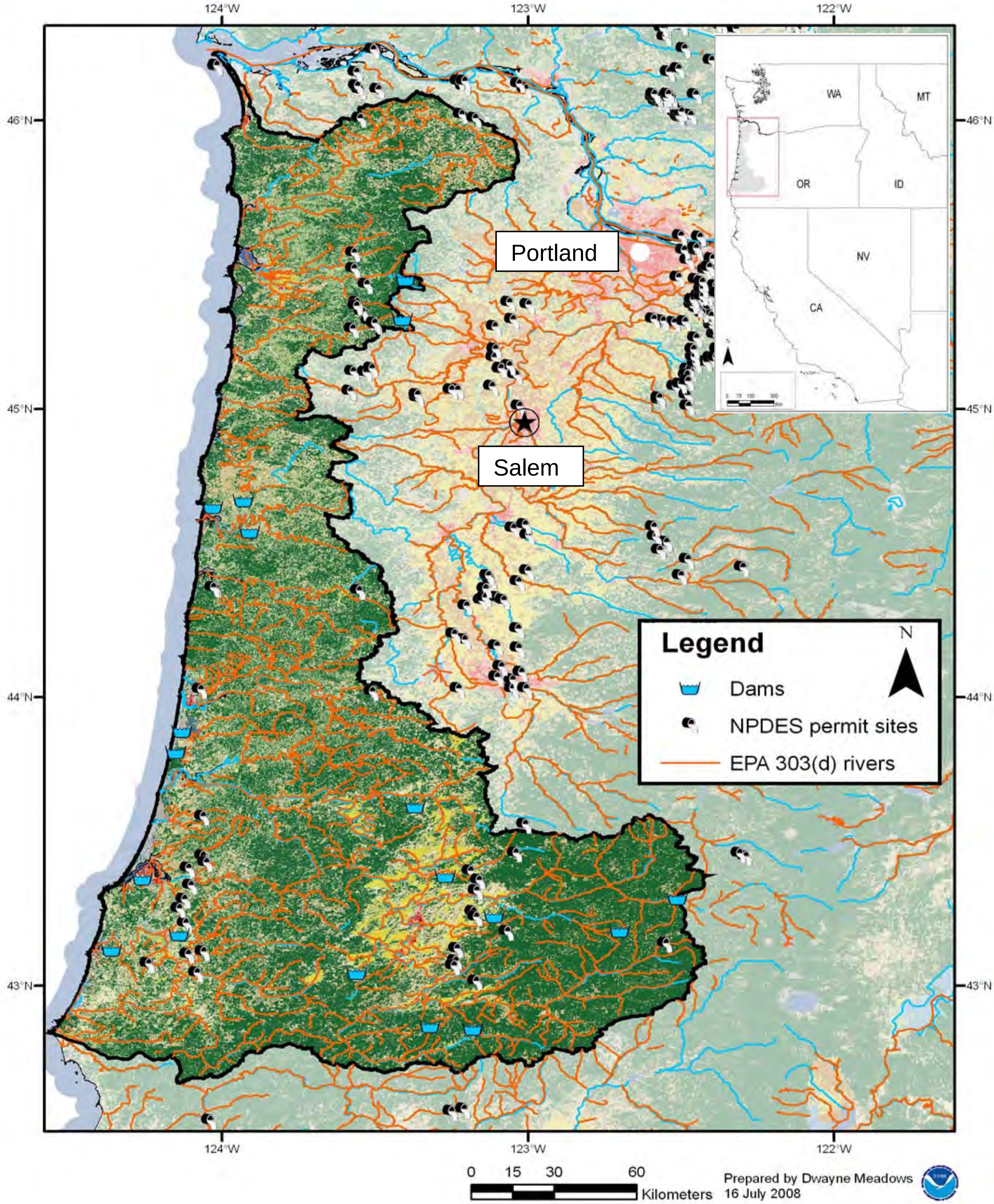


Figure 21. Oregon Coast Coho salmon distribution. Land Cover Class Legend in Figure 7.

Status and Trends

The OC coho salmon ESU was listed as a threatened species on February 11, 2008 (73 FR 7816). The most recent NMFS status review for the OC coho salmon ESU was conducted by the BRT in 2003, which assessed data through 2002. The abundance and productivity of OC coho salmon since the previous status review (Gustafson, Wainwright et al. 1997) represented some of the best and worst years on record. Yearly adult returns for this ESU were in excess of 160,000 natural spawners in 2001 and 2002, far exceeding the abundance observed for the past several decades. These encouraging increases in spawner abundance in 2000–2002 were preceded, however, by three consecutive brood years (the 1994–1996 brood years returning in 1997–1999, respectively) exhibiting recruitment failure. Recruitment failure is when a given year class of natural spawners fails to replace itself when its offspring return to the spawning grounds three years later. These three years of recruitment failure were the only such instances observed thus far in the entire 55-year abundance time series for OC coho salmon (although comprehensive population-level survey data have only been available since 1980). The encouraging 2000–2002 increases in natural spawner abundance occurred in many populations in the northern portion of the ESU, which were the most depressed at the time of the last review (Gustafson, Wainwright et al. 1997). Although encouraged by the increase in spawner abundance in 2000–2002, the BRT noted that the long-term trends in ESU productivity were still negative due to the low abundances observed during the 1990s (73 FR 7816). Since the BRT convened, the total abundance of natural spawners in the OC coho salmon ESU has declined each year (*i.e.*, 2003-2006). The abundance of total natural spawners in 2006 (111,025 spawners) was approximately 43% of the recent peak abundance in 2002 (255,372 spawners). In 2003, ESU-level productivity (evaluated in terms of the number of spawning recruits resulting from spawners three years earlier) was above replacement, and in 2004, productivity was approximately at replacement level. However, productivity was below replacement in 2005 and 2006, and dropped to the lowest level since 1991 in 2006.

Preliminary spawner survey data for 2007 (the average peak number of spawners per mile observed during random coho spawning surveys in 41 streams) suggest that the 2007-2008 return of Oregon Coast coho salmon is either: (1) much reduced from abundance levels in 2006, or (2) exhibiting delayed run timing from previous years. As of December 13, 2007, the average peak number of spawners per mile was below 2006 levels in 38 of 41 surveyed streams (ODFW 2007 *in* 73 FR 7816). It is possible that the timing of peak spawner abundance is delayed relative to previous years, and that increased spawner abundance in late December and January 2008 will compensate for the low levels observed thus far.

The recent five year geometric mean abundance (2002-2006) of approximately 152, 960 total natural spawners remains well above that of a decade ago (approximately 52,845 from 1992-1996). However, the decline in productivity from 2003 to 2006, despite generally favorable marine survival conditions and low harvest rates, is of concern. (73 FR 7816). The long-term trends in productivity in this ESU remain strongly negative.

Critical Habitat

Critical habitat was proposed for Oregon Coast coho salmon on December 14, 2004 (69 FR 74578). The final designation of critical habitat is included in the final rule published on February 11, 2008 (73 FR 7816). Approximately 6,568 stream miles (10,570 km) and 15 square miles (38.8 sq km) of lake habitat are designated critical habitat. Refer to the final rule for a detailed description of the watersheds included in the critical habitat, and a map for each subbasin.

Sockeye Salmon

Description of the Species

Sockeye salmon occur in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Klamath River in California and northern Hokkaido in Japan, to as far north as far as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia. We discuss the distribution, life history diversity, status, and critical habitat of the two endangered and threatened sockeye species separately.

The species exhibits riverine and lake life history strategies, the latter of which may be either freshwater resident forms or anadromous forms. The vast majority of sockeye salmon spawn in outlet streams of lakes or in the lakes themselves. These “lake-type” sockeye use the lake environment for rearing for up to three years and then migrate to sea, returning to their natal lake to spawn after one to four years at sea. Some sockeye spawn in rivers, however, without lake habitat for juvenile rearing. Offspring of these riverine spawners tend to use the lower velocity sections of rivers as the juvenile rearing environment for one to two years, or may migrate to sea in their first year.

Certain populations of *O. nerka* become resident in the lake environment over long periods of time and are called kokanee or little redfish (Burgner 1991). Kokanee and sockeye often co-occur in many interior lakes, where access to the sea is possible but energetically costly. On the other hand, coastal lakes where the migration to sea is relatively short and energetic costs are minimal, rarely support kokanee populations.

Spawning generally occurs in late summer and autumn, but the precise time can vary greatly among populations. Males often arrive earlier than females on the spawning grounds, and will persist longer during the spawning period. Average fecundity ranges from about 2,000 to 2,400 eggs per female to 5,000 eggs, depending upon the population and average age of the female. Fecundity in kokanee is much lower and may range from about 300 to less than 2,000 eggs.

Incubation is a function of water temperatures, but generally lasts between 100 and roughly 200 days (Burgner 1991). After emergence, fry move rapidly downstream or upstream along the banks to the lake rearing area. Fry emerging from lakeshore or island spawning grounds may simply move along the shoreline of the lake (Burgner 1991). Juvenile salmonids rely on a variety of non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle. Examples of off-channel habitat include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, and braids (Anderson 1999; Swift III 1979).

Sockeye salmon survive only in aquatic ecosystems and depend on the quantity and quality of those aquatic systems. Sockeye salmon, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the hydrogeomorphology of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems. These activities and features introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Ozette Lake Sockeye Salmon

Distribution

This ESU includes all naturally spawned populations of sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake, Washington. This ESU is composed of one historical population, with substantial substructuring of individuals into multiple spawning aggregations (Figure 22). The primary spawning aggregations occur in two beach locations – Allen's and Olsen's beaches, and in two tributaries Umbrella Creek and Big River (both tributary-spawning groups were initiated through a hatchery introduction program).

Sockeye salmon stock reared at the Makah Tribe's Umbrella Creek Hatchery were considered part of the ESU, but were not considered essential for recovery of the ESU. NMFS determined that it is presently not necessary to consider the progeny of intentional hatchery-wild or wild-wild crosses produced through the Makah Tribal hatchery program as listed under the ESA (March 25, 1999, 64 FR 14528). However, once the hatchery fish return and spawn in the wild, their progeny are considered listed.

Life History

The sockeye salmon life history is one of the most complex of any Pacific salmon species because of its variable freshwater residency (one to three years in freshwater), and because the species has several different forms: fish that go to the ocean and back, fish that remain in freshwater, and fish that do both.

Adult Ozette Lake sockeye salmon enter Ozette Lake through the Ozette River from April to early August. Adults remain in the lake for an extended period of time (return April – August; spawn late October-February) before spawning on beaches or in the tributaries. Sockeye salmon spawn primarily in lakeshore upwelling areas in Ozette Lake (at Allen's Bay and Olsen's Beach). Minor spawning may occur below Ozette Lake in the Ozette River or in Coal Creek, a tributary of the Ozette River. Sockeye salmon do not presently spawn in tributary streams to Ozette Lake. However, they may have spawned there historically. Eggs and alevins remain in gravel redds until the fish emerge as fry in spring. Fry then migrate immediately to the limnetic zone in Ozette Lake, where the fish rear. After one year of rearing, in late spring, Ozette Lake sockeye salmon emigrate seaward as one + smolts. The majority of Ozette Lake sockeye salmon return to spawn as four year old adult fish, having spent one winter in fresh water and two winters at sea (NMFS 2005b). As prespawning mortality is unknown, it is unclear what escapement levels to the spawning aggregations may be.

In Ozette Lake, naturally high water temperatures and low summer flows in the Ozette River may affect migration by altering timing of the runs (La Riviere 1991). Declines in abundance have been attributed to a combination of introduced species, predation, loss of tributary populations, decline in quality of beach spawning habitat, temporarily unfavorable ocean conditions, habitat degradation, and excessive historical harvests (Jacobs, Larson et al. 1996)

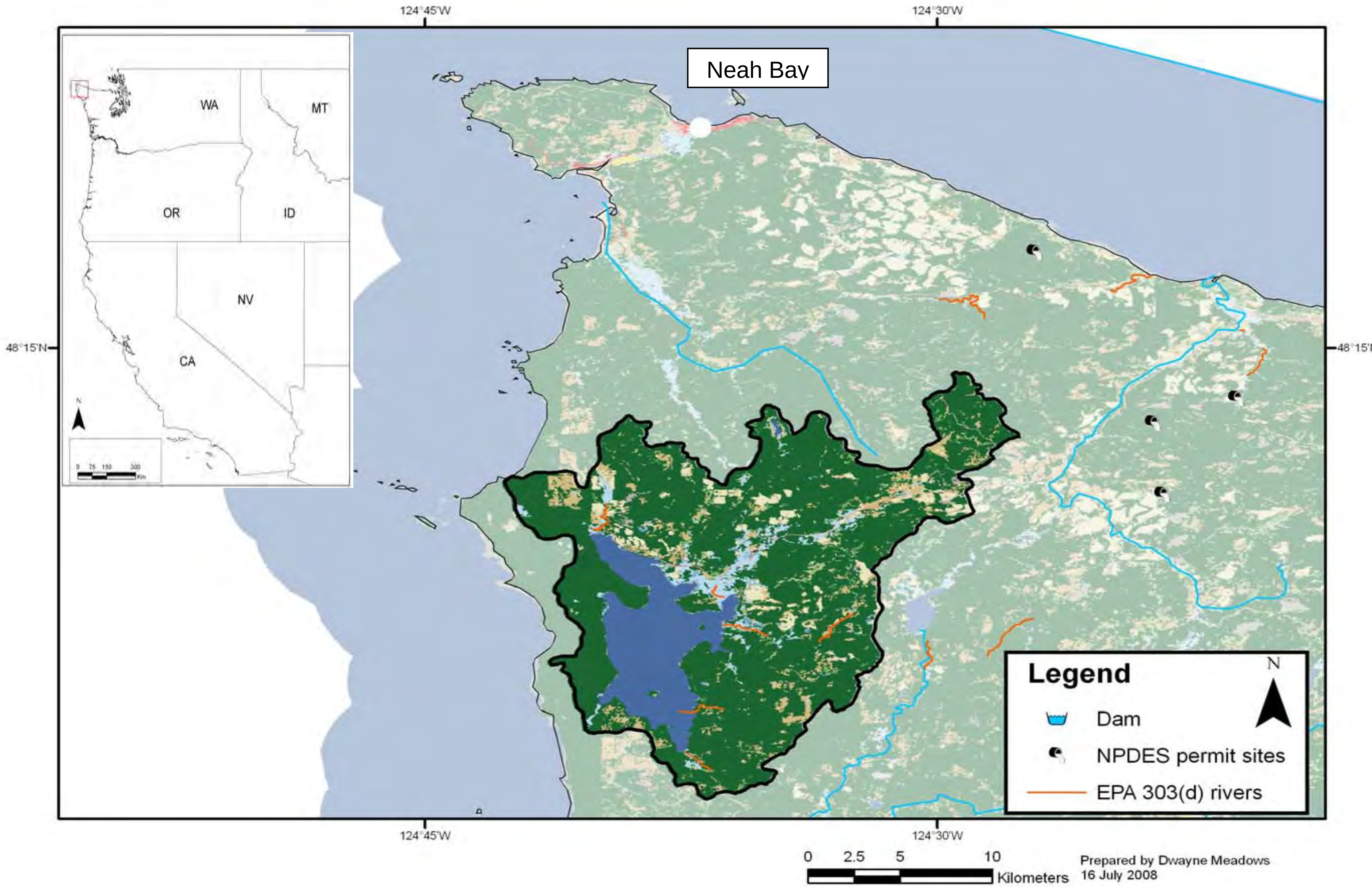


Figure 22. Ozette Lake Sockeye salmon distribution. Land Cover Class Legend in Figure 7.

Status and Trends

The Ozette Lake sockeye salmon ESU was originally listed as a threatened species in 1999 (64 FR 14528). This classification was retained following a species status review on June 28, 2005 (70 FR 37160).

The historical abundance of Ozette Lake sockeye salmon is poorly documented, but may have been as high as 50,000 individuals (Blum 1988). Nevertheless, the overall abundance of naturally–produced Ozette Lake sockeye salmon is believed to have declined substantially from historical levels. In the first study of lake escapement of Ozette Lake sockeye salmon (Kemmerich 1945), the run size entering the lake was estimated at a level of several thousand fish. These counts appear to be roughly double the current mean lake abundance, considering that they were likely conducted upstream from fisheries in or near to the Ozette River. Makah Fisheries Management (2000) concluded that there appears to be a substantial decline in the Tribal catch of Ozette Lake sockeye salmon beginning in the 1950s and a similar decline in the run size since the 1920s weir counts reported by Kemmerich (1945).

An updated NMFS analysis of total annual Ozette Lake sockeye salmon abundance (based on adult run size data presented in Jacobs et al. (1996)) indicates a trend in abundance averaging minus 2% per year over the period 1977 through 1998. The current tributary-based hatchery program was planned and initiated in response to the declining population trend identified for the Ozette Lake sockeye salmon population. The updated analysis also indicated that the most recent ten year (1989-98) trend for the population is plus 2% per year, improving from the minus 9.9% annual trend reported in Gustafson et al. (1999).

Data from the early 1900s indicate the spawning population was as large as 10,000 to 20,000 fish in large run years. Recent information on abundance of Ozette Lake sockeye salmon ESU comes from visual counts at a weir across the lake outlet. Therefore, the counts represent total run size. The estimates of total run size were revised upward after the 1997 status review due to resampling of data using new video counting technology. The Makah Fisheries biologists estimate that previous counts of adult sockeye salmon returning to the lake were underestimates, and they have attempted to correct run-size estimates based on their assessments of human error and variations in interannual run timing (Makah Fisheries Management 2000) *in* (Good, Waples et al. 2005).

The most recent (1996-2003) run-size estimates range from a low of 1,609 in 1997 to a high of 5,075 in 2003, averaging approximately 3,600 sockeye per year (Hard, Jones et al. 1992; Haggerty, Ritchie et al. 2007). For return years 2000 to 2003, the four-year average abundance estimate was slightly over 4,600 sockeye (Haggerty, Ritchie et al. 2007). Because run-size estimates before 1998 are likely to be even more unreliable than recent counts, and new counting technology has resulted in an increase in estimated run sizes, no statistical estimation of trends is reported. The current trends in abundance are unknown for the beach spawning aggregations. Although overall abundance appears to have declined from historical levels, whether this resulted in fewer spawning aggregations, lower abundances at each aggregation, or both, is unknown (Good, Waples et al. 2005). It is estimated that between 35,500 and 121,000 spawners could be normally carried after full recovery (Hard, Jones et al. 1992).

There has been no harvest of Ozette Lake sockeye salmon for the past four brood cycle years (since 1982). Prior to that time, ceremonial and subsistence harvests by the Makah Tribe were low, ranging from 0 to 84 fish per year. Harvest has not been an important mortality factor for the population in over 35 years. In addition, due to the early river entry timing of returning Ozette Lake sockeye salmon (beginning in late April, with the peak returns prior to late-May to mid-June), the fish are not intercepted in Canadian and U.S. marine area fisheries directed at Fraser River sockeye salmon. There are currently no known marine area harvest impacts on Ozette Lake sockeye salmon.

According to Good et al. (2005) it appears that overall abundance is low for this population, which represents an entire ESU, and may be substantially below historical levels. The number of returning adults in the last few years has increased. However, a substantial (but uncertain) fraction of these appear to be of hatchery origin. This condition leads to uncertainty regarding growth rate and productivity of the natural component of the ESU. Genetic integrity may have been compromised due to the artificial supplementation that has occurred in this population. Approximately one million sockeye have been released into the Ozette watershed from the late 1930s to present (Kemmerich 1945; Boomer 1995; Good, Waples et al. 2005).

Critical Habitat

On September 2, 2005, NMFS designated critical habitat for the Ozette Lake sockeye salmon ESU (70 FR 52630), and encompasses areas within the Hoh/Quillayute subbasin. Refer to the final rule for additional information on the watersheds within this subbasin, including a map of the area. Limiting factors for this species include siltation of beach-spawning habitat and logging.

Snake River Sockeye Salmon

Distribution

The SR sockeye salmon ESU includes all anadromous and residual sockeye from the SR basin Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake Captive Broodstock Program (Figure 23).

Life History

SR sockeye salmon are unique compared to other sockeye salmon populations. Sockeye salmon returning to Redfish Lake in Idaho's Stanley Basin travel a greater distance from the sea (approximately 900 miles) to a higher elevation (6,500 ft) than any other sockeye salmon population and are the southern-most population of sockeye salmon in the world (Bjornn, Craddock et al. 1968). Stanley Basin sockeye salmon are separated by 700 or more river miles from two other extant upper Columbia River populations in the Wenatchee River and Okanogan River drainages. These latter populations return to lakes at substantially lower elevations (Wenatchee at 1,870 ft, Okanagon at 912 ft) and occupy different ecoregions.

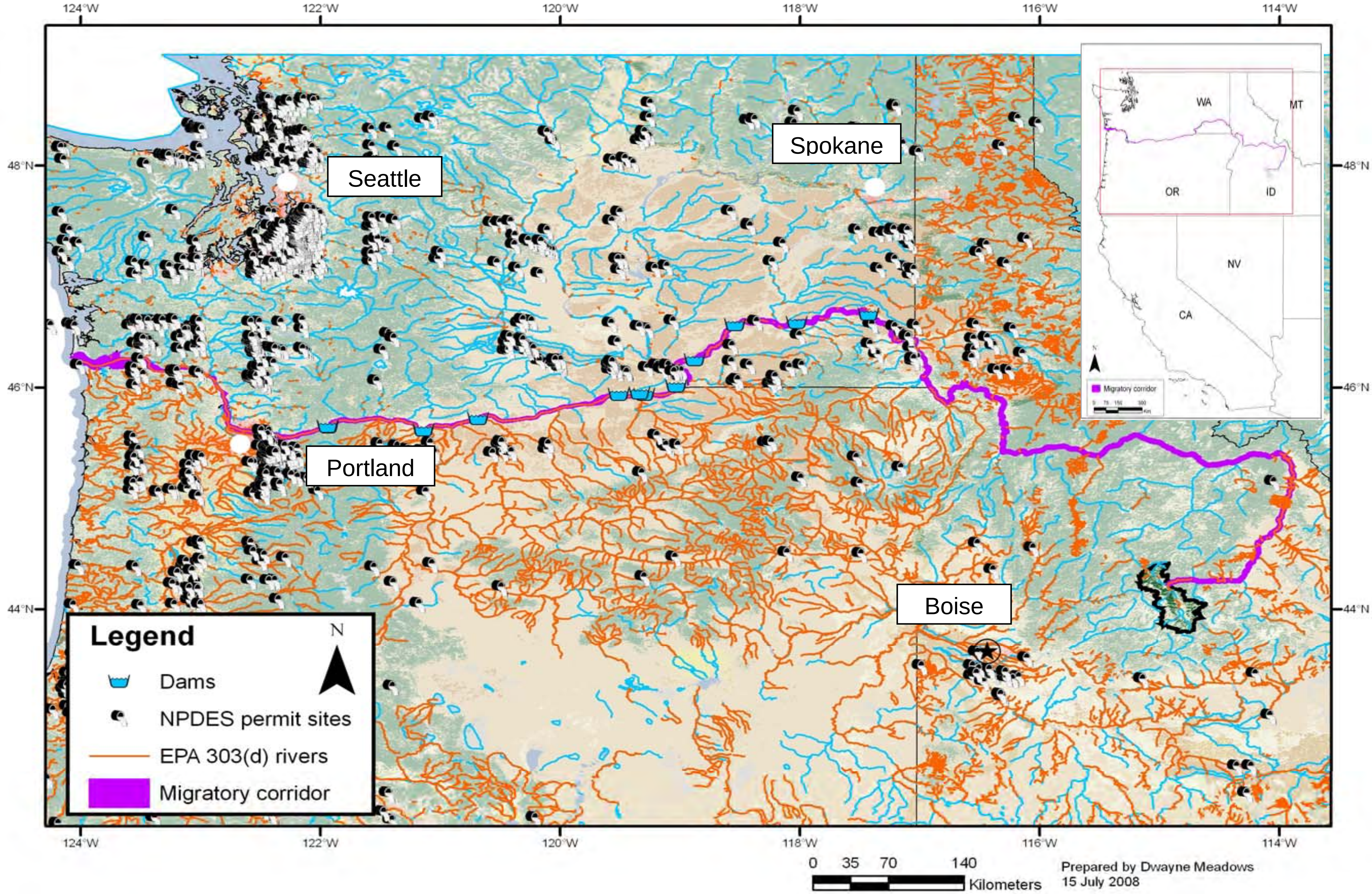


Figure 23. SR Sockeye Salmon distribution. Land Cover Class Legend in Figure 7.

Status and Trends

SR sockeye salmon were originally listed as endangered in 1991. Their classification was retained following a status review on June 28, 2005 (70 FR 37160). The only extant sockeye salmon population in the SR basin at the time of listing was that in Redfish Lake, in the Stanley Basin (upper Salmon River drainage) of Idaho. Other lakes in the SR basin historically supported sockeye salmon populations, including Wallowa Lake (Grande Ronde River drainage, Oregon), Payette Lake (Payette River drainage, Idaho) and Warm Lake (South Fork Salmon River drainage, Idaho) (Gustafson, Wainwright et al. 1997). These populations are now considered extinct. Although kokanee, a resident form of *O. nerka*, occur in numerous lakes in the SR basin, resident *O. nerka* were not considered part of the species at the time of listing in 1991. Subsequent to the 1991 listing, a residual form of sockeye residing in Redfish Lake was identified. The residuals are non-anadromous. They complete their entire life cycle in freshwater, but spawn at the same time and in the same location as anadromous sockeye salmon. In 1993, NMFS determined that residual sockeye salmon in Redfish Lake were part of the SR sockeye salmon. Also, artificially propagated sockeye salmon from the Redfish Lake Captive Propagation program are considered part of this species (June 28, 2005, 70 FR 37160).

NMFS has determined that this artificially propagated stock is genetically no more than moderately divergent from the natural population (Good, Waples et al. 2005). Five lakes in the Stanley Basin historically contained sockeye salmon: Alturas, Pettit, Redfish, Stanley and Yellowbelly (Bjornn, Craddock et al. 1968). It is generally believed that adults were prevented from returning to the Sawtooth Valley from 1910 to 1934 by Sunbeam Dam. Sunbeam Dam was constructed on the Salmon River approximately 20 miles downstream of Redfish Lake. Whether or not Sunbeam Dam was a complete barrier to adult migration remains unknown. It has been hypothesized that some passage occurred while the dam was in place, allowing the Stanley Basin population or populations to persist (Bjornn, Craddock et al. 1968; Matthews and Waples 1991).

Adult returns to Redfish Lake during the period 1954 through 1966 ranged from 11 to 4,361 fish (Bjornn, Craddock et al. 1968). Sockeye salmon in Alturas Lake were extirpated in the early 1900s as a result of irrigation diversions, although residual sockeye may still exist in the lake (Chapman and Witty 1993). From 1955 to 1965, the Idaho Department of Fish and Game eradicated sockeye salmon from Pettit, Stanley, and Yellowbelly lakes, and built permanent structures on each of the lake outlets that prevented re-entry of anadromous sockeye salmon (Chapman and Witty 1993). In 1985, 1986, and 1987, 11, 29, and 16 sockeye, respectively, were counted at the Redfish Lake weir (Good, Waples et al. 2005). Only 18 natural origin sockeye salmon have returned to the Stanley Basin since 1987. The first adult returns from the captive brood stock program returned to the Stanley Basin in 1999. From 1999 through 2005, a total of 345 captive brood program adults that had migrated to the ocean returned to the Stanley Basin.

Recent annual abundances of natural origin sockeye salmon in the Stanley Basin have been extremely low. No natural origin anadromous adults have returned since 1998 and the abundance of residual sockeye salmon in Redfish Lake is unknown. This species is entirely supported by adults produced through the captive propagation program at the present time. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely greater than 0.3% (Hebdon, Kline et al. 2004). Based on current abundance and productivity information, the SR sockeye salmon ESU does not meet the ESU-level viability criteria (non-negligible risk of extinction over a 100-year time period).

Critical Habitat

Critical habitat for these salmon was designated on December 28, 1993 (58 FR 68543). Designated habitats encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed SR salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 ft from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat areas include the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side), all river reaches from the estuary upstream to the confluence of the SR, and all SR reaches upstream to the confluence of the Salmon River; all Salmon River reaches to Alturas Lake Creek; Stanley, Redfish, yellow Belly, Pettit, and Alturas Lakes (including their inlet and outlet creeks); Alturas Lake Creek and that portion of Valley Creek between Stanley Lake Creek; and the Salmon River. Limiting factors identified for SR sockeye include: (1) Reduced tributary stream flow, (2) impaired tributary passage and blocks to migration, (3) degraded water quality; and (4) mainstem Columbia River hydropower system mortality.

Steelhead

Description of the Species

Steelhead are native to Pacific Coast streams extending from Alaska south to northwestern Mexico (Moyle 1976; Gustafson, Wainwright et al. 1997; Good, Waples et al. 2005). We discuss the distribution, life history diversity, status, and critical habitat of the 11 endangered and threatened steelhead species separately.

Steelhead can be divided into two basic run-types: the stream-maturing type, or summer steelhead and the ocean-maturing type, or winter steelhead. The stream-maturing type or summer steelhead enters fresh water in a sexually immature condition. It requires several months in freshwater to mature and spawn. The ocean-maturing type or winter steelhead enters freshwater with well-developed gonads and spawns shortly after river entry. Variations in migration timing exist between populations. Some river basins have both summer and winter steelhead, while others only have one run-type.

Summer steelhead enter freshwater between May and October in the Pacific Northwest (Nickelsen, Nicholas et al. 1992; Busby, Wainwright et al. 1996). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelsen, Nicholas et al. 1992). They migrate inland toward spawning areas, overwinter in the larger rivers, resume migration in early spring to natal streams, and then spawn (Meehan and Bjornn 1991; Nickelsen, Nicholas et al. 1992) in January and February (Barnhart 1986). Winter steelhead enter freshwater between November and April in the Pacific Northwest (Nickelsen, Nicholas et al. 1992; Busby, Wainwright et al. 1996), migrate to spawning areas, and then spawn, generally in April and May (Barnhart 1986). Some adults, however, do not enter some coastal streams until spring, just before spawning (Meehan and Bjornn 1991).

There is a high degree of overlap in spawn timing between populations regardless of run type (Busby, Wainwright et al. 1996). Difficult field conditions at that time of year and the remoteness of spawning grounds contribute to the relative lack of specific information on steelhead spawning. Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby, Wainwright et al. 1996), although steelhead rarely spawn more than twice before dying; most that do so are females (Nickelsen, Nicholas et al. 1992). Iteroparity is more common among southern steelhead populations than northern populations (Busby, Wainwright et al. 1996).

After two to three weeks, in late spring, and following yolk sac absorption, alevins emerge from the gravel and begin actively feeding. After emerging from the gravel, fry usually inhabit shallow water along banks of perennial streams. Fry occupy stream margins (Nickelsen, Nicholas et al. 1992). Summer rearing takes place primarily in the faster parts of pools, although young-of-the-year are abundant in glides and riffles. Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelsen, Nicholas et al. 1992).

Juvenile steelhead migrate little during their first summer and occupy a range of habitats featuring moderate to high water velocity and variable depths (Bisson, Sullivan et al. 1988). Juvenile steelhead feed on a wide variety of aquatic and terrestrial insects (Chapman and Bjornn 1969), and older juveniles sometimes prey on emerging fry. Steelhead hold territories close to the substratum where flows are lower and sometimes counter to the main stream; from these, they can make forays up into surface currents to take drifting food (Kalleberg 1958). Juveniles rear in freshwater from one to four years, then smolt and migrate to the ocean in March and April (Barnhart 1986). Winter steelhead juveniles generally smolt after two years in freshwater (Busby, Wainwright et al. 1996). Juvenile steelhead tend to migrate directly offshore during their first summer from whatever point they enter the ocean rather than migrating along the coastal belt as salmon do. During the fall and winter, juveniles move southward and eastward (Hartt and Dell 1986) *op. cit.* (Nickelsen, Nicholas et al. 1992). Steelhead typically reside in marine waters for two or three years prior to returning to their natal stream to spawn as four or five year olds. Juvenile salmonids rely on a variety of non-main channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle. Examples of off-channel habitat include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, and braids (Anderson 1999; Swift III 1979).

Status and Trends

Steelhead, like the other salmon discussed previously, survive only in aquatic ecosystems and, therefore, depend on the quantity and quality of those aquatic systems. Steelhead, like the other salmon NMFS has listed, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the hydrogeomorphology of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems. These same activities and features introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Central California Coast Steelhead

Distribution

The Central California Coast (CCC) steelhead DPS includes all naturally spawned anadromous *O. mykiss* (steelhead) populations below natural and manmade impassable barriers in California streams from the Russian River (inclusive) to Aptos Creek (inclusive), and the drainages of San Francisco, San Pablo, and Suisun Bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers (Figure 24). Tributary streams to Suisun Marsh including Suisun Creek, Green Valley Creek, and an unnamed tributary to Cordelia Slough (commonly referred to as Red Top Creek), excluding the Sacramento-San Joaquin River Basin, as well as two artificial propagation programs: the Don Clausen Fish Hatchery, and Kingfisher Flat Hatchery/ Scott Creek (Monterey Bay Salmon and Trout Project) steelhead hatchery programs. Table 18 identifies populations within the CCC Steelhead salmon ESU, their abundances, and hatchery input.

Table 18. CCC Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Basin	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Russian River	65,000 (1970)	1,750-7,000 (1994)	Unknown
Lagunitas	Unknown	400-500 (1990s)	Unknown
San Gregorio	1,000 (1973)	Unknown	Unknown
Waddell Creek	481	150 (1994)	Unknown
Scott Creek	Unknown	<100 (1991)	Unknown
San Vicente Creek	150 (1982)	50 (1994)	Unknown
San Lorenzo River	20,000	<150 (1994)	Unknown
Soquel Creek	500-800 (1982)	<100 (1991)	Unknown
Aptos Creek	200 (1982)	50-75 (1994)	Unknown
Total	**94,000**	**2,400-8,125**	

Life History

Only winter steelhead are found in this DPS and those to the south. Migration and spawn timing are similar to adjacent steelhead populations. There is little other life history information for steelhead in this DPS.

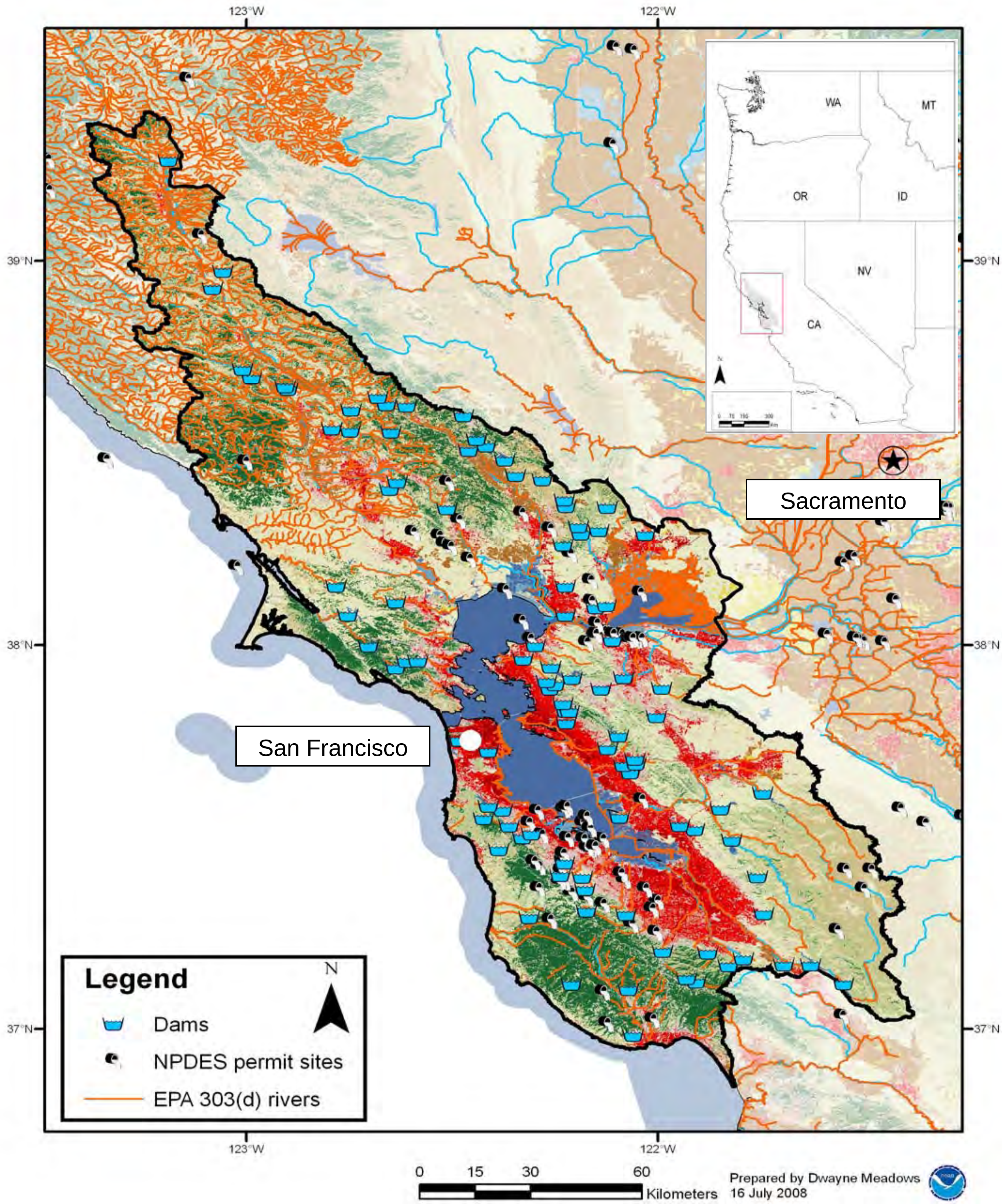


Figure 24. CCC steelhead. Land Cover Class Legend in Figure 7.

Status and Trends

The CCC steelhead DPS was listed as a threatened species on August 18, 1997(62 FR 43937). Its threatened status was reaffirmed on January 5, 2006 (71 FR 834). Busby et al. (1996) reported one estimate of historical (pre-1960s) abundance. Shapovalov and Taft (1954) described an average of about 500 adults in Waddell Creek (Santa Cruz County) for the 1930s and early 1940s. Johnson (1964) estimated a run size of 20,000 steelhead in the San Lorenzo River before 1965. The CDFG (1965) estimated an average run size of 94,000 steelhead for the entire DPS, for the period 1959–1963. The analysis by CDFG (1965) was compromised for many basins, as the data did not exist for the full 5-year analytical period. The authors of CDFG (1965) state that "estimates given here which are based on little or no data should be used only in outlining the major and critical factors of the resource."

Recent data for the Russian and San Lorenzo rivers (Reavis 1991; CDFG 1994; Shumann 1994) suggested that these basins had populations smaller than 15% of their size 30 years earlier. These two basins were thought to have originally contained the two largest steelhead populations in the CCC steelhead ESU.

A status review update in 1997 (Gustafson, Wainwright et al. 1997) concluded that slight increases in abundance occurred in the three years following the status review. However, the analyses on which these conclusions were based had various problems. They include the inability to distinguish hatchery and wild fish, unjustified expansion factors, and variance in sampling efficiency on the San Lorenzo River. Presence-absence data indicated that most (82%) sampled streams (a subset of all historical steelhead streams) had extant populations of juvenile *O. mykiss* (Adams 2000; Good, Waples et al. 2005).

The majority (69%) of BRT votes were for "likely to become endangered," and another 25% were for "in danger of extinction". Abundance and productivity were of relatively high concern (as a contributing factor to risk of extinction), and spatial structure was also of concern. Predation by pinnipeds at river mouths and during the ocean phase was noted as a recent development posing significant risk. There were no time-series data for the CCC steelhead DPS. A variety of evidence suggested the ESU's largest run (the Russian River winter steelhead run) has been, and continues to be, reduced in size. Concern was also expressed about populations in the southern part of the DPS's range—notably those in Santa Cruz County and the South Bay area (Good, Waples et al. 2005).

Critical Habitat

Critical habitat was designated for the CCC steelhead DPS on September 2, 2005 (70 FR 52488), and includes areas within the following hydrologic units: Russian River, Bodega, Marin Coastal, San Mateo, Bay Bridges, Santa Clara, San Pablo, Big Basin. Refer to the final rule for a more detailed description of critical habitat, including a map for each hydrologic unit.

California Central Valley Steelhead

Distribution

California CV steelhead occupy the Sacramento and San Joaquin Rivers and its tributaries (Figure 25).

Life History

California CV steelhead are considered winter steelhead by the CDFG, although "three distinct runs," including summer steelhead, may have occurred there as recently as 1947(CDFG 1995; McEwan and Jackson 1996). Steelhead within this DPS have the longest freshwater migration of any population of winter steelhead. There is essentially a single continuous run of steelhead in the upper Sacramento River. River entry ranges from July through May, with peaks in September and February. Spawning begins in late December and can extend into April (McEwan and Jackson 1996).

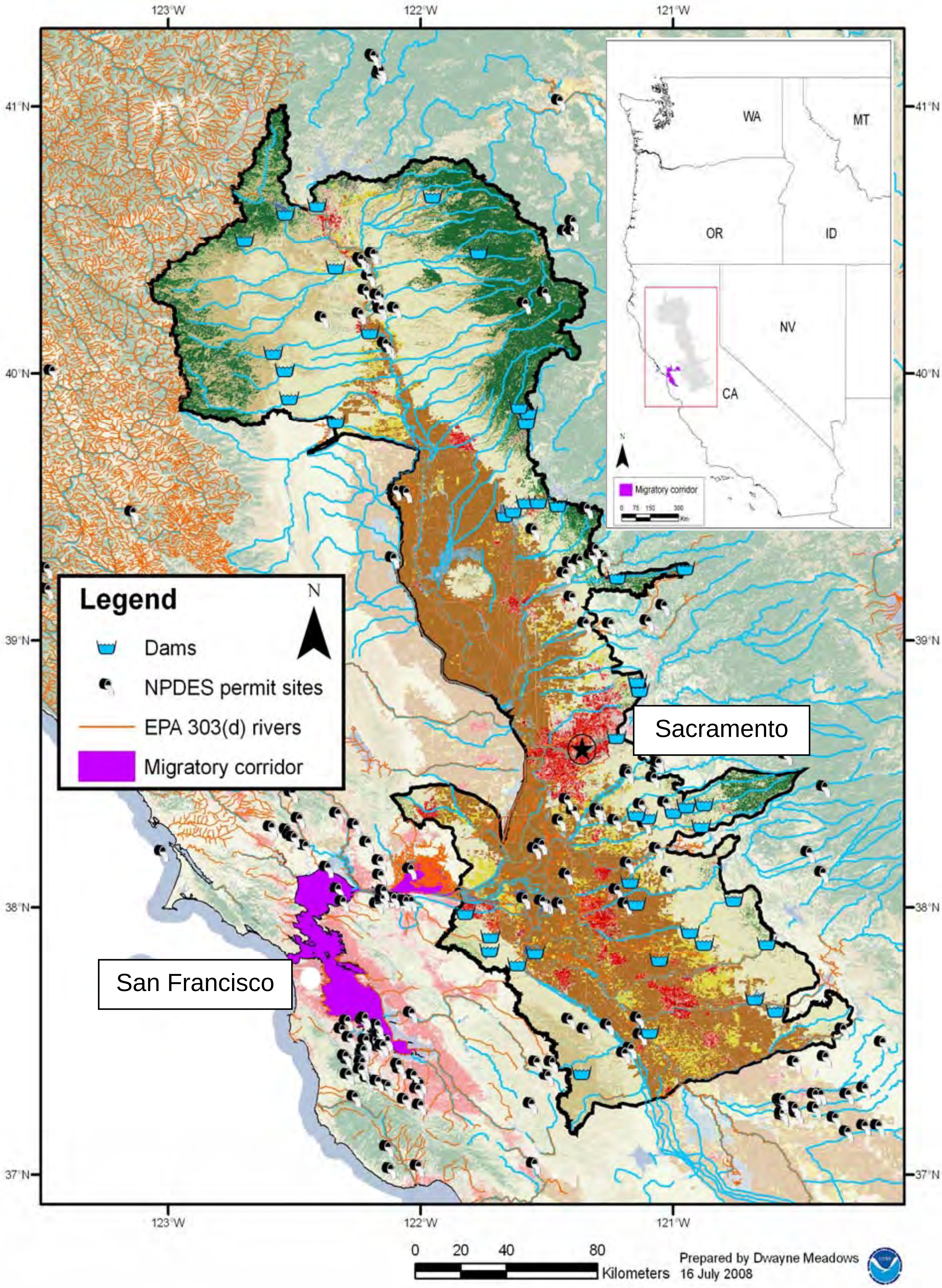


Figure 25. California CV steelhead distribution. Land Cover Class Legend in Figure 7.

Status and Trends

California CV steelhead were listed as threatened on March 19, 1998. Their classification was retained following a status review on January 5, 2006 (71 FR 834). This DPS consists of steelhead populations in the Sacramento and San Joaquin River (inclusive of and downstream of the Merced River) basins in California's CV. Steelhead historically were well distributed throughout the Sacramento and San Joaquin Rivers (Busby, Wainwright et al. 1996). Steelhead were found from the upper Sacramento and Pit River systems (now inaccessible due to Shasta and Keswick Dams), south to the Kings and possibly the Kern River systems (now inaccessible due to extensive alteration from water diversion projects), and in both east- and west-side Sacramento River tributaries (Yoshiyama, Gerstung et al. 1996). The present distribution has been greatly reduced (McEwan and Jackson 1996). The California Advisory Committee on Salmon and Steelhead (1988) reported a reduction of steelhead habitat from 6,000 miles historically to 300 miles today. Historically, steelhead probably ascended Clear Creek past the French Gulch area, but access to the upper basin was blocked by Whiskeytown Dam in 1964 (Yoshiyama, Gerstung et al. 1996). Steelhead also occurred in the upper drainages of the Feather, American, Yuba, and Stanislaus Rivers which are now inaccessible (McEwan and Jackson 1996; Yoshiyama, Gerstung et al. 1996).

Historic CV steelhead run size is difficult to estimate given limited data, but may have approached one to two million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock et al. (1961) estimated an average of 20,540 adult steelhead in the Sacramento River, upstream of the Feather River, through the 1960s. Steelhead counts at Red Bluff Diversion Dam declined from an average of 11,187 for the period of 1967 to 1977, to an average of approximately 2,000 through the early 1990s, with an estimated total annual run size for the entire Sacramento-San Joaquin system, based on Red Bluff Diversion Dam counts, to be no more than 10,000 adults (McEwan and Jackson 1996; McEwan 2001). Steelhead escapement surveys at Red Bluff Diversion Dam ended in 1993 due to changes in dam operations.

The only consistent data available on steelhead numbers in the San Joaquin River basin come from CDFG mid-water trawling samples collected on the lower San Joaquin River at Mossdale. These data indicate a decline in steelhead numbers in the early 1990s, which have remained low through 2002 (CDFG 2003). In 2004, a total of 12 steelhead smolts were collected at Mossdale (CDFG unpublished data).

Existing wild steelhead stocks in the CV are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks. A few wild steelhead are produced in the American and Feather Rivers (McEwan and Jackson 1996).

Snorkel surveys from 1999 to 2002 indicate that steelhead are present in Clear Creek (J. Newton, FWS, pers. comm. 2002, as reported *in* Good et al. (2005). Because of the large resident *O. mykiss* population in Clear Creek, steelhead spawner abundance has not been estimated.

Until recently, steelhead were thought to be extirpated from the San Joaquin River system. Recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, Calaveras, and other streams previously thought to be void of steelhead (McEwan 2001). On the Stanislaus River, steelhead smolts have been captured in rotary screw traps at Caswell State Park and Oakdale each year since 1995 (Demko and Cramer 2000). It is possible that naturally spawning populations exist in many other streams. However, these populations are undetected due to lack of monitoring programs (IEPSPWT 1999).

The majority (66%) of BRT votes was for "in danger of extinction," and the remainder was for "likely to become endangered". Abundance, productivity, and spatial structure were of highest concern. Diversity considerations were of significant concern. The BRT was concerned with what little new information was available and indicated that the monotonic decline in total abundance and in the proportion of wild fish in the California CV steelhead ESU was continuing.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005. The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. Specific sites include: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, and adequate forage.

Lower Columbia River Steelhead

Distribution

LCR steelhead DPS includes 23 historical anadromous populations in four MPGs. This DPS includes naturally-produced steelhead returning to Columbia River tributaries on the Washington side between the Cowlitz and Wind rivers in Washington and on the Oregon side between the Willamette and Hood rivers, inclusive (Figure 26). In the Willamette River, the upstream boundary of this species is at Willamette Falls. This species includes both winter and summer steelhead. Two hatchery populations are included in this species, the Cowlitz Trout Hatchery winter-run stock and the Clackamas River stock. However, neither hatchery population was listed as threatened.
Table 19 identifies populations within the LCR Steelhead salmon DPS, their abundances, and hatchery input.

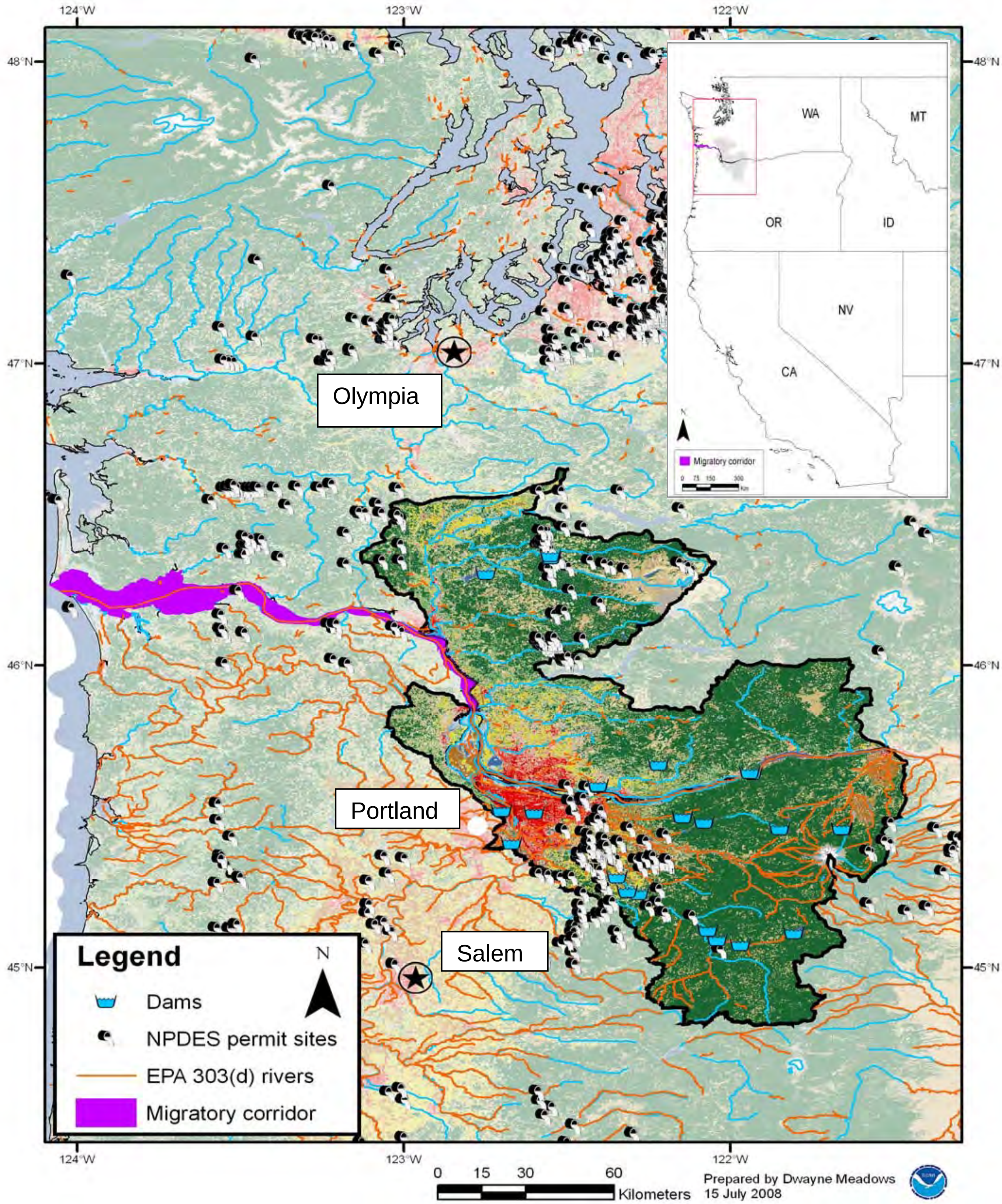


Figure 26. Lower Columbia River Steelhead distribution. Land Cover Class Legend in Figure 7.

Table 19. LCR Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Cispus River	Unknown	Unknown	Unknown
Tilton River	Unknown	2,787	~73%
Upper Cowlitz River	Unknown	Unknown	Unknown
Lower Cowlitz River	1,672	Unknown	Unknown
Coweeman River	2,243	466	~50%
South Fork Toutle River	2,627	504	~2%
North Fork Toutle River	3,770	196	0%
Kalama River-winter run	554	726	0%
Kalama River-summer run	3,165	474	~32%
North Fork Lewis River-winter run	713	Unknown	Unknown
North Fork Lewis River-summer run	Unknown	Unknown	Unknown
East Fork Lewis River-winter run	3,131	Unknown	Unknown
East Fork Lewis River-summer run	422	434	~25%
Salmon Creek	Unknown	Unknown	Unknown
Washougal River-winter run	2,497	323	0%
Washougal River-summer run	1,419	264	~8%
Clackamas River	Unknown	560	41%
Sandy River	Unknown	977	42%
Lower Columbia gorge tributaries	793	Unknown	Unknown
Upper Columbia gorge tributaries	243	Unknown	Unknown
Hood River-winter run	Unknown	756	~52%
Hood River-summer run	Unknown	931	~83%
Wind River	2,288	472	~5%
Total	**25,537 (min)**	**9,870 (min)**	

Life History

Summer steelhead return to freshwater from May to November, entering the Columbia River in a sexually immature condition and requiring several months in freshwater before spawning. Winter steelhead enter freshwater from November to April. They are close to sexual maturation and spawn shortly after arrival in their natal stream. Where both races spawn in the same stream, summer steelhead tend to spawn at higher elevations than the winter forms. Juveniles rear in freshwater (stream-type life history).

Status and Trends

LCR steelhead were listed as threatened on March 19, 1998 (63 FR 13347), and reaffirmed as threatened on January 5, 2006 (71 FR 834). The 1998 status review noted that this ESU is characterized by populations at low abundance relative to historical levels, significant population declines since the mid-1980s, and widespread occurrence of hatchery fish in naturally-spawning steelhead populations. During this review NMFS was unable to identify any natural populations that would be considered at low risk.

All populations declined from 1980 to 2000, with sharp declines beginning in 1995. Historical counts in some of the larger tributaries (Cowlitz, Kalama, and Sandy Rivers) suggest the population probably exceeded 20,000 fish. During the 1990s, fish abundance dropped to 1,000 to 2,000 fish. Recent abundance estimates of natural-origin spawners range from completely extirpated for some populations above impassable barriers to over 700 for the Kalama and Sandy winter-run populations. A number of the populations have a substantial fraction of hatchery-origin spawners in spawning areas. These populations are hypothesized to be sustained largely by hatchery production. Exceptions are the Kalama, the Toutle, and East Fork Lewis winter-run populations. These populations have relatively low recent mean abundance estimates with the largest being the Kalama (geometric mean of 728 spawners).

According to Good et al. (2005), most populations are at relatively low abundance. Those with adequate data for modeling are estimated to have a relatively high extinction probability. Some populations, particularly summer run, have shown higher return in the last two to three years. Many of the long-and short-term trends in abundance of individual populations are negative, some severely so. The trend in natural spawners is <1; indicating the population is not replacing itself and in decline. Spatial structure has been substantially reduced by the loss of access to the upper portions of some basins due to tributary hydro development. Finally, a number of the populations have a substantial fraction of hatchery-origin spawners. Exceptions are the Kalama, North and South Fork Toutle, and East Fork Lewis winter-run populations, which have few hatchery fish spawning in natural spawning areas.

Over 73% of the BRT votes for this species fell in the “likely to become endangered” category. There were small minorities falling in the “danger of extinction” and “not likely to become endangered” categories. The BRT found moderate risks in all VSP categories, with mean risk matrix scores ranging from moderately low for spatial structure to moderately high for abundance and productivity (population growth rate).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages. Specific sites include: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity.

Of 47 subbasins reviewed in NMFS' assessment of critical habitat for the LCR steelhead, 34 subbasins were rated as having a high conservation value. Eleven subbasins were rated as having a medium value and two were rated as having a low value to the conservation of the DPS. Limiting factors identified for LCR steelhead include: (1) Degraded floodplain and steam channel structure and function, (2) reduced access to spawning/rearing habitat, (3) altered streamflow in tributaries, (4) excessive sediment and elevated water temperatures in tributaries, and (5) hatchery impacts (NMFS 2005b). The above conditions also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Middle Columbia River Steelhead

Distribution

Middle Columbia River (MCR) steelhead DPS includes anadromous populations in Oregon and Washington subbasins upstream of the Hood and Wind River systems to and including the Yakima River (Figure 27). There are four MPGs with 17 populations in this DPS. Steelhead from the SR Basin (described elsewhere) are excluded. This species includes the only populations of inland winter steelhead in the U.S., in the Klickitat River and Fifteenmile Creek (Busby, Wainwright et al. 1996).

Two hatchery populations are considered part of this species, the Deschutes River stock and the Umatilla River stock. Listing for neither of these stocks was considered warranted. MCR steelhead occupy the intermontane region which includes some of the driest areas of the Pacific Northwest, generally receiving less than 15.7 inches of rainfall annually. Vegetation is of the shrub-steppe province, reflecting the dry climate and harsh temperature extremes. Because of this habitat, occupied by the species, factors contributing to the decline include agricultural practices, especially grazing, and water diversions and withdrawals. In addition, hydropower development has impacted the species by preventing these steelhead from migrating to habitat above dams, and by killing some of them when they try to migrate through the Columbia River hydroelectric system. Table 20 identifies populations within the MCR Steelhead salmon DPS, their abundances, and hatchery input.

Table 20. MCR Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Klickitat River	Unknown	97-261 reds	Unknown
Yakima River	Unknown	1,058-4,061	97%
Fifteenmile Creek	Unknown	2.87 rpm	100%
Deschutes River	Unknown	10,026-21,457	38%
John Day upper main stream	Unknown	926-4,168	96%
John Day lower main stream	Unknown	1.4 rpm	0%
John Day upper north fork	Unknown	2.57 rpm	0%
John Day lower north fork	Unknown	.52 rpm	0%
John Day middle fork	Unknown	3.7 rpm	0%
John Day south fork	Unknown	2.52 rpm	0%
Umatilla River	Unknown	1,480-5,157	60%
Touchet River	Unknown	273-527	84%
Total	Unknown		

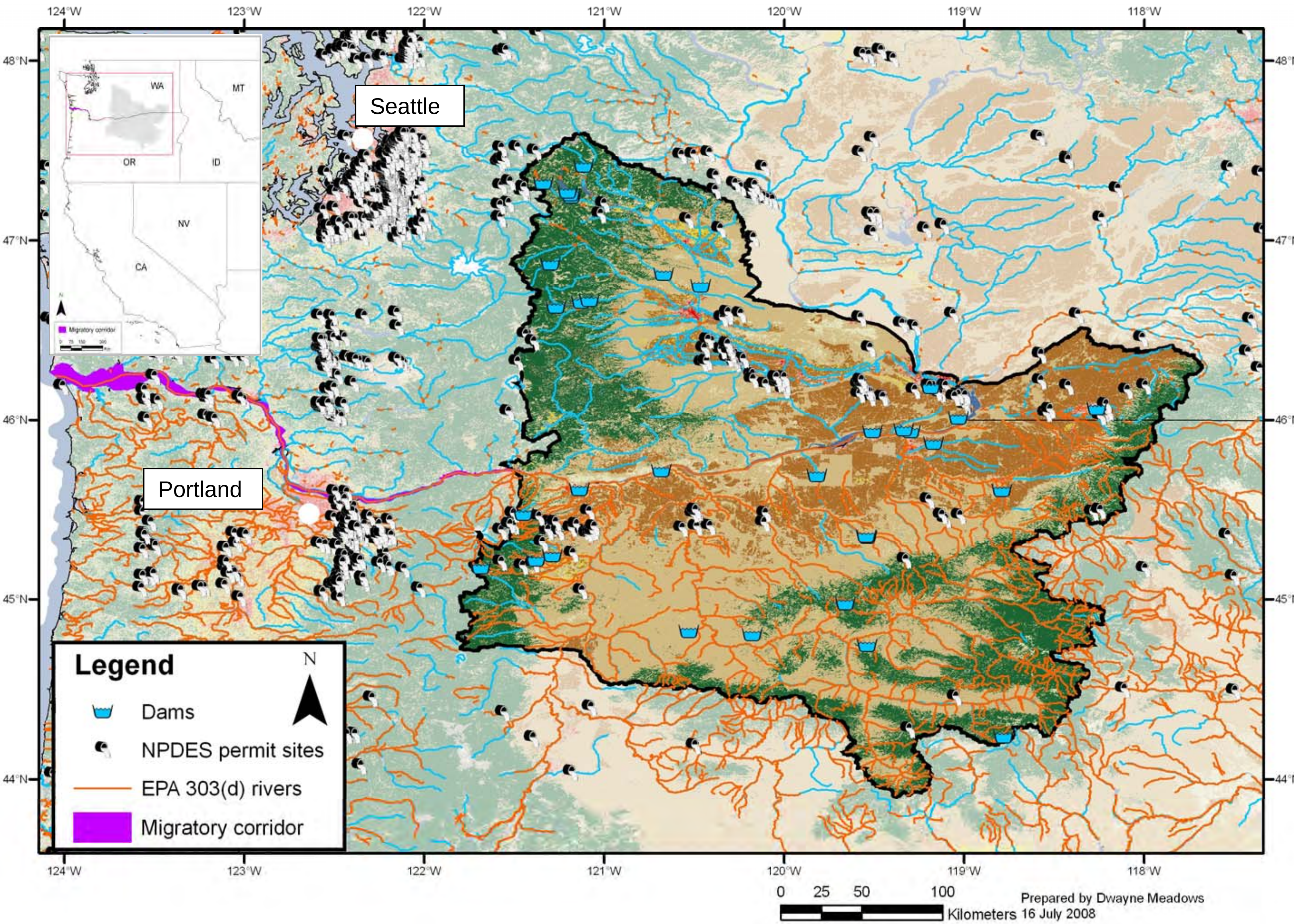


Figure 27. MCR Steelhead distribution. Land Cover Class Legend in Figure 7.

Life History

Most MCR steelhead smolt at two years and spend one to two years in saltwater prior to re-entering freshwater. Here they may remain up to a year prior to spawning (Howell, Jones et al. 1985). Within this ESU, the Klickitat River is unusual as it produces both summer and winter steelhead. The summer steelhead are dominated by age two ocean steelhead. Most other rivers in this region produce about equal numbers of both age one and two ocean steelhead.

Status and Trends

MCR steelhead were listed as threatened in 1999 (64 FR 14517), and their status was reaffirmed on January 5, 2006 (71 FR 834). The ICBTRT (2003) identified 15 populations in four MPGs (Cascades Eastern Slopes Tributaries, John Day River, the Walla Walla and Umatilla Rivers, and the Yakima River) and one unaffiliated independent population (Rock Creek) in this species. There are two extinct populations in the Cascades Eastern Slope MPG: the White Salmon River and Deschutes Crooked River above the Pelton/Round Butte Dam complex.

Seven hatchery steelhead programs are considered part of the MCR steelhead species. These programs propagate steelhead in three of 16 populations and improve kelt survival in one population. No artificial programs produce the winter-run life history in the Klickitat River and Fifteenmile Creek populations. All of the MCR steelhead hatchery programs are designed to produce fish for harvest. However, two hatchery programs are also implemented to augment the naturally spawning populations in the basins where the fish are released. The NMFS assessment of the effects of artificial propagation on MCR steelhead extinction risk concluded that these hatchery programs collectively do not substantially reduce the extinction risk. Artificial propagation increases total species abundance, principally in the Umatilla and Deschutes Rivers. The kelt reconditioning efforts in the Yakima River do not augment natural abundance and benefit the survival of the natural populations. The Touchet River Hatchery program has only recently been established, and its contribution to species viability is uncertain. The hatchery programs affect a small proportion of the species. Collectively, artificial propagation programs provide a slight beneficial effect to species abundance and have neutral or uncertain effects on species productivity, spatial structure, and diversity.

The precise pre-1960 abundance of this species is unknown. However, historic run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby, Wainwright et al. 1996). MCR steelhead run estimates between 1982 and 2004 were calculated by subtracting adult counts for Lower Granite and Priest Rapids Dams from those at Bonneville Dam. The five year average (geometric mean) return of natural MCR steelhead for 1997 to 2001 was up from previous years' basin estimates. Returns to the Yakima River, the Deschutes River, and sections of the John Day River system were substantially higher compared to 1992 to 1997 (Good, Waples et al. 2005). Yakima River returns are still substantially below interim target levels of 8,900 (the current five year average is 1,747 fish) and estimated historical return levels, with the majority of spawning occurring in one tributary, Satus Creek (Berg 2001). The recent five year geometric mean return of the natural-origin component of the Deschutes River run exceeded interim target levels (Good, Waples et al. 2005). Recent five year geometric mean annual returns to the John Day River basin are generally below the corresponding mean returns reported in previous status reviews. However, each major production area in the John Day system has shown upward trends since the 1999 return year (Good, Waples et al. 2005). The Touchet and Umatilla are below their interim abundance targets of 900 and 2,300, respectively. The five year average for these basins is 298 and 1,492 fish, respectively (Good, Waples et al. 2005).

As per the FCRPS (2008), during the most recent 10-year period (for which trends in abundance could be estimated), trends were positive for approximately half of the populations and negative for the remainder. On average, when only natural production is considered, most of the MCR steelhead populations have replaced themselves. The ICBTRT characterizes the diversity risk to all but one MCR steelhead population as "low" to "moderate". The Upper Yakima is rated as having "high" diversity risk because of introgression with resident *O. mykiss* and the loss of presmolt migration pathways.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. MCR steelhead have PCEs of: (1) freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, and adequate passage conditions. Although pristine habitat conditions are still present in some wilderness, roadless, and undeveloped areas, habitat complexity has been greatly reduced in many areas of designated critical habitat for MCR steelhead. Limiting factors identified for MCR steelhead include: (1) Hydropower system mortality; (2) reduced stream flow; (3) impaired passage; (4) excessive sediment; (5) degraded water quality; and (6) altered channel morphology and floodplain.

Northern California Steelhead

Distribution

Northern California steelhead includes steelhead in CC river basins from Redwood Creek south to the Gualala River, inclusive (Figure 28). Table 21 identifies populations within the Northern California Steelhead salmon ESU, their abundances, and hatchery input.

Table 21. Northern California Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Redwood Creek	10,000	Unknown	Unknown
Mad River	6,000	162-384	Unknown
Eel River	82,000	3,127-21,903	Unknown
Mattole River	12,000	Unknown	Unknown
Ten Mile River	9,000	Unknown	Unknown
Noyo River	8,000	Unknown	Unknown
Big River	12,000	Unknown	Unknown
Navarro River	16,000	Unknown	Unknown
Garcia River	4,000	Unknown	Unknown
Gualala River	16,000	Unknown	Unknown
Other Humboldt County streams	3,000	Unknown	Unknown
Other Mendocino County streams	20,000	Unknown	Unknown
Total	**198,000**	**Unknown**	

Life History

Steelhead within this DPS include winter and summer steelhead. Half-pounder juveniles occur in the Mad and Eel Rivers. Half-pounders are immature steelhead that returns to freshwater after only two to four months in the ocean, and generally overwinter in freshwater. These juveniles then outmigrate in the following spring.

Status and Trends

NC steelhead were listed as threatened on June 7, 2000 (65 FR 36074). They retained that classification following a status review on January 5, 2006 (71 FR 834). Long-term data sets are limited for this NC steelhead. Before 1960, estimates of abundance specific to this DPS were available from dam counts in the upper Eel River (Cape Horn Dam–annual avg. no. adults was 4,400 in the 1930s), the South Fork Eel River (Benbow Dam–annual avg. no. adults was 19,000 in the 1940s), and the Mad River (Sweasey Dam–annual avg. no. adults was 3,800 in the 1940s). Estimates of steelhead spawning populations for many rivers in this DPS totaled 198,000 by the mid-1960s.

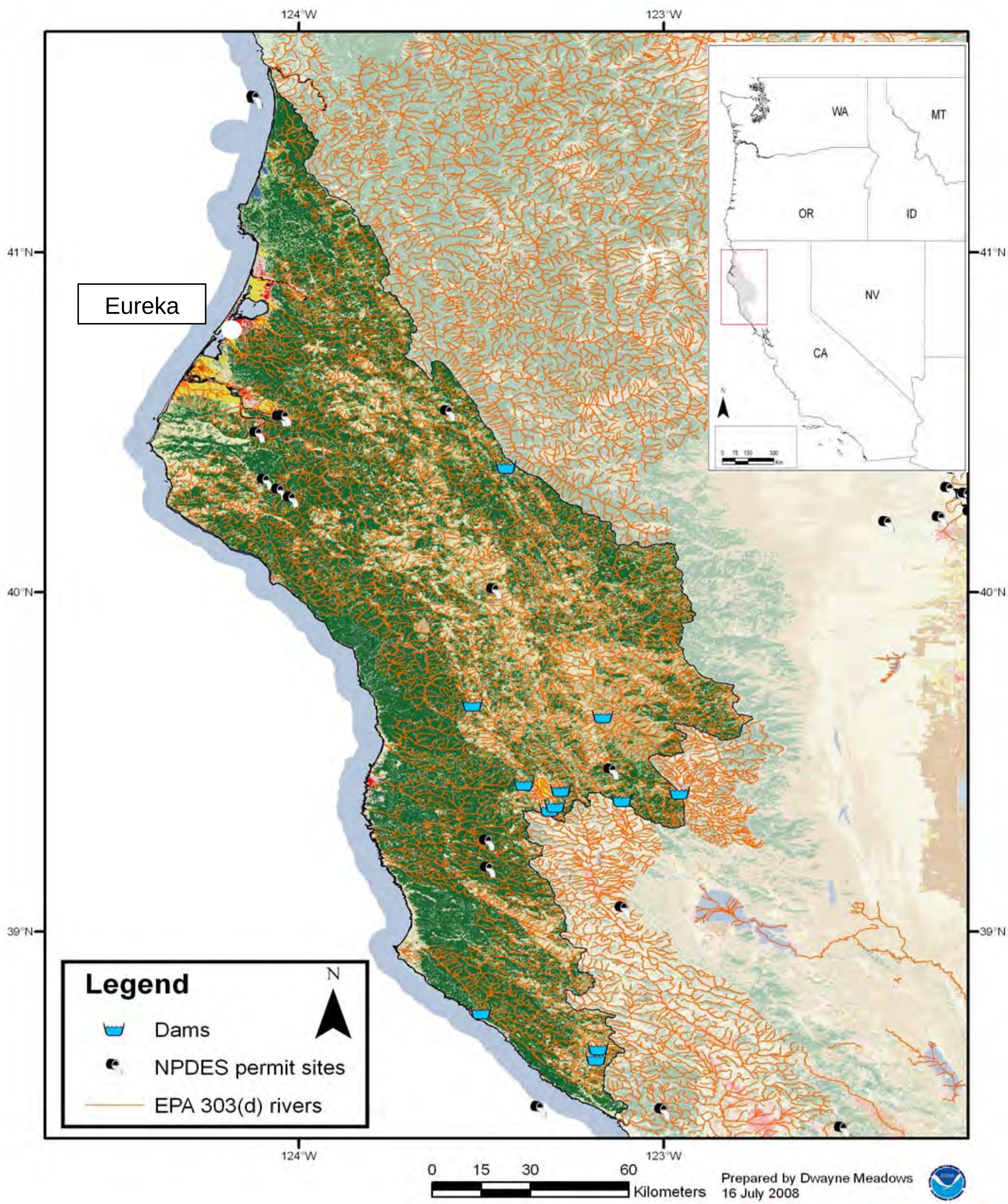


Figure 28. Northern California Steelhead distribution. Land Cover Class Legend in Figure 7.

During the first status review on this population, adult escapement trends could be computed on seven populations. Five of the seven populations exhibited declines while two exhibited increases with a range of almost 6% annual decline to a 3.5% increase. At the time little information was available on the actual contribution of hatchery fish to natural spawning, and on present total run sizes for the DPS (Busby, Wainwright et al. 1996).

More recent time series data are from snorkel counts conducted on summer-run steelhead in the Middle Fork Eel River. An estimate of lambda over the interval 1966 to 2002 was made and a random-walk with drift model fitted using Bayesian assumptions. Good et al. (2005) estimated lambda at 0.98 with a 95% confidence interval of 0.93 and 1.04. The result is an overall downward trend in both the long- and short- term. Juvenile data were also recently examined. Both upward and downward trends were apparent (Good, Waples et al. 2005). The majority (74%) of BRT votes were for "likely to become endangered," with the remaining votes split equally between "in danger of extinction" and "not warranted".

Critical Habitat

Critical habitat was designated for NC steelhead on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. Specific sites include: (1) freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, and adequate forage.

Puget Sound Steelhead

Distribution

Puget Sound steelhead occupy river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington. Included are river basins as far west as the Elwha River and as far north as the Nooksack River (Figure 29). Puget Sound's fjord-like structure may affect steelhead migration patterns. For example, some populations of coho and Chinook salmon, at least historically, remained within Puget Sound and did not migrate to the Pacific Ocean. Even when Puget Sound steelhead migrate to the high seas, they may spend considerable time as juveniles or adults in the protected marine environment of Puget Sound. This is a feature not readily accessible to steelhead from other areas of the Pacific Northwest. The species is primarily composed of winter steelhead but includes several stocks of summer steelhead, usually in subbasins of large river systems and above seasonal hydrologic barriers.

Life History

Life history attributes of Puget Sound steelhead (migration and spawn timing, smolt age, ocean age, and total age at first spawning) appear similar to those of other west coast steelhead. Ocean age for Puget Sound summer steelhead varies among populations.

Status and Trends

Puget Sound steelhead were listed as a threatened species on May 11, 2007 (72 FR 26722). Run size for this DPS, was calculated in the early 1980s at about 100,000 winter-run fish and 20,000 summer-run fish. It is unclear what portion were hatchery fish. However, a combined estimate with coastal steelhead suggested that roughly 70% of steelhead in ocean runs were of hatchery origin. The percentage in escapement to spawning grounds would be substantially lower due to differential harvest and hatchery rack returns. By the 1990s, total run size for four major stocks exceeded 45,000, roughly half of which was natural escapement.

Nehlsen et al. (1991) identified nine Puget Sound steelhead stocks at some degree of risk or concern. The WDFW et al. (1993) estimated that 31 of 53 stocks were of native origin and predominantly natural production. The WDFW assessment of the status of these 31 stocks was 11 healthy, three depressed, one critical, and 16 of unknown status. Their assessment of the status of the remaining (not native/natural) stocks was three healthy, 11 depressed, and eight of unknown status.

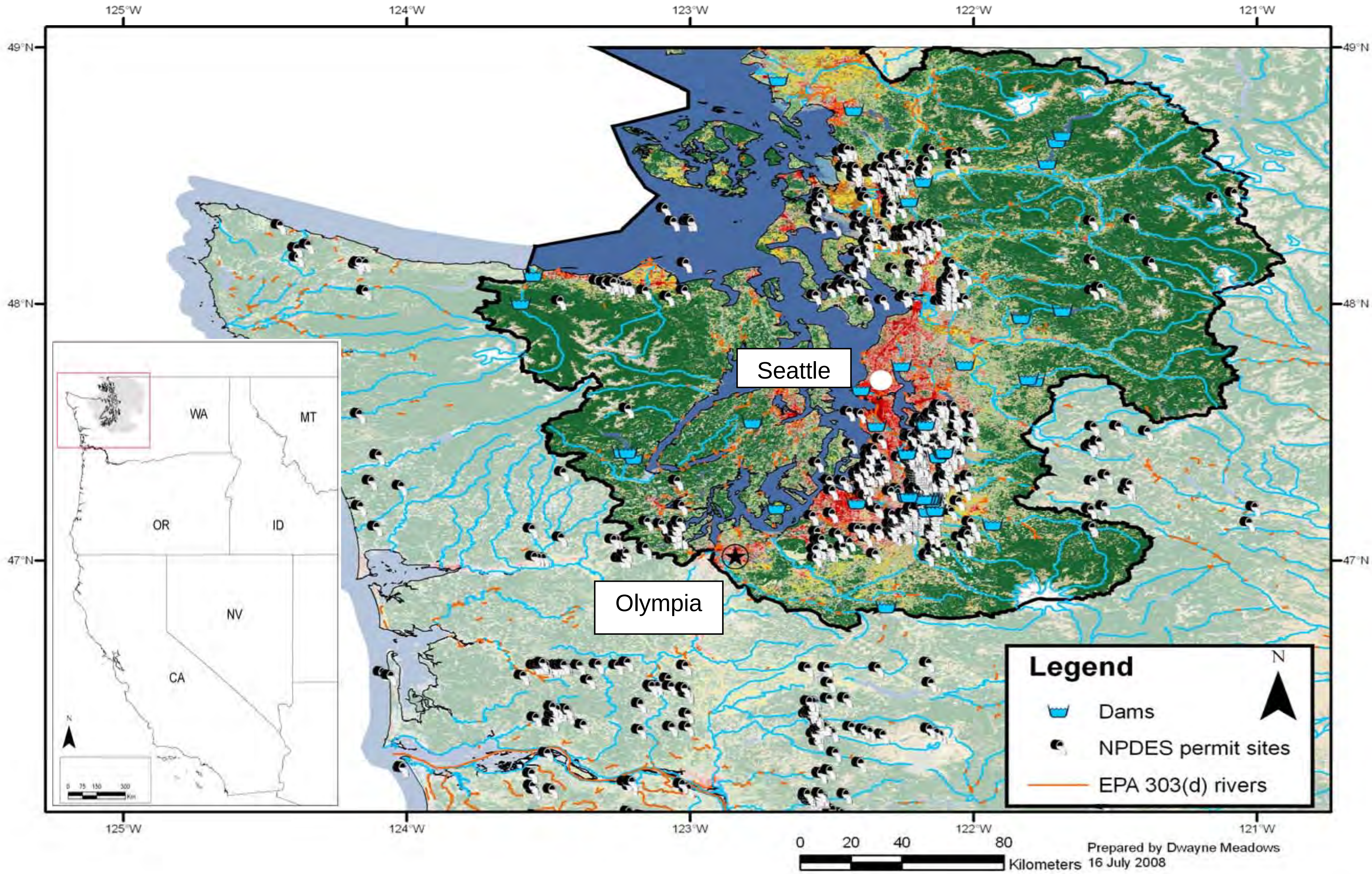


Figure 29. Puget Sound steelhead distribution. Land Cover Class Legend in Figure 7.

Of the 21 populations in the Puget Sound ESU reviewed by Busby et al. (1996), 17 had declining and four had increasing trends, with a range from 18% annual decline (Lake Washington winter-run steelhead) to 7% annual increase (Skykomish River winter-run steelhead). Eleven of these trends (nine negative, two positive) were significantly different from zero. These trends were for the late-run naturally produced component of winter-run steelhead populations. No adult trend data were available for summer-run steelhead. Most of these trends were based on relatively short data series. The Skagit and Snohomish River winter-run populations have been approximately three to five times larger than the other populations in the DPS, with average annual spawning of approximately 5,000 and 3,000 total adult spawners, respectively. These two basins exhibited modest overall upward trends at the time of the Busby et al. (1996) report. Busby et al. (1996) estimated five-year average natural escapements for streams with adequate data range from less than 100 to 7,200, with corresponding total run sizes of 550 to 19,800.

Critical Habitat

Critical habitat is not currently designated for Puget Sound steelhead. However, factors for essential habitat are under evaluation to designate future critical habitat.

Snake River Steelhead

Distribution

SR Basin steelhead is an inland species that occupies the SR basin of Idaho, northeast Oregon, and southeast Washington. The SR Basin steelhead species includes all naturally spawned populations of steelhead (and their progeny) in streams in the SR Basin of Idaho, northeast Oregon, and southeast Washington SR Basin steelhead do not include resident forms of *O. mykiss* (rainbow trout) co-occurring with these steelhead. The historic spawning range of this species included the Salmon, Pahsimeroi, Lemhi, Selway, Clearwater, Wallowa, Grande Ronde, Imnaha, and Tucannon Rivers.

Managers classify up-river summer steelhead runs into two groups based on ocean age and adult size upon return to the Columbia River. A-run steelhead are predominately age-one-ocean fish. B-run steelhead are larger, predominated by age-two-ocean fish. A-run populations are found in the tributaries to the lower Clearwater River, the upper Salmon River and its tributaries, the lower Salmon River and its tributaries, the Grand Ronde River, Imnaha River, and possibly the SR's mainstem tributaries below Hells Canyon Dam. B-run steelhead occupy four major subbasins. They include two on the Clearwater River (Lochsa and Selway) and two on the Salmon River (Middle Fork and South Fork Salmon); areas not occupied by A-run steelhead. Some natural B-run steelhead are also produced in parts of the mainstem Clearwater and its major tributaries. There are alternative escapement objectives of 10,000 (Columbia River Fisheries Management Plan) and 31,400 (Idaho) for B-run steelhead. B-run steelhead represent at least one-third and as much as three-fifths of the production capacity of the DPS. Table 22 identifies populations within the SR Basin Steelhead salmon ESU, their abundances, and hatchery input.

Table 22. SR Basin Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile.

River	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Tucannon River	3,000	257-628	26%
Lower Granite run	Unknown	70,721-259,145	86%
Snake A run	Unknown	50,974-25,950	85%
Snake B run	Unknown	9,736-33,195	89%
Asotin Creek	Unknown	0-543 redds	Unknown
Upper Grande Ronde River	15,000	1.54 rpm	23%
Joseph Creek	Unknown	1,077-2,385	0%
Imnaha River	4,000	3.7 rpm	20%
Camp Creek	Unknown	55-307	0%
Total	**22,000 (min)**	**?**	

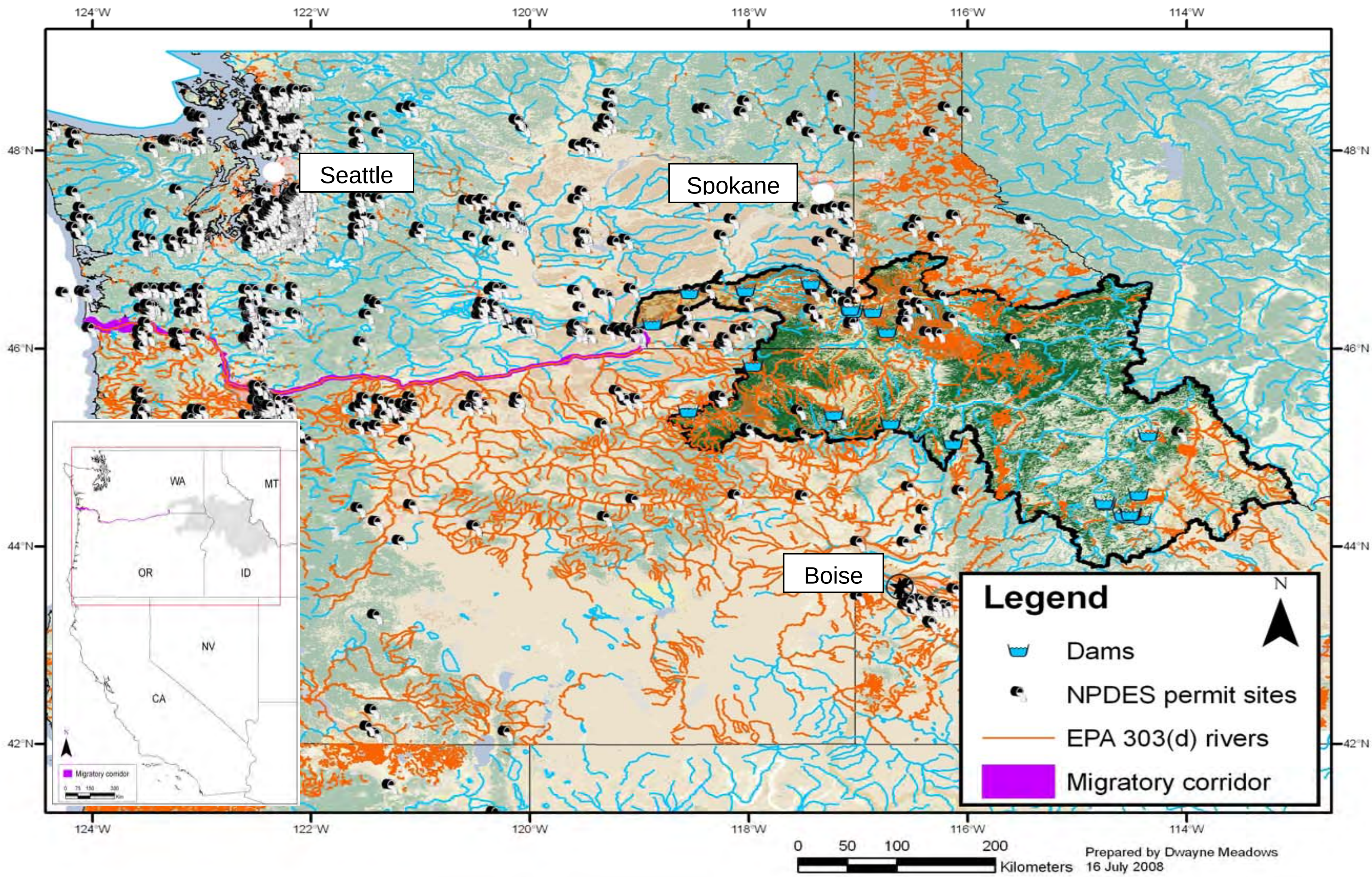


Figure 30. SR Basin Steelhead distribution. Land Cover Class Legend in Figure 7.

Life History

SR Basin steelhead occupy habitat that is considerably warmer and drier (on an annual basis) than other steelhead DPSs. SR Basin steelhead are generally classified as summer run, based on their adult run timing pattern. Sexually immature adult SR Basin summer steelheads enter the Columbia River from late June to October. SR Basin steelhead returns consist of A-run fish that spend one year in the ocean, and larger B-run fish that spend two years at sea. Adults typically migrate upriver until they reach tributaries from 1,000 to 2,000 m above sea level where they spawn between March and May of the following year. Unlike other anadromous members of the *Oncorhynchus* genus, some adult steelhead survive spawning, return to the sea, and later return to spawn a second time. After hatching, juvenile SR Basin steelhead typically spend two to three years in fresh water before they smolt and migrate to the ocean.

Status and Trends

SR Basin steelhead were listed as threatened in 1997 (62 FR 43937). Their classification status was reaffirmed following a status review on January 5, 2006 (71 FR 834). The ICBTRT (2003) identified 23 populations in the following six MPGs: Clearwater River, Grande Ronde River, Hells Canyon, Imnaha River, Lower SR, and Salmon River. SR Basin steelhead remain spatially well distributed in each of the six major geographic areas in the SR basin (Good, Waples et al. 2005). Environmental conditions are generally drier and warmer in these areas than in areas occupied by other steelhead species in the Pacific Northwest. SR Basin steelhead were blocked from portions of the upper SR beginning in the late 1800s and culminating with the construction of Hells Canyon Dam in the 1960s. The SR Basin steelhead "B run" population-levels remain particularly depressed. The ICBTRT has not completed a viability assessment for SR Basin steelhead.

Limited information on adult spawning escapement for specific tributary production areas for SR Basin steelhead made a quantitative assessment of viability difficult. Annual return estimates are limited to counts of the aggregate return over Lower Granite Dam, and spawner estimates for the Tucannon, Grande Ronde, and Imnaha Rivers. The 2001 return over Lower Granite Dam was substantially higher relative to the low levels seen in the 1990s; the recent five-year mean abundance (14,768 natural returns) was approximately 28% of the interim recovery target level. The 10-year average for natural-origin steelhead passing Lower Granite Dam between 1996 and 2005 is 28,303 adults. Parr densities in natural production areas, which are another indicator of population status, have been substantially below estimated capacity for several decades. The SR supports approximately 63% of the total natural-origin production of steelhead in the Columbia River Basin. The current condition of SR Basin steelhead (Good, Waples et al. 2005) is summarized below:

There is uncertainty for wild populations given limited data for adult spawners in individual populations. Dam counts are currently 28% of interim recovery target for the SR Basin (52,000 natural spawners). Only the Joseph Creek population exceeds the interim recovery target. Regarding population growth rate, there are mixed long- and short-term trends in abundance and productivity. Regarding spatial structure, the SR Basin steelhead are well distributed with populations remaining in six major areas. However, the core area for B-run steelhead, once located in the North Fork of the Clearwater River, is now inaccessible to steelhead. Finally, genetic diversity is affected by the displacement of natural fish by hatchery fish (declining proportion of natural-origin spawners). Homogenization of hatchery stocks occurs within basins, and some stocks exhibit high stray rates.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more steelhead life stages. Specific sites include: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality.

Of the 291 fifth order streams reviewed in this DPS, 220 were rated as high, 44 were rated as medium, and 27 were rated as low conservation value. The physical or biological features that characterize these sites include water quality and quantity, natural cover, and adequate forage. Limiting factors identified for SR Basin salmonids include: (1) Hydrosystem mortality, (2) reduced stream flow, (3) altered channel morphology and floodplain, (4) excessive sediment, (5) degraded water quality, (6) harvest impacts, and (7) hatchery impacts (Myers, Kope et al. 1998).

South-Central California Coast Steelhead

Distribution

The South-Central California Coast (S-CCC) steelhead DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California (Figure 31).

Life History

Only winter steelhead are found in this DPS. Migration and spawn timing are similar to adjacent steelhead populations. There is little other life history information for steelhead in this DPS.

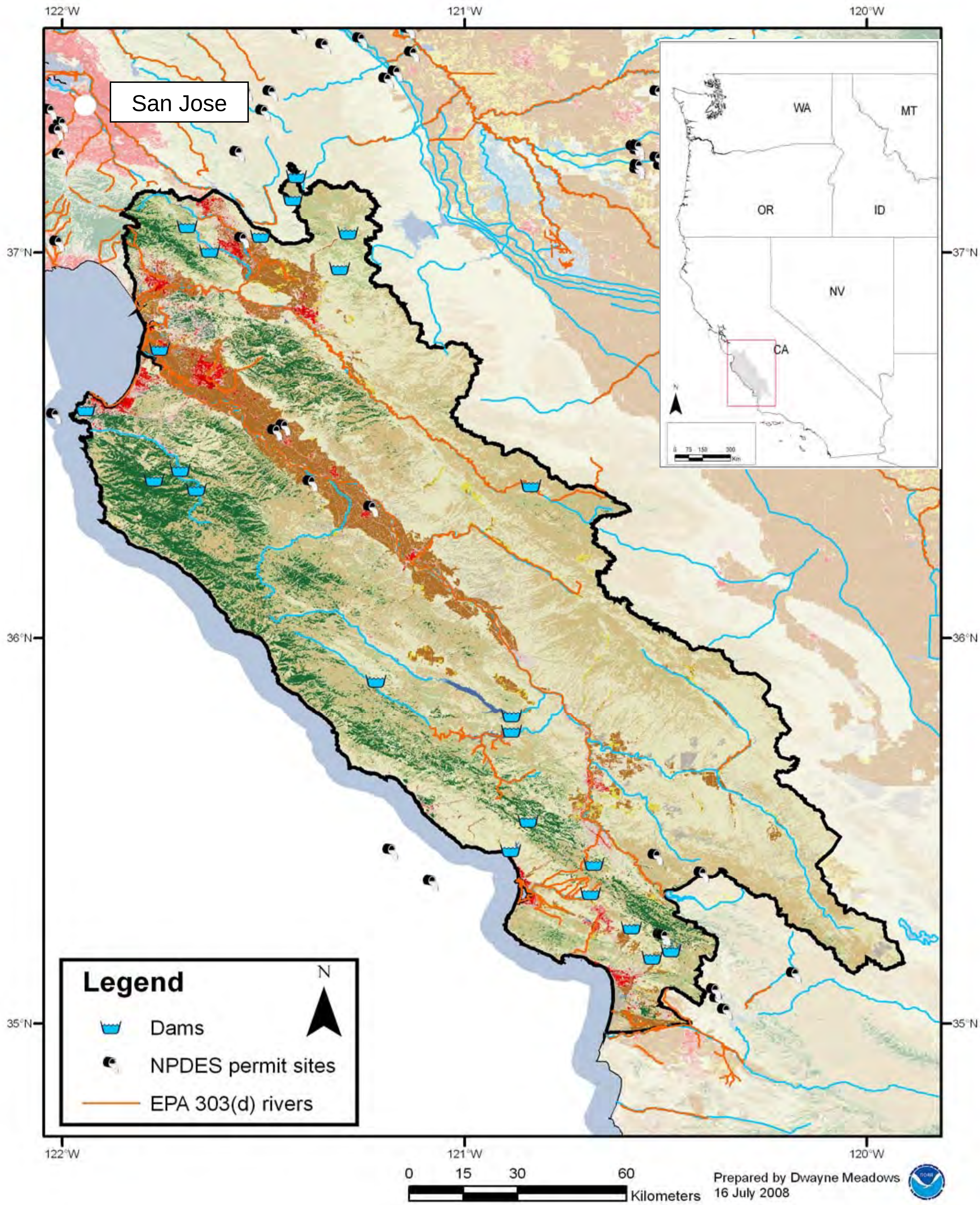


Figure 31. S-CCC steelhead distribution. Land Cover Class Legend in Figure 7.

Status and Trends

S-CCC steelhead were listed as threatened in 1997. Their classification was retained following a status review on January 5, 2006 (71 FR 834). Historical data on the S-C CC steelhead DPS are limited. In the mid-1960s, the CDFG estimated the adult population at about 18,000. We know of no recent estimates of the total DPS. However, five river systems, the Pajaro, Salinas, Carmel, Little Sur, and Big Sur, indicate that runs are currently less than 500 adults. Past estimates for these basins were almost 5,000 fish. Carmel River time series data indicate that the population declined by about 22% per year between 1963 and 1993 (Good, Waples et al. 2005). From 1991 the population increased from one adult, to 775 adults at San Clemente Dam. Good et al. (2005) thought that this recent increase seemed too great to attribute simply to improved reproduction and survival of the local steelhead population. Other possibilities were considered including that the substantial immigration or transplantation occurred, or that resident trout production increased as a result of improved environmental conditions within the basin. Nevertheless, the majority (68%) of BRT votes were for “likely to become endangered,” and another 25% were for “in danger of extinction”.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages. Specific sites include: (1) freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, and adequate forage.

Southern California Steelhead

Distribution

Southern California (SC) steelhead occupy rivers from the Santa Maria River to the U.S. –Mexico border (Figure 32).

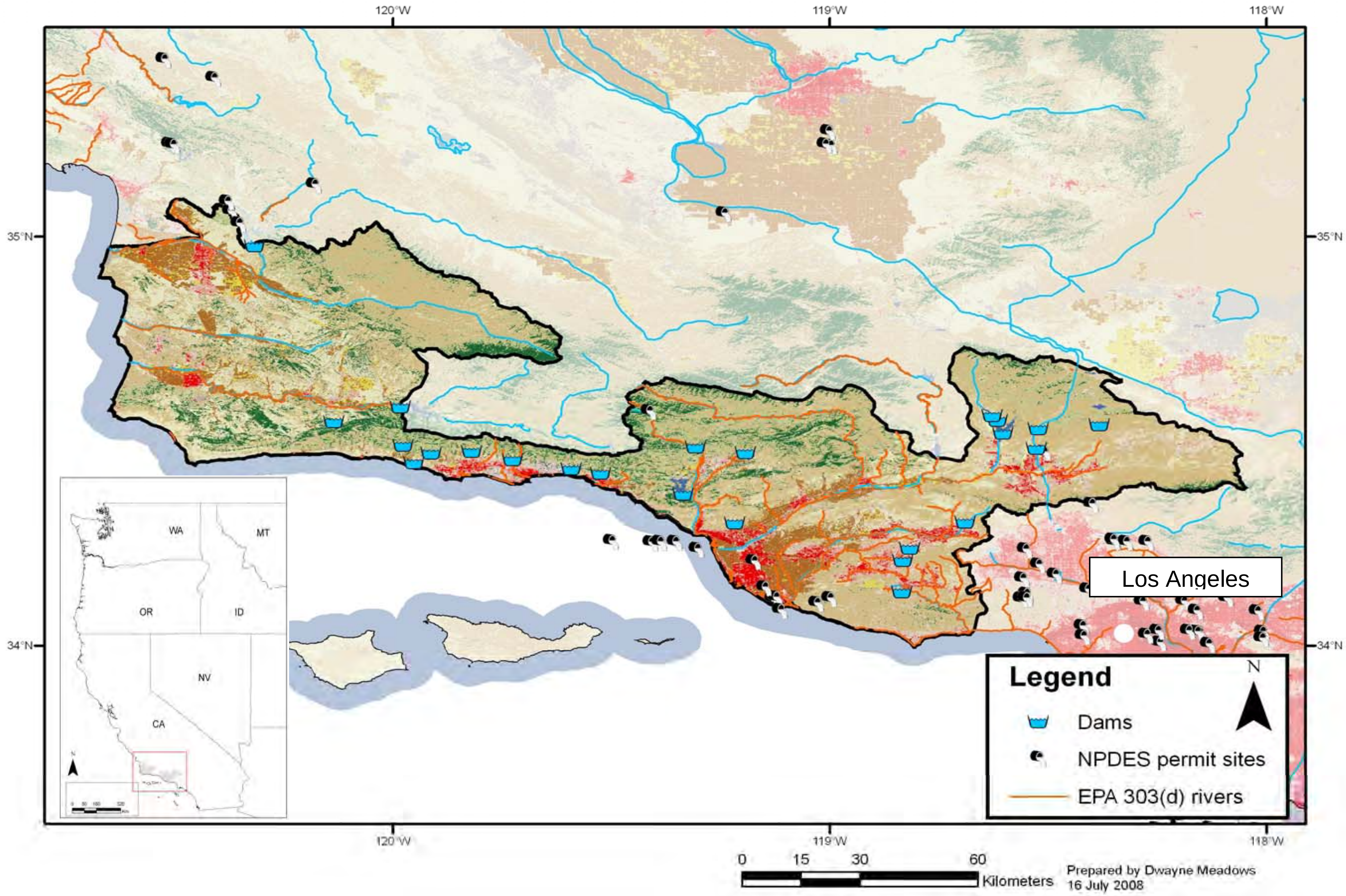


Figure 32. Southern California steelhead distribution. Land Cover Class Legend in Figure 7.

Table 23 identifies populations within the Southern California Steelhead salmon ESU, their abundances, and hatchery input.

Table 23. Southern California Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Santa Ynez River	12,995-30,000	Unknown	Unknown
Ventura River	4,000-6,000	Unknown	Unknown
Matilija River	2,000-2,500	Unknown	Unknown
Creek River	Unknown	Unknown	Unknown
Santa Clara River	7,000-9,000	Unknown	Unknown
Total	**32,000-46,000**	**<500**	

Life History

Migration and life history patterns of SC steelhead are dependent on rainfall and streamflow (Moore 1980). Steelhead within this DPS can withstand higher temperatures than populations to the north. The relatively warm and productive waters of the Ventura River have resulted in more rapid growth of juvenile steelhead than occurs in more northerly populations (Moore 1980). There is little life history information for steelhead in this DPS.

Status and Trends

SC steelhead were listed as endangered in 1997 (62 FR 43937). Their classification was retained following a status review on January 5, 2006 (71 FR 834). In many watersheds throughout Southern California, dams isolate steelhead from historical spawning and rearing habitats. Dams also alter the hydrology of the basin (*e.g.*, Twitchell Reservoir within the Santa Maria River watershed, Bradbury Dam within the Santa Ynez River watershed, Matilija and Casitas dams within the Ventura River watershed, Rindge Dam within the Malibu Creek watershed). Based on combined estimates for the Santa Ynez, Ventura, and Santa Clara rivers, and Malibu Creek, an estimated 32,000 to 46,000 adult steelhead occupied this DPS. In contrast, less than 500 adults are estimated to occupy the same four waterways presently. The last estimated run size for steelhead in the Ventura River, which has its headwaters in Los Padres National Forest, is 200 adults (Busby, Wainwright et al. 1996). The majority (81%) of the BRT votes were for “in danger of extinction," with the remaining 19% of votes for "likely to become endangered. This was based on extremely strong concern for abundance, productivity, and spatial concern (as per the risk matrix); diversity was also of concern. The BRT also expressed concern about the lack of data on the SC steelhead DPS, including uncertainty on the metapopulation dynamics in the southern part of the DPS's range and the fish's nearly complete extirpation from the southern part of the range.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005. The designation identifies PCEs that include sites necessary to support one or more steelhead life stages. These sites contain the physical or biological features essential for the species conservation. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The physical or biological features that characterize these sites include water quantity, depth, and velocity, shelter, cover, living space and passage conditions.

Upper Columbia River Steelhead

Distribution

UCR steelhead occupy the Columbia River Basin upstream from the Yakima River, Washington, to the border between the U.S. and Canada (Figure 33). This area includes the Wenatchee, Entiat, and Okanogan Rivers. All UCR steelhead are summer steelhead. Steelhead primarily use streams of this region that drain the northern Cascade Mountains of Washington State. This species includes hatchery populations of summer steelhead from the Wells Hatchery because it probably retains the genetic resources of steelhead populations that once occurred above the Grand Coulee Dam. This species does not include the Skamania Hatchery stock because of its non-native genetic heritage.

Abundance estimates of returning naturally produced UCR steelhead have been based on extrapolations from mainstem dam counts and associated sampling information (*e.g.*, hatchery/wild fraction, age composition). The natural component of the annual steelhead run over Priest Rapids Dam increased from an average of 1,040 (1992-1996), representing about 10% of the total adult count, to 2,200 (1997-2001), representing about 17% of the adult count during this period of time (ICBTRT 2003). Table 24 identifies populations within the UCR Steelhead salmon DPS, their abundances, and hatchery input.

Table 24. UCR Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Wenatchee/Entiat rivers	Unknown	1,899-8,036	71%
Methow/Okanogan rivers	Unknown	1,879-12,801	91%
Total	**Unknown**	**3,778-20,837**	

Life History

The life history patterns of UCR steelhead are complex. Adults return to the Columbia River in the late summer and early fall. Most migrate relatively quickly up the mainstem to their natal tributaries. A portion of the returning run overwinters in the mainstem reservoirs, passing over the upper-mid-Columbia dams in April and May of the following year. Spawning occurs in the late spring of the calendar year following entry into the river. Juvenile steelhead spend one to seven years rearing in freshwater before migrating to sea. Smolt outmigrations are predominantly age-two and age-three juveniles. Most adult steelhead return after one or two years at sea, starting the cycle again.

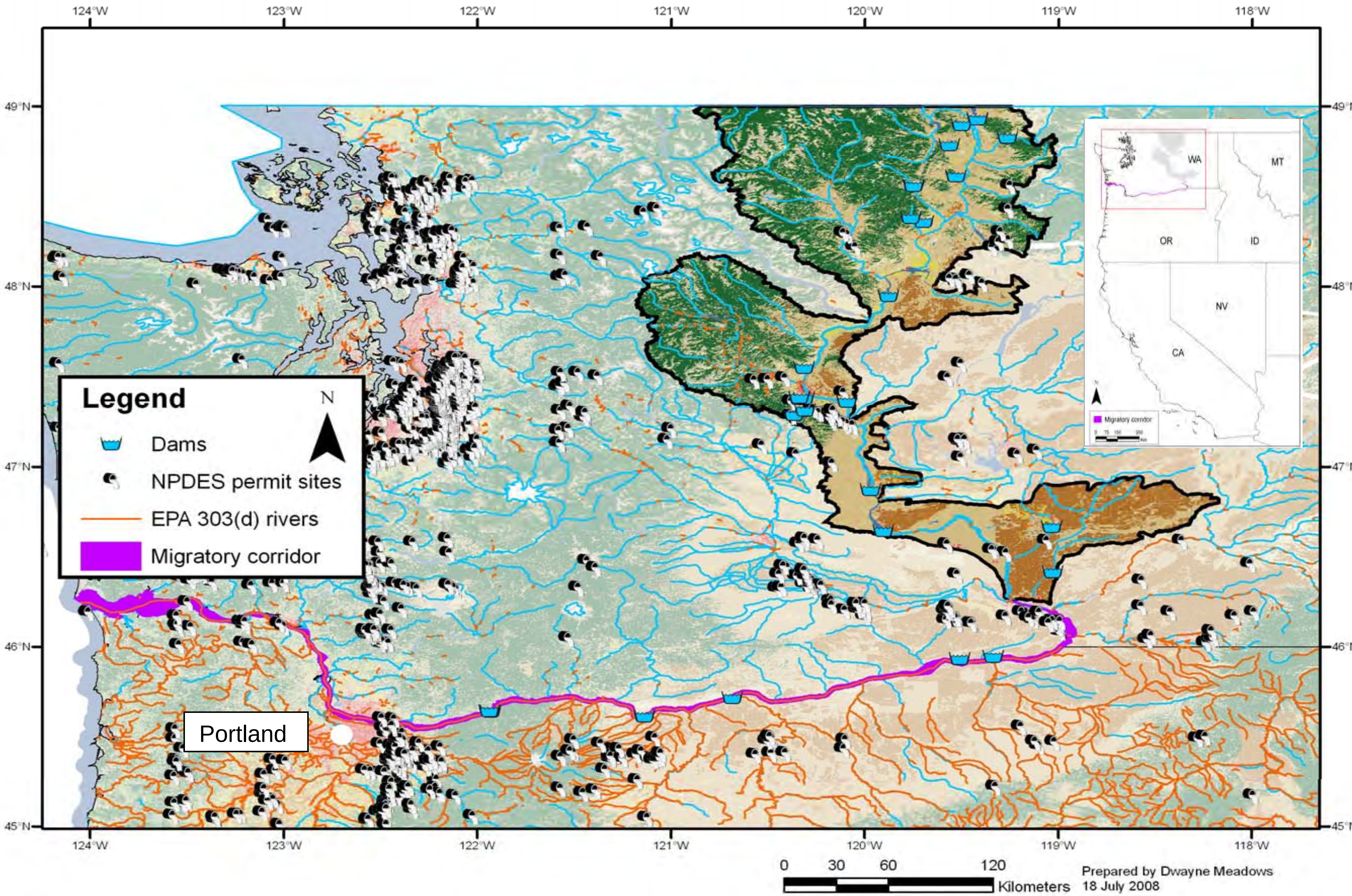


Figure 33. UCR Steelhead distribution. Land Cover Class Legend in Figure 7.

Returns of both hatchery and naturally produced steelhead to the UCR have increased in recent years. The average 1997 to 2001 return counted through the Priest Rapids fish ladder was approximately 12,900 fish. The average for the previous five years (1992 to 1996) was 7,800 fish. Abundance estimates of returning naturally produced UCR steelhead have been based on extrapolations from mainstem dam counts and associated sampling information (*e.g.*, hatchery/wild fraction, age composition). The natural component of the annual steelhead run over Priest Rapids Dam increased from an average of 1,040 (1992-1996), representing about 10% of the total adult count, to 2,200 (1997-2001), representing about 17% of the adult count during this period of time (ICBTRT 2003).

In terms of natural production, recent population abundances for both the Wenatchee and Entiat aggregate population and the Methow population remain well below the minimum abundance thresholds developed for these populations (ICBTRT 2005). A five-year geometric mean (1997 to 2001) of approximately 900 naturally produced steelhead returned to the Wenatchee and Entiat rivers (combined). Although this is well below the minimum abundance thresholds, it represents an improvement over the past (an increasing trend of 3.4% per year). However, the average percentage of natural fish for the recent five-year period dropped from 35% to 29%, compared to the previous status review. For the Methow population, the five-year geometric mean of natural returns over Wells Dam was 358. Although this is well below the minimum abundance thresholds, it is an improvement over the recent past (an increasing trend of 5.9% per year). In addition, the 2001 return (1,380 naturally produced spawners) was the highest single annual return in the 25-year data series. However, the average percentage of wild origin spawners dropped from 19% for the period prior to the 1998 status review to 9% for the 1997 to 2001 returns.

Regarding the population growth rate of natural production, on average, over the last 20 full brood year returns (1980/81 through 1999/2000 brood years), including adult returns through 2004-2005, UCR steelhead populations have not replaced themselves. The ICBTRT has characterized the spatial structure risk to UCR steelhead populations as “low” for the Wenatchee and Methow, “moderate” for the Entiat, and “high” for the Okanogan. Overall adult returns are dominated by hatchery fish, and detailed information is lacking on the productivity of the natural population. All UCR steelhead populations have reduced genetic diversity from homogenization of populations that occurred during the Grand Coulee Fish Maintenance project from 1939-1943, from 1960, and 1981 (Chapman, Hillman et al. 1994).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages. They include all Columbia River estuarine areas and river reaches upstream to Chief Joseph Dam and several tributary subbasins. Specific sites include freshwater spawning and rearing sites, freshwater migration corridors, estuarine areas free of obstruction, and offshore marine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, and adequate passage conditions.

The UCR steelhead DPS has 42 watersheds within its range. Three watersheds received a low rating, eight received a medium rating, and 31 rated a high conservation value to the DPS. In addition, the Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Limiting factors identified for the UCR steelhead include: (1) Mainstem Columbia River hydropower system mortality, (2) reduced tributary streamflow, (3) tributary riparian degradation and loss of in-river wood, (4) altered tributary floodplain and channel morphology, and (5) excessive fine sediment and degraded tributary water quality. The above activities and features also introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest.

Upper Willamette River Steelhead

Distribution

UWR steelhead occupy the Willamette River and its tributaries upstream of Willamette Falls (Figure 34). This is a late-migrating winter group that enters freshwater in March and April (Howell, Jones et al. 1985). Only the late run was included in the listing of this species, which is the largest remaining population in the Santiam River system. Table 25 identifies populations within the UWR Steelhead salmon ESU, their abundances, and hatchery input.

Table 25. UWR Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile.

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Mollala Rivers	Unknown	0.972 rpm	Unknown
North Santiam River	Unknown	0.963 rpm	Unknown
South Santiam River	Unknown	0.917 rpm	Unknown
Calapooia River	Unknown	1.053 rpm	Unknown
Total	**Unknown**	**5,819**	

Life History

Winter steelhead enter the Willamette River beginning in January and February. They do not ascend to their spawning areas until late March or April (Dimick and Merryfield 1945). Spawning occurs from April to June 1st and redd counts are conducted in May. The smolt migration past Willamette Falls also begins in early April and extends through early June (Howell, Jones et al. 1985) Migration peaks in early- to mid-May. Steelhead smolts generally migrate away from the shoreline and enter the Columbia via Multnomah Channel rather than the mouth of the Willamette. Most spend two years in the ocean before re-entering fresh water to span (Busby, Wainwright et al. 1996). Steelhead in the UWR DPS generally spawn once or twice. A few fish may spawn three times based on patterns found in the LCR steelhead DPS. Repeat spawners are predominantly female and generally account for less than 10% of the total run size (Busby, Wainwright et al. 1996).

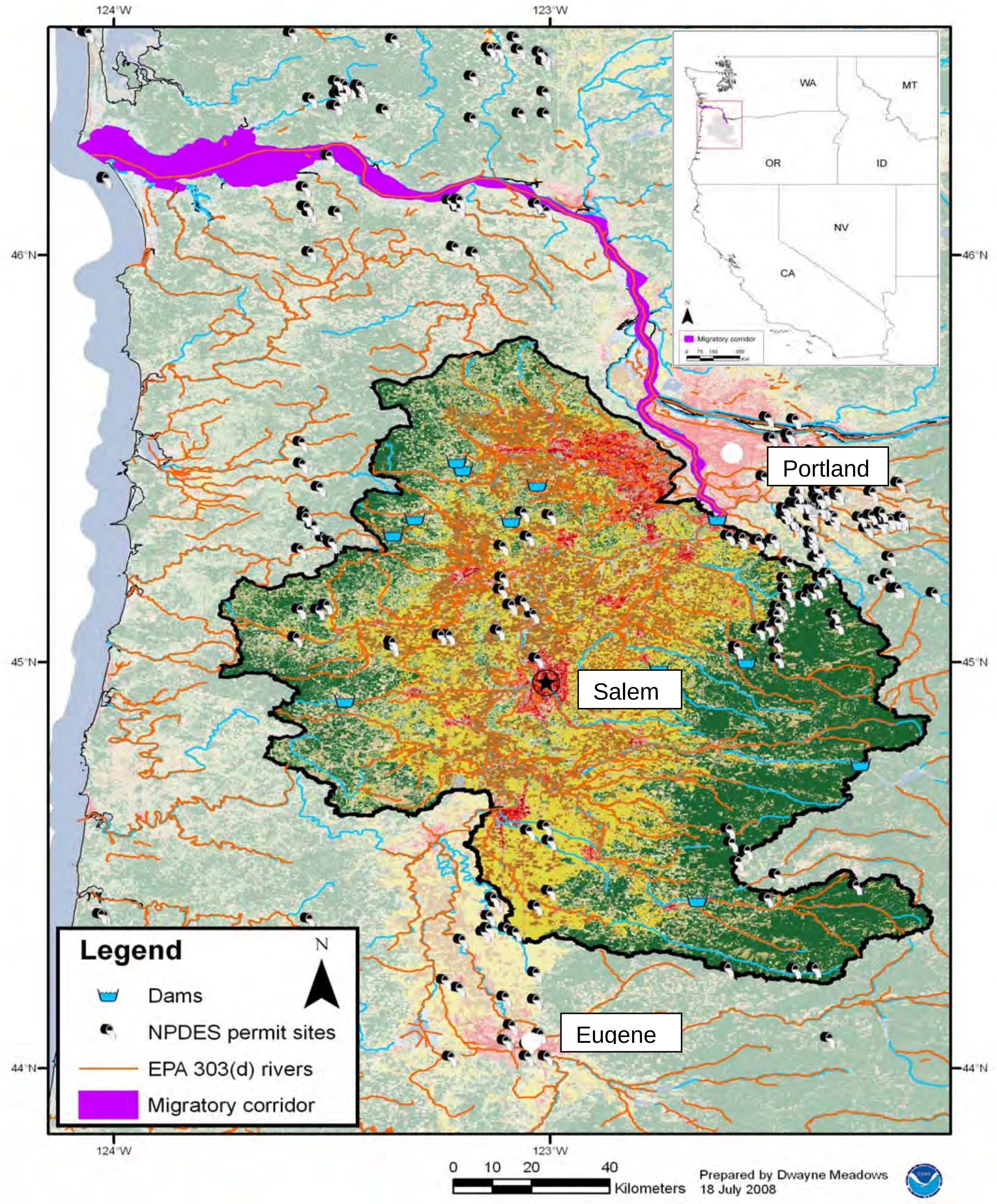


Figure 34. UWR Steelhead distribution. Land Cover Class Legend in Figure 7.

Status and Trends

UWR steelhead were listed as threatened in 1999 (64 FR 14517). Their classification was retained following a status review on January 5, 2006 (71 FR 834). A major threat to Willamette River steelhead results from artificial production practices. Fishways built at Willamette Falls in 1885 have allowed Skamania-stock summer steelhead and early-migrating winter steelhead of Big Creek stock to enter the range of UWR steelhead. The population of summer steelhead is almost entirely maintained by hatchery salmon, although natural-origin, Big Creek-stock winter steelhead occur in the basin (Howell, Jones et al. 1985). In recent years, releases of winter steelhead are primarily of native stock from the Santiam River system.

Steelhead in this DPS are depressed from historical levels, but to a much lesser extent than are spring Chinook in the Willamette basin (McElhaney, Chilcote et al. 2007). All of the historical populations remain extant and moderate numbers of wild steelhead are produced each year. The population growth rate data indicate long-term trends are <1; short-term trends are 1 or higher (McElhaney, Chilcote et al. 2007). Spatial structure for the North and South Santiam populations has been substantially reduced by the loss of access to the upper North Santiam basin and the Quartzville Creek watershed in the South Santiam subbasin due to construction of the dams owned and operated by the U.S. Army Corps of Engineers without passage facilities (McElhaney, Chilcote et al. 2007). Additionally, the spatial structure in the Molalla subbasin has been reduced significantly by habitat degradation and in the Calapooia by habitat degradation and passage barriers. Finally, the diversity of some populations have been eroded by small population size, the loss of access to historical habitat, legacy effects of past winter-run hatchery releases, and the ongoing release of summer steelhead (McElhaney, Chilcote et al. 2007).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). It includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Willamette River as well as specific steam reaches in the following subbasins: Upper Willamette, North Santiam, South Santiam, Middle Willamette, Molalla/Pudding, Yamhill, Tualatin, and Lower Willamette (NMFS 2005b). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages. Specific sites include: (1) Freshwater spawning, (2) freshwater rearing, (3) freshwater migration, (4) estuarine areas free of obstruction, (5) nearshore marine areas free of obstructions, and (6) offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Anthropogenic land uses introduce sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest. These human impacts affect the essential feature requirements for this DPS.

Of 43 subbasins reviewed in NMFS' assessment of critical habitat for the UWR steelhead, 20 subbasins were rated as having a high conservation value, while six were rated as having a medium value and 17 were rated as having a low value to the conservation of the DPS.

Environmental Baseline

By regulation, environmental baselines for Opinions include the past and present impacts of all state, federal or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR §402.02). The environmental baseline for this Opinion includes a general description of the natural and anthropogenic factors influencing the current status of listed Pacific salmonids and the environment within the action area.

Our summary of the environmental baseline complements the information provided in the *Status of Listed Resources* section of this Opinion, and provides the background necessary to understand information presented in the *Effects of the Action*, and *Cumulative Effects* sections of this Opinion. We then evaluate these consequences in combination with the environmental baseline to determine the likelihood of jeopardy or adverse modification of designated critical habitat.

The proposed action under consultation is geographically focused on the aquatic ecosystems in the states of California, Idaho, Oregon, and Washington. Accordingly, the environmental baseline for this consultation focuses on the general status and trends of the aquatic ecosystems in these four states and the consequences of that status for listed resources under NMFS' jurisdiction. We describe the overall principal natural phenomena affecting all listed Pacific salmonids under NMFS jurisdiction in the action area.

We further describe anthropogenic factors through the predominant land and water uses within a region, as land use patterns vary by region. Background information on pesticides in the aquatic environment is also provided. This context illustrates how the physical and chemical health of regional waters and the impact of human activities have contributed to the current status of listed resources in the action area.

Natural Mortality Factors

Available data indicate high natural mortality rates for salmonids, especially in the open ocean/marine environment. According to Bradford (1997), salmonid mortality rates range from 90 to 99%, depending on the species, the size at ocean entry, and the length of time spent in the ocean. Predation, inter- and intraspecific competition, food availability, smolt quality and health, and physical ocean conditions likely influence the survival of salmon in the marine environment (Bradford 1997; Brodeur, Fisher et al. 2004). In freshwater rearing habitats, the natural mortality rate averages about 70% for all salmonid species (Bradford 1997). Past studies in the Pacific Northwest suggest that the average freshwater survival rate (from egg to smolt) is 2 to 3% throughout the region (Marshall and Britton 1990; Bradford 1997). A number of suspected causes contributing to natural mortality include parasites and/or disease, predation, water temperature, low water flow, wildland fire, and oceanographic features and climatic variability.

Parasites and/or Disease

Most young fish are highly susceptible to disease during the first two months of life. The cumulative mortality in young animals can reach 90 to 95%. Although fish disease organisms occur naturally in the water, native fish have co-evolved with them. Fish can carry these diseases at less than lethal levels (Kier Associates 1991; Walker and Foott 1993; Foott, Harmon et al. 2003). However, disease outbreaks may occur when water quality is diminished and fish are stressed from crowding and diminished flows (Spence, Lomnicky et al. 1996; Guillen 2003). Young coho salmon or other salmonid species may become stressed and lose their resistance in higher temperatures (Spence, Lomnicky et al. 1996). Consequently, diseased fish become more susceptible to predation and are less able to perform essential functions, such as feeding, swimming, and defending territories (McCullough 1999). Examples of parasites and disease for salmonids include whirling disease, infectious hematopoietic necrosis (IHN), sea-lice (*Lepeophtheirus salmonis*), *Henneguya salminicola*, *Ichthyopthirius multifiliis* or Ich, and Columnaris (*Flavobacterium columnare*).

Whirling disease is a parasitic infection caused by the microscopic parasite *Myxobolus cerebrali*. Infected fish continually swim in circular motions and eventually expire from exhaustion. The disease occurs in the wild and in hatcheries and results in losses to fry and fingerling salmonids, especially rainbow trout. The disease is transmitted by infected fish and fish parts and birds.

IHN is a viral disease in many wild and farmed salmonid stocks in the Pacific Northwest. This disease affects rainbow/steelhead trout, cutthroat trout (*Salmo clarki*), brown trout (*Salmo trutta*), Atlantic salmon (*Salmo salar*), and Pacific salmon including Chinook, sockeye, chum, and coho. The virus is triggered by low water temperatures and is shed in the feces, urine, sexual fluids, and external mucus of salmonids. Transmission is mainly from fish to fish, primarily by direct contact and through the water.

Sea lice also cause deadly infestations of wild and farm-grown salmon. *Henneguya salminicola*, a protozoan parasite, is commonly found in the flesh of salmonids. The fish responds by walling off the parasitic infection into a number of cysts that contain milky fluid. This fluid is an accumulation of a large number of parasites. Fish with the longest freshwater residence time as juveniles have the most noticeable infection. The order of prevalence for infection is coho followed by sockeye, Chinook, chum, and pink salmon.

Additionally, ich (a protozoan) and Columnaris (a bacterium) are two common fish diseases that were implicated in the massive kill of adult salmon in the Lower Klamath River in September 2002 (CDFG 2003; Guillen 2003).

Predation

Salmonids are exposed to high rates of natural predation, during freshwater rearing and migration stages, as well as during ocean migration. Salmon along the U.S. west coast are prey for marine mammals, birds, sharks, and other fishes. Concentrations of juvenile salmon in the coastal zone experience high rates of predation. In the Pacific Northwest, the increasing size of tern, seal, and sea lion populations may have reduced the survival of some salmon ESUs.

Marine Mammal Predation

Marine mammals are known to attack and eat salmonids. Harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and killer whales (*Orcinus orca*) prey on juvenile or adult salmon. Killer whales have a strong preference for Chinook salmon (up to 78% of identified prey) during late spring to fall (Hard, Jones et al. 1992; Hanson, Baird et al. 2005; Ford and Ellis 2006). Generally, harbor seals do not feed on salmonids as frequently as California sea lions (Pearcy 1997). California sea lions from the Ballard Locks in Seattle, Washington have been estimated to consume about 40% of the steelhead runs since 1985/1986 (Gustafson, Wainwright et al. 1997). In the Columbia River, salmonids may contribute substantially to sea lion diet at specific times and locations (Pearcy 1997). Spring Chinook salmon and steelhead are subject to pinniped predation when they return to the estuary as adults (NMFS 2006). Adult Chinook salmon in the Columbia River immediately downstream of Bonneville Dam have also experienced increased predation by California sea lions. In recent years, sea lion predation of adult LCR winter steelhead in the Bonneville tailrace has increased. This prompted ongoing actions to reduce predation effects. They include the exclusion, hazing, and in some cases, lethal take of marine mammals near Bonneville Dam (FCRPS 2008).

NOAA Fisheries has granted permits to the states of Idaho, Oregon, and Washington for the lethal removal of individual California sea lions that prey on adult spring-run Chinook salmon in the tail race of Bonneville Dam under section 120 of the Marine Mammal Protection Act (NMFS 2006). This action may increase the survival of adult Chinook salmon and steelhead. The Humane Society of the U.S. unsuccessfully challenged the issuance of these permits. The case is now on appeal.

Avian Predation

Large numbers of fry and juveniles are eaten by birds such as mergansers (*Mergus* spp.), common murre (*Uria aalage*), gulls (*Larus* spp.), and belted kingfishers (*Megaceryle alcyon*). Avian predators of adult salmonids include bald eagles (*Haliaeetus leucocephalus*) and osprey (*Pandion haliaetus*) (Pearcy 1997). Caspian terns (*Sterna* *caspia*) and cormorants (*Phalacrocorax* spp.) also take significant numbers of juvenile or adult salmon. Stream-type juveniles, especially yearling smolts from spring-run populations, are vulnerable to bird predation in the estuary. This vulnerability is due to salmonid use of the deeper, less turbid water over the channel, which is located near habitat preferred by piscivorous birds (Binelli, Ricciardi et al. 2005). Recent research shows that subyearlings from the LCR Chinook salmon ESU are also subject to tern predation. This may be due to the long estuarine residence time of the LCR Chinook salmon (Ryan, Carper et al. 2006). Caspian terns and cormorants may be responsible for the mortality of up to 6% of the outmigrating stream-type juveniles in the Columbia River basin (Roby, Collis et al. 2006; Collis 2007).

Antolos *et al.* (2005) quantified predation on juvenile salmonids by Caspian terns nesting on Crescent Island in the mid-Columbia reach. Between 1,000 and 1,300 adult terns were associated with the colony during 2000 and 2001, respectively. These birds consumed about 465,000 juvenile salmonids in the first and approximately 679,000 salmonids in the second year. However, caspian tern predation in the estuary was reduced from a total of 13,790,000 smolts to 8,201,000 smolts after relocation of the colony from Rice to East Sand Island in 1999. Based on PIT-tag recoveries at the colony, these were primarily steelhead for Upper Columbia River stocks. Less than 0.1% of the inriver migrating yearling Chinook salmon from the Snake River and less than 1% of the yearling Chinook salmon from the Upper Columbia were consumed. PIT-tagged coho smolts (originating above Bonnevile Dam) were second only to steelhead in predation rates at the East Sand Island colony in 2007 (Roby, Colis et al. 2008). There are few quantitative data on avian predation rates on Snake River sockeye salmon. Based on the above, avian predators are assumed to have a minimal effect on the long-term survival of Pacific salmon (FCRPS 2008).

Fish Predation

Pikeminnows (*Ptychocheilus oregonensis*) are significant predators of yearling juvenile migrants (Friesen and Ward 1999). Chinook salmon were 29% of the prey of northern pikeminnows in lower Columbia reservoirs, 49% in the lower Snake River, and 64%

downstream of Bonneville Dam. Sockeye smolts comprise a very small fraction of the overall number of migrating smolts (Ferguson 2006) in any given year. The significance of fish predation on juvenile chum is unknown. There is little direct evidence that piscivorous fish in the Columbia River consume juvenile sockeye salmon. Nevertheless, predation of juvenile sockeye likely occurs. The ongoing Northern Pikeminnow Management Program (NPMP) has reduced predation-related juvenile salmonid mortality since 1990. Benefits of recent northern pikeminnow management activities to chum salmon are unknown. However, it may be comparable to those for other salmon species with a subyearling juvenile life history (Friesen and Ward 1999).

The primary fish predators in estuaries are probably adult salmonids or juvenile salmonids which emigrate at older and larger sizes than others. They include cutthroat trout (*O. clarki*) or steelhead smolts preying on chum or pink salmon smolts. Outside estuaries, many large fish population reside just offshore and may consume large numbers of smolts. These fishes include Pacific hake (*Merluccius productus*), Pacific mackerel (*Scomber japonicus*), lingcod (*Ophiodon elongates*), spiny dogfish (*Squalus acanthias*), various rock fish, and lamprey (Beamish, Thomson et al. 1992; Pearcy 1992; Beamish and Neville 1995).

Wildland Fire

Wildland fires that are allowed to burn naturally in riparian or upland areas may benefit or harm aquatic species, depending on the degree of departure from natural fire regimes. Although most fires are small in size, large size fires increase the chances of adverse effects on aquatic species. Large fires that burn near the shores of streams and rivers can have biologically significant short-term effects. They include increased water temperatures, ash, nutrients, pH, sediment, toxic chemicals, and large woody debris (Buchwalter, Sandahl et al. 2004; Rinne 2004). Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams (Bisson, Rieman et al. 2003). As a result, many large fires burning near streams can result in fish kills with the survivors actively moving downstream to avoid poor water quality conditions (Greswell 1999; Rinne 2004). The patchy, mosaic pattern burned by fires provides a refuge for those fish and invertebrates that leave a burning area or simply spares some fish that were in a different location at the time of the fire (USFS 2000). Small fires or fires that burn entirely in upland areas also cause ash to enter rivers and increase smoke in the atmosphere, contributing to ammonia concentrations in rivers as the smoke adsorbs into the water (Greswell 1999).

The presence of ash also has indirect effects on aquatic species depending on the amount of ash entry into the water. All ESA-listed fishes rely on macroinvertebrates as a food source for at least a portion of their life histories. When small amounts of ash enter the water, there are usually no noticeable changes to the macroinvertebrate community or the water quality (Bowman and Minshall 2000). When significant amounts of ash are deposited into rivers, the macroinvertebrate community density and composition may be moderately to drastically reduced for a full year with long-term effects lasting 10 years or more (Buchwalter, Jenkins et al. 2003), (Minshall, Royer et al. 2001; Buchwalter, Sandahl et al. 2004). Larger fires can also indirectly affect fish by altering water quality. Ash and smoke contribute to elevated ammonium, nitrate, phosphorous, potassium, and pH, which can remain elevated for up to four months after forest fires (Buchwalter, Jenkins et al. 2003).

Many species have evolved in the presence of regular fires and have developed population-level mechanisms to withstand even the most intense fires (Greswell 1999). These same species have come to rely on fire’s disturbance to provide habitat heterogeneity. In the past century, the human population has increased dramatically, resulting in urban sprawl and the development of formerly remote locations. This condition has increased the urban/wildland interface. As a result, the threat of fires to personal property and people has increased, including the demand for protection of their safety and belongings. We expect listed fish species will be exposed to an increasing number of fires and fire fighting techniques over time. Currently, federal, state, and local resource agencies lack long-term monitoring data on the effects of wildland fire on listed Pacific salmonids and their habitats. Thus, we are unable to quantify the overall effects of wildland fire on the long-term survival of listed Pacific salmonids at this time.

Oceanographic Features and Climatic Variability

Oceanographic features of the action area may influence prey availability and habitat for Pacific salmonids. The action area includes important spawning and rearing grounds and physical and biological features essential to the conservation of listed Pacific salmonids - *i.e.*, water quality, prey, and passage conditions. Ocean conditions and climatic variability may affect salmonids in the action area.

The primary effects of the ocean on salmon productivity involve growth and survival of salmon. All salmon growth is completed in the ocean. According to Welch (1996), fish growth will not reach its maximum potential if food density (food available divided by ocean volume) is insufficient to provide the maximum daily ration. If this critical level of food is not exceeded, then the potential for the ocean to limit salmon growth exists.

The decline in salmon survival in Oregon and Washington since 1977 may be caused by poorly understood processes in the marine (as opposed to freshwater) environment (Welch 1996). Current findings also indicate that the primary control on salmon distribution is temperature. However, the upper thermal limit varies throughout the year (Welch 1996).

Naturally occurring climatic patterns, such as the Pacific Decadal Oscillation and the El Niño and La Niña events, are major causes of changing marine productivity. Recent studies have shown that long-term changes in climate affect oceanic structure and produce abrupt differences in salmon marine survival and returns (Mantua, Hare et al. 1997; Hare, Mantua et al. 1999). A major regime shift in the subarctic and California Current ecosystems during the late 1970s may have been a factor in reducing ocean survival of salmon in the Pacific Northwest and in increasing the marine survival in Alaska (Hare, Mantua et al. 1999). Fluctuations in mortality of salmon in the freshwater and marine environment have been shown to be almost equally significant sources of annual recruitment variability (Bradford 1997). These events and changes in ocean temperature may also influence salmonid abundance in the action area. In years when ocean conditions are cooler than usual, the majority of sockeye salmon returning to the

Fraser River do so via this route. However, when warmer conditions prevail, migration patterns shift to the north through the Johnstone Strait (Groot and Quinn 1987).

Climate Change

Anthropogenic climate change, caused by factors such as the continuing build-up of human-produced atmospheric carbon dioxide, is predicted to have major environmental impacts along the west coast of North America during the 21st century and beyond (IPCC 2001; CIG 2004). Warming trends continue in both water and air temperatures. Projections of the consequences of climate change include disruption of annual cycles of rain and snow, alteration of prevailing patterns of winds and ocean currents, and increases in sea levels (Glick 2005; Snover, Mote et al. 2005). Oceanographic models project a weakening of the thermohaline circulation resulting in a reduction of heat transport into high latitudes of Europe, an increase in the mass of the Antarctic ice sheet, and a decrease in the Greenland ice sheet (IPCC 2001). These changes, coupled with increased acidification of ocean waters, are expected to have substantial effects on marine productivity and food webs, including populations of salmon and other salmonid prey (Hard, Jones et al. 1992).

Climate change poses significant hazards to the survival and recovery of salmonids along the west coast. Changes in water temperature can alter migration timing, reduce growth, reduce the supply of available oxygen in the water, reduce insect availability as prey, and increase the susceptibility of fish to toxicants, parasites, and disease (Fresh, Casillas et al. 2005; NMFS 2007). Earlier spring runoff and lower summer flows make it difficult for returning adult salmon to negotiate obstacles (NMFS 2007). Excessively high levels of winter flooding can scour eggs from their nests in the stream beds and increase mortalities among overwintering juvenile salmon. The predicted increased winter flooding, decreased summer and fall stream flows, and elevated warm season temperatures in the streams and estuaries may further degrade conditions for salmon that are already stressed from habitat degradation. Although the impacts of global climate change are less clear in the ocean environment, early modeling efforts suggest that increased temperatures will likely increase ocean stratification. This stratification coincides with relatively poor ocean habitat for most Pacific Northwest salmon populations (IPCC 2001; CIG 2004).

We expect changing weather and oceanographic conditions may affect prey availability, temperature and water flow in habitat conditions, and growth for all 28 ESUs. Consequently, we expect the long-term survival and reproductive success for listed salmonids to be greatly affected by global climate change.

Anthropogenic Mortality Factors

In this section we address anthropogenic threats in the geographic regions across the action area. Among the threats discussed are the "four Hs": hatcheries, harvest, hydropower, and habitat. Prior to discussion of each geographic region, three major issues are highlighted: pesticide contamination, elevated water temperature, and loss of habitat/habitat connectivity. These three factors are the most relevant to the current analysis. To address these issues, we provide information on pesticide detections in the aquatic environment and highlight their background levels from past and ongoing anthropogenic activities. This information is pertinent to EPA's proposed registration of carbaryl, carbofuran, and methomyl in the U.S. and its territories. As water temperature plays such a strong role in salmonid distribution, we also provide a general discussion of anthropogenic temperature changes. Finally, we discuss the health of riparian systems and floodplain connectivity, as this habitat is vital to salmonid survival.

Baseline Pesticide Detections in Aquatic Environments

In the environmental baseline, we address pesticide detections reported as part of the U.S. Geological Survey (USGS) National Water-Quality Assessment Program's (NAWQA) national assessment (Gilliom, Barbash et al. 2006). We chose this approach for *Environmental Baseline* as the NAWQA studies present the same level of analysis for each area. Further, given the lack of reporting standards, we are unable to present a comprehensive basin-specific analysis of detections from other sources.

In the exposure section of the *Effects of the Proposed Action* we also present more recent unpublished data on the chemicals and degradates addressed in this Opinion from the NAWQA program and state databases maintained by California and Washington. As far as NMFS was able to ascertain, neither Oregon nor Idaho maintain publically available state-wide water quality databases. The California and Washington databases include some data from the NAWQA, but mostly the data are from more localized studies. Overall, data from those databases are relatively consistent in regards to pesticides addressed in this Opinion, with carbaryl generally being the most frequently quantifiable parent compound. Carbaryl and carbofuran were measured in concentrations ranging from 0.0001-33.5 μg/L. Methomyl generally was measured at slightly lower concentrations, ranging from 0.004-5.4 μg/L. Methomyl is also detected less frequently in some monitoring datasets, as it dissipates rapidly in aquatic systems, and non-targeted monitoring does not necessarily coincide with applications. Both 1-napthol (methomyl degradate) and 3-hydroxycarbofuran (carbofuran degradate) were detected in slightly lower concentrations, ranging from 0.0007-0.64 μg/L, than any of the parent compounds.

According to Gilliom *et al.*(2006), the distributions of the most prevalent pesticides in streams and ground water correlate with land use patterns and associated present or past pesticide use. When pesticides are released into the environment, they frequently end up as contaminants in aquatic environments. Depending on their physical properties some are rapidly transformed via chemical, photochemical, and biologically mediated reactions into other compounds, known as degradates. These degradates may become as prevalent as the parent pesticides depending on their rate of formation and their relative persistence.

National Water-Quality Assessment Program.

From 1992-2001, the USGS sampled water from 186 stream sites within 51 study units; bed-sediment samples from 1,052 stream sites, and fish from 700 stream sites across the continental U.S. Concentrations of pesticides were detected in streams and groundwater within most areas sampled with substantial agricultural or urban land uses. NAWQA results further detected at least one pesticide or degradate more than 90% of the time in water, in more than 80% in fish samples, and greater than 50% of bed-sediment samples from streams in watersheds with agricultural, urban, and mixed land use (Gilliom, Barbash et al. 2006).

About 40 pesticide compounds accounted for most detections in water, fish, or bed sediment. Twenty-four pesticides and one degradate were each detected in more than 10% of streams in agricultural, urban, or mixed land use settings. These 25 pesticide compounds include 11 herbicides used most heavily in agriculture during the study period (plus the atrazine degradate, deethylatrazine); 7 herbicides used extensively for non-agricultural purposes; and 6 insecticides used in both agricultural and urban settings. Three of those insecticides were chlorpyrifos, diazinon, and malathion. Thirteen organochlorine pesticide compounds, including historically used parent pesticides and their degradates and by-products, were each found in more than 10% of fish or bed-sediment samples from streams draining watersheds with either agricultural, urban, or mixed land use (Gilliom, Barbash et al. 2006).

Additionally, more frequent detections and higher concentrations of insecticides occur in sampled urban streams (Gilliom, Barbash et al. 2006). Diazinon, chlorpyrifos, carbaryl, and malathion nationally ranked 2nd, 4th, 8th, and 15th among pesticides in frequencies of outdoor applications for home- and garden use in 1992 (Whitmore, Kelly et al. 1992). These same insecticides accounted for the most insecticide detections in urban streams. Diazinon and carbaryl were the most frequently detected and were found at frequencies and levels comparable to those for the common herbicides. Historically used insecticides were also found most frequently in fish and bed sediment from urban streams. The highest detection frequencies were for chlordane compounds, dichloro-diphenyl-trichloroethane (DDT) compounds, and dieldrin. Urban streams also had the highest concentrations of total chlordane and dieldrin in both sediment and fish tissue. Chlordane and aldrin were widely used for termite control until the mid-to-late 1980s. Their agricultural uses were restricted during the 1970s.

Chlorpyrifos and diazinon were commonly used in agricultural and urban areas from 1992-2001 and prior to the sampling period. About 13 million lbs of chlorpyrifos and about 1 million lbs of diazinon were applied for agricultural use. Non-agricultural uses of chlorpyrifos and diazinon totaled about 5 million and 4 million lbs per year in 2001, respectively (Gilliom, Barbash et al. 2006). For both insecticides, concentrations in most urban streams were higher than in most agricultural streams, and were similar to those found in agricultural areas with the greatest intensities of use. Diazinon and chlorpyrifos were detected about 75% and 30% of the time in urban streams, respectively (Gilliom, Barbash et al. 2006). NMFS (2008) determined that current use of chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 listed salmonid ESUs. NMFS provided EPA with reasonable and prudent alternatives (RPAs), including buffers and vegetative strips, to reduce pesticide exposure to listed salmon. Until the EPA implements the RPAs, we must assume current exposure will continue.

Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (Gilliom, Barbash et al. 2006). Mixtures result from the use of different pesticides for multiple purposes within a watershed or groundwater recharge area. Pesticides generally occur more often in natural waterbodies as mixtures than as individual compounds. Mixtures of pesticides were detected more often in streams than in ground water and at relatively similar frequencies in streams draining areas of agricultural, urban, and mixed land use. More than 90% of the time, water from streams in these developed land use settings had detections of two or more pesticides or degradates. About 70% and 20% of the time, streams had five or more and ten or more pesticides or degradates, respectively (Gilliom, Barbash et al. 2006). Fish experiencing coincident exposure to multiple pesticides may also experience additive and synergistic effects. If the effects on a biological endpoint from concurrent exposure to multiple pesticides can be predicted by adding the potency of the pesticides involved, the effects are said to be additive. If, however, the response to a mixture lead to a greater than expected effect on the endpoint, and the pesticides within the mixture enhance the toxicity of one another, the effects are characterized as synergistic. These effects are of particular concern when the pesticides share a mode of action. Carbaryl, carbofuran, and methomyl are all AChE inhibitors. In California, there are 61 pesticides that inhibit AChE approved for use (CDPR 2007). According to CDPR, the amount of these chemicals used has decreased (Table 26). However, some AChE a.i.s – such as bensulide and naled – are increasing in use (CDPR 2007). While the trend indicates decreased reliance on these products, we note that their current use remains significant.

Table 26. Use figures for AChE inhibiting pesticides in California (CDPR 2007)

	1996	2006
lbs a.i. applied	15,473,843	6,857,530
Acres treated (agriculture use only)	11,720,058	5,729,958

Mixtures of organochlorine pesticide compounds were also common in fish-tissue samples from most streams. About 90% of fish samples collected from urban steams contained two or more pesticide compounds and 33% contained 10 or more pesticides. Similarly, 75% of fish samples from streams draining watersheds with agricultural and mixed land use contained 2 or more pesticide compounds and 10% had 10 or more compounds (Gilliom, Barbash et al. 2006).

NAWQA analysis of all detections indicates that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams (Gilliom, Barbash et al. 2006). The number of unique mixtures varied with land use. Mixtures of the most often detected individual pesticides include the herbicides atrazine (and its degradate deethylatrazine), metolachlor, simazine, and prometon. Each herbicide occurred in more than 30% of all mixtures found in agricultural and urban uses in streams. Also present in more than 30% of the mixtures were cyanazine, alachlor, metribuzin, and trifluralin in agricultural streams. Dacthal and the insecticides diazinon, chlorpyrifos, carbaryl, and malathion were also present in urban streams. Carbaryl occurred in at least 50% of urban streams. In 15% of urban streams carbaryl concentration was over 0.1µg/L (Gilliom, Barbash et al. 2006). Insecticides are typical constituents in environmental mixtures and are commonly found in both agriculatural and urban streams.

The numbers of unique mixtures of organochlorine pesticide compounds found in whole-fish tissue samples were greater in urban streams than in streams from agricultural or mixed land use watersheds. About 1,400 unique 5-compound mixtures were found in fish from urban steams compared to fewer than 800 unique 5-compound mixtures detected in fish from agricultural and mixed land use steams. The relative contributions of most organochlorine compounds to mixtures in fish were about the same for urban and agricultural streams.

More than half of all agricultural streams sampled and more than three-quarters of all urban streams had concentrations of pesticides in water that exceeded one or more benchmarks for aquatic life. Aquatic life criteria are EPA water-quality guidelines for protection of aquatic life. Exceedance of an aquatic life benchmark level indicates a strong probability that aquatic species are being adversely affected. However, aquatic species may also be affected at levels below criteria. Finally, organochlorine pesticides that were discontinued 15 to 30 years ago still exceeded benchmarks for aquatic life and fish-eating wildlife in bed sediment or fish-tissue samples from many streams.

National Pollutant Discharge Elimination System

Pollution originating from a discrete location such as a pipe discharge or wastewater treatment outfall is known as a point source. Point sources of pollution require a National Pollutant Discharge Elimination System (NPDES) permit. These permits are issued for aquaculture, concentrated animal feeding operations, industrial wastewater treatment plants, biosolids (sewer/sludge), pre-treatment and stormwater overflows. The EPA administers the NPDES permit program and states certify that NPDES permit holders comply with state water quality standards. Nonpoint source discharges do not originate from discrete points; thus, nonpoint sources are difficult to identify, quantify, and are not regulated. Examples of nonpoint source pollution include, but are not limited to, urban runoff from impervious surfaces, areas of fertilizer and pesticide application, and manure.

According to EPA's database of NPDES permits, about 243 NPDES permits are co-located with listed Pacific salmonids in California. Collectively, the total number of EPA-recorded NPDES permits in Idaho, Oregon, and Washington, that are co-located with listed Pacific salmonids is 1,978. See ESU Figures in the *Status of Listed Resources* section for NPDES permits co-located within listed salmonid ESUs within the states of

California, Idaho, Oregon, and Washington.

On November 27, 2006, EPA issued a final rule which exempted pesticides from the NPDES permit process, provided that application was approved under FIFRA. The NPDES permits, then, do not include any point source application of pesticides to waterways in accordance with FIFRA labels. This rule was vacated by the courts on January 7, 2009 (National Cotton Council v. EPA, 553 F.3d 927 (6^{th} Cir. 2009)).

Baseline Water Temperature- Clean Water Act

Elevated temperature is considered a water pollutant in most states with approved Water Quality Standards under the federal Clean Water Act (CWA) of 1972. As per the CWA, states periodically prepare a list of all surface waters in the state for which beneficial uses - such as drinking, recreation, aquatic habitat, and industrial use – are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that do not meet state surface water quality standards, and are not expected to improve within the next two years. This process is in accordance with section 303(d) of the CWA. There are five categories a waterway can be classified under as per section 303(d):

- Category 1: Meets tested water quality standards;
- Category 2: Some evidence of a water quality problem, action not yet required;
- Category 3: Insufficient data;
- Category 4: Polluted waterway with solution being implemented; and
- Category 5: Polluted waterway, action is required

Water bodies listed under Category 5 are those that are considered impaired or threatened by pollution. The “303(d) list” is generally considered synonymous with the Category 5 waters, and will be treated as such within this Opinion.

Each state has separate and different 303(d) listing criteria and processes. Generally a water body is listed separately for each standard it exceeds, so it may appear on the list more than once. If a water body is not on the 303(d) list, it is not necessarily contaminant-free; rather it may not have been tested. Therefore, the 303(d) list is a minimum list for the each state regarding polluted water bodies by parameter.

After states develop their lists of impaired waters, they are required to prioritize and submit their lists to EPA for review and approval. States are expected to identify high priority waters targeted for Total Maximum Daily Load (TMDL) development within two years of the 303(d) listing process. A TMDL includes a plan for reducing contaminant loading and is required for all impaired waterways. Each state also establishes a priority ranking for the development of TMDLs for such waters, considering the severity of the pollution and the uses to be made of such waters.

Federal non-priority water quality standards have been established for carbaryl and carbofuran, but not for methomyl. The California 303(d) list includes a 49 mile section of the Colusa Basin Drain that exceeds carbofuran standards (Category 5). Several areas in Washington and Oregon have been listed under Category 2 for carbaryl and carbofuran. They include: Willapa Bay, WA (carbaryl, 18 separate listings); Grays Harbor County Drainage Ditch #1, WA (carbaryl); Pacific County Drainage Ditch #1, WA (carbaryl); North River, WA (carbaryl); Palix River, WA (carbaryl); Johnson Creek, OR (carbaryl & carbofuran, 23.7 river miles); Beaverton Creek, OR (carbaryl, 9.8 river miles); Tualatin River, OR (carbaryl, 44.7 river miles); and Mill Creek, OR (carbofuran, 25.7 river miles). In addition to specific compounds, water bodies are listed as impaired due to "pesticides" as a general category. We did not consider these waterways as there was no way to tell what compounds were present.

Temperature is significant for the health of aquatic life. Water temperatures affect the distribution, health, and survival of native cold-blooded salmonids in the Pacific Northwest. These fish will experience adverse health effects when exposed to temperatures outside their optimal range. For listed Pacific salmonids, water temperature tolerance varies between species and life stages. Optimal temperatures for rearing salmonids range from 10ºC and 16ºC. In general, the increased exposure to stressful water temperatures and the reduction of suitable habitat caused by drought conditions reduce the abundance of salmon. Warm temperatures can reduce fecundity, increase egg survival, retard growth of fry and smolts, reduce rearing densities, increase susceptibility to disease, decrease the ability of young salmon and trout to compete with other species for food, and to avoid predation (Spence, Lomnicky et al. 1996; McCullough 1999). Migrating adult salmonids and upstream migration can be delayed by excessively warm stream temperatures. Excessive stream temperatures may also negatively affect incubating and rearing salmonids (Gregory and Bisson 1997).

Sublethal temperatures (above 24ºC) could be detrimental to salmon by increasing susceptibility to disease (Colgrove and Wood 1966) or elevating metabolic demand (Brett 1995). Substantial research demonstrates that many fish diseases become more virulent at temperatures over 15.6ºC (McCullough 1999). Due to the sensitivity of salmonids to temperature, states have established lower temperature thresholds for salmonid habitat as part of their water quality standards. A water body is listed for temperature on the 303(d) list if the 7-day average of the daily maximum temperatures (7-DADMax) exceeds the temperature threshold (Table 27).

Table 27. Washington State water temperature thresholds for salmonid habitat. These temperatures are representative of limits set by California, Idaho, and Oregon (WSDE 2006).

Category	Highest 7-DADMax
Salmon and Trout Spawning	13°C (55.4°F)
Core Summer Salmonid Habitat	16°C (60.8°F)
Salmonid Spawning, Rearing, and Migration	17.5°C (63.5°F)
Salmonid Rearing and Migration Only	17.5°C (63.5°F)

Water bodies that are not designated salmonid habitat are also listed if they have a one-day maximum over a given background temperature. Using publicly available GIS layers, we determined the number of km on the 303(d) list for exceeding temperature thresholds within the boundaries of each ESU (Table 28). Because the 303(d) list is limited to the subset of rivers tested, the chart values should be regarded aslower-end estimates.

While some ESU ranges do not contain any 303(d) rivers listed for temperature, others show considerable overlap. These comparisons demonstrate the relative significance of elevated temperature among ESUs. Increased water temperature may result in wastewater discharge, decreased water flow, minimal shading by riparian areas, and climatic variation.

Table 28. Number of kilometers of river, stream and estuaries included in state 303(d) lists due to temperature that are located within each salmonid ESU. Data was taken from the most recent GIS layers available from state water quality assessments reports*

Species	ESU	California	Oregon	Washington	Idaho	Total
Chinook Salmon	California Coastal	39.3	–	–	–	39.3
	Central Valley Spring - Run	0.0	–	–	–	0.0
	Lower Columbia River	–	56.6	229.8	–	286.4
	Upper Columbia River Spring - Run	–	–	254.6	–	254.6
	Puget Sound	–	–	705.0	–	705.0
	Sacramento River Winter - Run	0.0	–	–	–	0.0
	Snake River Fall - Run	–	610.1	246.6	400.2	1,256.9
	Snake River Spring / Summer - Run	–	809.3	243.2	543.8	1,596.3
	Upper Williamette River	–	2,468.0	–	–	2,468.0
Chum Salmon	Columbia River	–	56.6	225.0	–	281.6
	Hood Canal Summer - Run	–	–	90.1	–	90.1
Coho Salmon	Central California Coast	39.3	–	–	–	39.3
	Lower Columbia River	–	291.9	233.5	–	525.4
	Southern Oregon and Northern California Coast	1,416.2	1,833.0	–	–	3,249.2
	Oregon Coast	–	3,715.8	–	–	3,715.8
Sockeye Salmon	Ozette Lake	–	–	4.8	–	4.8
	Snake River	–	–	–	0.0	0.0
Steelhead	Central California Coast	0.0	–	–	–	0.0
	California Central Valley	0.0	–	–	–	0.0
	Lower Columbia River	–	201.2	169.3	–	370.5
	Middle Columbia River	–	3,518.5	386.2	–	3,904.7
	Northern California	39.3	–	–	–	39.3
	Puget Sound	–	–	704.9	–	704.9
	Snake River	–	990.7	246.6	737.6	1,974.9
	South-Central California Coast	0.0	–	–	–	0.0
	Southern California	0.0	–	–	–	0.0
	Upper Columbia River	–	–	282.3	–	282.3
	Upper Williamette River	–	1,668.0	–	–	1,668.0

*CA 2006, Oregon 2004/2006, Washington 2004, and Idaho 1998. (California EPA TMDL Program 2007b, Oregon Department of Environmental Quality 2007, Washington State Department of Ecology 2005, Idaho Department of Environmental Quality 2001).

Baseline Habitat Condition

Riparian zones are the areas of land adjacent to rivers and streams. These systems serve as the interface between the aquatic and terrestrial environments. Riparian vegetation is characterized by emergent aquatic plants and species that thrive on close proximity to water, such as willows. This vegetation maintains a healthy river system by reducing erosion, stabilizing main channels, and providing shade. Leaf litter that enters the river becomes an important source of nutrients for invertebrates (Bisson and Bilby 2001). Riparian zones are also the major source of large woody debris (LWD). When trees fall and enter the water, they become an important part of the ecosystem. The LWD alters the flow, creating the pools of slower moving water preferred by salmon (Bilby, Fransen et al. 2001). While not necessary for pool formation, LWD is associated with around 80% of pools in northern California, Washington, and the Idaho pan-handle (Bilby and Bisson 2001).

Bilby and Bisson (2001) discuss several studies that associate increased LWD with increased pools, and both pools and LWD with salmonid productivity. Their review also includes documented decreases in salmonid productivity following the removal of LWD. Other benefits of LWD include deeper pools, increased sediment retention, and channel stabilization.

Floodplains are relatively flat areas adjacent to larger streams and rivers. They allow for the lateral movement of the main channel and provide storage for floodwaters during periods of high flow. Water stored in the floodplain is later released during periods of low flow. This process ensures adequate flows for salmonids during the summer months, and reduces the possibility of high-energy flood events destroying salmonid redds (Smith 2005).

Periodic flooding of these areas creates habitat used by salmonids. Storms also wash sediment and LWD into the main stem river, often resulting in blockages. These blockages may force the water to take an alternate path and result in the formation of side channels and sloughs (Benda, Miller et al. 2001). Side channels and sloughs are important spawning and rearing habitat for salmonids. The degree to which these off-channel habitats are linked to the main channel via surface water connections is referred to as connectivity (PNERC 2002). As river height increases with heavier flows, more side channels form and connectivity increases.

Healthy riparian habitat and floodplain connectivity are vital for supporting a salmonid population. Once the area has been disturbed, it can take decades to recover (Smith 2005). Consequently, most land use practices cause some degree of impairment. Development leads to construction of levees and dikes, which isolate the mainstem river from the floodplain. Agricultural development and grazing in riparian areas also significantly change the landscape. Riparian areas managed for logging, or logged in the past, are often impaired by a change in species composition. Most areas in the northwest were historically dominated by conifers. Logging results in recruitment of deciduous trees, decreasing the quality of LWD in the rivers. Deciduous trees have smaller diameters than conifers; they decompose faster and are more likely to be displaced (Smith 2005).

Without a properly functioning riparian zone, salmonids contend with a number of limiting factors. They face reductions in quantity and quality of both off-channel and pool habitats. Also, when seasonal flows are not moderated, both higher and lower flow conditions exist. Higher flows can displace fish and destroy redds, while lower flows cut off access to parts of their habitat. Finally, decreased vegetation limits the available shade and cover, exposing individuals to higher temperatures and increased predation.

Geographic Regions

For a more fine scale analysis, we divided the action area into geographic regions: the Southwest Coast Region (California) and the Pacific Northwest Region (Idaho, Oregon, and Washington). The Pacific Northwest Region was further subdivided according to ecoregions or other natural features important to NMFS trust resources. Use of these geographic regions is consistent with previous NMFS consultations conducted at the national level (NMFS 2007). We summarize the principal anthropogenic factors occurring in the environment that influence the current status of listed species within each region. Table 29 provides a breakdown of these regions and includes the USGS subregions and accounting units for each region. It also provides a list of ESUs found in each accounting unit, as indicated by Federal Register listing notices.

Southwest Coast Region

The basins in this section occur in the State of California and the southern parts of the State of Oregon. Table 30 and Table 31 show land area in km² for each ESU /DPS located in the Southwest Coast Region.

Table 29. USGS Subregions and accounting units within the Northwest and Southwest Regions, along with ESUs present within the area (Seaber, Kapinos et al. 1987).

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU
Pacific Northwest: Columbia River Basin	Upper Columbia River Basin	—	WA	170200	Upper Columbia Spring-run Chinook; Upper Columbia Steelhead; Middle Columbia Steelhead
	Yakima River Basin	—	WA	170300	Middle Columbia Steelhead
	Lower Snake River Basin	Lower Snake River Basin	ID, OR, WA	170601	Snake River Steelhead; Snake River Spring/Summer-run Chinook; Snake River Fall-run Chinook; Snake River Sockeye
		Salmon River Basin	ID	170602	Snake River Steelhead; Snake River Spring/Summer - Run Chinook; Snake River Fall - Run Chinook; Snake River Sockeye
		Clearwater River Basin	ID, WA	170603	Snake River Steelhead; Snake River Fall - Run Chinook
	Middle Columbia River Basin	Middle Columbia River Basin	OR, WA	170701	Middle Columbia Steelhead; Lower Columbia Chinook; Columbia Chum; Lower Columbia Coho
		John Day River Basin	OR	170702	Middle Columbia Steelhead
		Deschutes River Basin	OR	170703	Middle Columbia Steelhead
	Lower Columbia River Basin	—	OR, WA	170800	Lower Columbia Chinook; Columbia Chum; Lower Columbia Steelhead; Lower Columbia Coho

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU
	Willamette River Basin	—	OR	170900	Upper Willamette Chinook; Upper Willamette Steelhead; Lower Columbia Chinook; Lower Columbia Steelhead; Lower Columbia Coho
Pacific Northwest: Coastal Drainages	Oregon-Washington Coastal Basin	Washington Coastal	WA	171001	Ozette Lake Sockeye
		Northern Oregon Coastal	OR	171002	Oregon Coast Coho
		Southern Oregon Coastal	OR	171003	Oregon Coast Coho; Southern Oregon and Northern California Coast Coho
Pacific Northwest: Puget Sound	Puget Sound	—	WA	171100	Puget Sound Chinook; Hood Canal Summer -Run Chum; Puget Sound Steelhead
Southwest Coast	Klamath-Northern California Coastal	Northern California Coastal	CA	180101	Southern Oregon and Northern California Coast Coho; California Coastal Chinook; Northern California Steelhead; Central California Coast Steelhead; Central California Coast Coho
		Klamath River Basin	CA, OR	180102	Southern Oregon and Northern California Coast Coho
	Sacramento River Basin	Lower Sacramento River Basin	CA	180201	Central Valley Spring-run Chinook; California Central Valley Steelhead; Sacramento River Winter-run Chinook
	San Joaquin River Basin	—	CA	180400	California Central Valley Steelhead
	San Francisco Bay	—	CA	180500	Central California Coast Steelhead; Southern Oregon and Northern California Coast Coho; Central California Coast Coho; Sacramento River Winter-run Chinook

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU
		—	CA	180600	Central California Coast Steelhead; Southern Oregon and Northern California Coast Coho; South-Central California Coast Steelhead; Southern California Steelhead; Central California Coast Coho; Sacramento River Winter-run Chinook
	Southern California Coastal	Ventura-San Gabriel Coastal	CA	180701	Southern California Steelhead
		Laguna-San Diego Coastal	CA	180703	Southern California Steelhead

Table 30. Area of land use categories within the range Chinook and Coho Salmon ESUs in km². Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type	code	Chinook Salmon			Coho Salmon	
		CA Coastal	Central Valley	Sacramento River	Central CA Coast	So. Oregon and No. CA
Open Water	11	128	346	0	157	197
Perennial Snow/Ice	12	0	0	12	0	11
Developed, Open Space	21	826	1,150	16	629	1,384
Developed, Low Intensity	22	137	578	313	171	225
Developed, Medium Intensity	23	95	567	0	138	92
Developed, High Intensity	24	10	135	313	30	23
Barren Land	31	70	158	40	23	261
Deciduous Forest	41	850	664	7	208	1,057
Evergreen Forest	42	10,700	3,761	1	4,752	28,080
Mixed Forest	43	1,554	479	51	922	2,426
Shrub/Scrub	52	3,801	3,203	0	1,620	8,864
Herbaceous	71	2,114	6,317	12	1,646	2,708
Hay/Pasture	81	183	769	11	6	736
Cultivated Crops	82	212	5,110	0	233	454
Woody Wetlands	90	42	191	0	25	130
Emergent Herbaceous Wetlands	95	18	553	18	13	50
TOTAL (inc. open water)		20,740	23,982	792	10,572	46,697
TOTAL (w/o open water)		20,612	23,636	792	10,415	46,499

Table 31. Area of Land Use Categories within the Range of Steelhead Trout DPSs (km²). Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type	code	Steelhead				
		Central CA Coast	CA Central Valley	Northern CA	South-Central CA Coast	Southern CA
Open Water	11	1,406	409	106	127	86
Perennial Snow/Ice	12	0	0	0	0	0
Developed, Open Space	21	1,224	1,431	610	1,019	685
Developed, Low Intensity	22	876	693	50	247	364
Developed, Medium Intensity	23	1,223	744	32	168	262
Developed, High Intensity	24	327	181	3	23	12
Barren Land	31	26	202	63	303	62
Deciduous Forest	41	179	751	763	1	0
Evergreen Forest	42	2,506	3,990	9,790	1,721	835
Mixed Forest	43	2,086	598	1,159	1,925	897
Shrub/Scrub	52	2,253	3,745	2,878	4,952	4,370
Herbacous	71	3,588	9,435	1,478	6,194	1,516
Hay/Pasture	81	36	1,671	179	203	141
Cultivated Crops	82	486	9,054	14	1,297	653
Woody Wetlands	90	36	248	32	93	35
Emergent Herbacous Wetlands	95	392	450	17	73	35
TOTAL (inc. open water)		16,645	33,601	17,173	18,345	9,954
TOTAL (w/o open water)		15,240	33,193	17,067	18,218	9,868

Select watersheds described herein characterize the past, present, and future human activities and their impacts on the area. The Southwest Coast region encompasses all Pacific Coast rivers south of Cape Blanco, Oregon through southern California. NMFS has identified the Cape Blanco area as an ESU/DPS biogeographic boundary for Chinook and coho salmon, and steelhead based on strong genetic, life history, ecological and habitat differences north and south of this landmark. Major rivers contained in this grouping of watersheds are the Sacramento, San Joaquin, Salinas, Klamath, Russian, Santa Ana, and Santa Margarita Rivers (Table 32).

Table 32. Select rivers in the southwest coast region (Carter and Resh 2005).

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Rogue River	211	5,154	CS, PB	38	10,065	23 (14)	11
Klamath River	287	15,679	PB, B/R, CS	33	17,693	48 (30)	41
Eel River	200	3,651	PB	52	7,416	25 (15)	12
Russian River	110	1,439	PB	41	2,331	41 (20)	43
Sacramento River	400	27,850	PB, CS, B/R	35	23,202	69 (29)	>50 T & E spp.
San Joaquin River	348	83,409	PB, CS	49	4,662	63	>50 T & E spp.
Salinas River	179	4,241	PB	14	448	36 (16)	42 T & E spp.
Santa Ana River	110	2,438	PB	13	60	45 (9)	54
Santa Margarita River	27	1,896	LC, PB	49.5	42	17 (6)	52

* Physiographic Provinces: PB = Pacific Border, CS = Cascades-Sierra Nevada Range, B/R = Basin & Range.

Land Use

Forest and vacant land are the dominant land uses in the northern basins. Grass, shrubland, and urban uses are the dominant land uses in the southern basins (Table 33). Overall, the most developed watersheds are the Santa Ana, Russian, and Santa Margarita rivers. The Santa Ana watershed encompasses portions of San Bernardino, Los Angeles, Riverside, and Orange counties. About 50% of the coastal subbasin in the Santa Ana watershed is dominated by urban land uses and the population density is about 1,500 people per square mile. When steep and undevelopable lands are excluded from this area, the population density in the watershed is about 3,000 people per square mile. However, the most densely populated portion of the basin is near the City of Santa Ana. Here, the population density reaches 20,000 people per square mile (Burton, Izbicki et al. 1998; Belitz, Hamlin et al. 2004). The basin is home to nearly 5 million people. However, this population is projected to increase two-fold in the next 50 years (Burton, Izbicki et al. 1998; Belitz, Hamlin et al. 2004).

Table 33. Land uses and population density in several southwest coast watersheds (Carter and Resh 2005).

Watershed	Land Use Categories (Percent)				Density (people/mi^2)
	Agriculture	Forest	Urban	Other	
Rogue River	6	83	<1	9 grass & shrub	32
Klamath River	6	66	<1	24 grass, shrub, wetland	5
Eel River	2	65	<1	31 grass & shrub	9
Russian River	14	50	3	31 (23 grassland)	162
Sacramento River	15	49	2	30 grass & shrub	61
San Joaquin River	30	27	2	36 grass & shrub	76
Salinas River	13	17	1	65 (49 grassland)	26
Santa Ana River	11	57	32	---	865
Santa Margarita River	12	11	3	71 grass & shrub	135

As a watershed becomes urbanized, population increases and changes occur in stream habitat, water chemistry, and the biota (plants and animals) that live there. The most obvious effect of urbanization is the loss of natural vegetation which results in an increase in impervious cover and dramatic changes to the natural hydrology of urban streams. Urbanization generally results in land clearing, soil compaction, modification and/or loss of riparian buffers, and modifications to natural drainage features (Richter 2002). The increased impervious cover in urban areas leads to increased volumes of runoff, increased peak flows and flow duration, and greater stream velocity during storm events.

Runoff from urban areas also contains all the chemical pollutants from automobile traffic and roads as well as those from industrial sources and residential use. Urban runoff is also typically warmer than receiving waters and can significantly increase temperatures in small urban streams. Warm stream water is detrimental to native aquatic life resident fish and the rearing and spawning needs of anadromous fish. Wastewater treatment plants replace septic systems, resulting in point dischages of nutrients and other contaminants not removed in the processing. Additionally, some cities have combined sewer/stormwater overflows (CSOs) and older systems may discharge untreated sewage following heavy rainstorms. Wastewater treatment plant outfalls often discharge directly into the rivers containing salmonids. These urban nonpoint and point source discharges affect the water quality and quantity in basin surface waters.

In many basins, agriculture is the major water user and the major source of water pollution to surface waters. In 1990, nearly 95% of the water diverted from the San Joaquin River was diverted for agriculture. Additionally, 1.5% of the water was diverted for livestock (Carter and Resh 2005). The amount and extent of water withdrawals or diversions for agriculture impact streams and their inhabitants via reduced water flow/velocity and dissolved oxygen levels. For example, adequate water flow is required for migrating salmon along freshwater, estuarine, and marine environments in order to complete their life cycle. Low flow events may delay salmonid migration or lengthen fish presence in a particular water body until favorable flow conditions permit fish migration along the migratory corridor or into the open ocean.

Water diversions may also increase nutrient load, sediments (from bank erosion), and temperature. Flow management and climate changes have decreased the delivery of suspended particulate matter and fine sediment to the estuary. The conditions of the habitat (shade, woody debris, overhanging vegetation) whereby salmonids are constrained by low flows also may make them more or less vulnerable to predation, elevated temperatures, crowding, and disease. Water flow effects on salmonids may seriously impact adult migration and water quality conditions for spawning and rearing salmonids. High temperature may also result from the loss of vegetation along streams that used to shade the water and from new land uses (buildings and pavement) whereby rainfall picks up heat before it runs off into the stream.

Currently, California has over 500 water bodies on its 303(d) list (Wu 2000). The 2006 list includes 779 stream segments, rivers, lakes, and estuaries and 12 pollutant categories (CEPA 2007). Pollutants represented on the list include pesticides, metals, sediments, nutrients or low dissolved oxygen, temperature, bacteria and pathogens, and trash or debris. There are 2,237 water body/pollutant listings; a water body is listed separately for each pollutant detected (CEPA 2007). The 2006 303(d) list identifies water bodies listed due to the presence of specific pollutants, including carbofuran and elevated temperature

(Table 34). See species ESU/DPS maps for NPDES permits and 303(d) waters co-located within listed salmonid ESUs/DPSs in California.

Table 34. California's 2006 Section 303(d) List of Water Quality Limited Segments: segments listed for exceeding temperature and carbofuran limits (CEPA 2007).

Pollutant	Estuary Acres Affected	River / Stream Miles Affected	# Water Bodies
Temperature	-	16,907.2	41
Carbofuran	-	49	1

Estuary systems of the region are consistently exposed to anthropogenic pressures stemming from high human density sources. For example, the largest west coast estuary is the San Francisco Estuary. This water body provides drinking water to 23 million people, irrigates 4.5 million acres of farmland, and drains roughly 40% of California's land area. As a result of high use, many environmental measures of the San Francisco Estuary are poor. Water quality suffers from high phosphorus and nitrogen loads, primarily from agricultural, sewage, and storm water runoff. Water clarity is also compromised. Sediments from urban runoff and historical activities contain high levels of contaminants. They include pesticides,polychlorinated biphenyls (PCBs), nickel, selenium, cadmium, , mercury, copper, and silver. Specific pesticides include pyrethroids, malathion, carbaryl, and diazinon. Other pollutants include DDT and polynuclear aromatic hydrocarbons (PAHs).

Other wastes are also discharged into San Francisco Bay. Approximately 150 industries discharge wastewater into the bay. Discharge of hot water from power plants and industrial sources may elevate temperatures and negatively affect aquatic life. Additionally, about 60 sewage treatment plants discharge treated effluent into the bay and elevate nutrient loads. However, since 1993, many of the point sources of pollution have been greatly reduced. Pollution from oil spills also occur due to refineries in the bay area. As these stressors persist in the marine environment, the estuary system will likely carry loads for future years, even with strict regulation. Gold mining has also reduced estuary depths in much of the region, causing drastic changes to habitat.

Large urban centers are foci for contaminants. Contaminant levels in surface waters near San Francisco, Oakland, and San Jose are highest. These areas are also where water clarity is at its worst. Some of the most persistent contaminants (PCBs, dioxins, DDT, etc.) are bioaccumulated by aquatic biota and can biomagnify in the food chain. Fish tissues contain high levels of PCB and mercury. Concentrations of PCB were 10 times above human health guidelines for consumption. Birds, some of which are endangered (clapper rail and least tern), have also concentrated these toxins.

Santa Ana Basin: NAWQA assessment

The Santa Ana watershed is the most heavily populated study site out of more than 50 assessment sites studied across the nation by the NAWQA Program. According to Belitz *et al.* (2004), treated wastewater effluent is the primary source of baseflow to the Santa Ana River. Secondary sources that influence peak river flows include stormwater runoff from urban, agricultural, and undeveloped lands (Belitz, Hamlin et al. 2004). Stormwater and agricultural runoff frequently contain pesticides, fertilizers, sediments, nutrients, pathogenic bacteria, and other chemical pollutants to waterways and degrade water quality. The above inputs have resulted in elevated concentrations of nitrates and pesticides in surface waters of the basin. Nitrates and pesticides were more frequently detected here than in other national NAWQA sites (Belitz, Hamlin et al. 2004).

Additionally, Belitz *et al.* (2004) found that pesticides and volatile organic compounds (VOCs) were frequently detected in surface and ground water in the Santa Ana Basin. Of the 103 pesticides and degradates routinely analyzed for in surface and ground water, 58 were detected. Pesticides included diuron, diazinon, carbaryl, chlorpyrifos, lindane, malathion, and chlorothalonil. Carbaryl was detected in 42% of urban samples, though it generally did not exceed the standard for protection of aquatic life (Belitz *et al.* 2004). Carbofuran was also detected, but did not exceed any water quality standards. Methomyl was tested for but not detected. Of the 85 VOCs routinely analyzed for, 49 were detected. VOCs included methyl *tert*-butyl ether (MTBE), chloroform, and trichloroethylene (TCE). Organochlorine compounds were also detected in bed sediment and fish tissue. Organochlorine concentrations were also higher at urban sites than at undeveloped sites in the Santa Ana Basin. Organochlorine compounds include DDT and its breakdown product diphenyl dicloroethylene (DDE), and chlordane. Other contaminants detected at high levels included trace elements such as lead, zinc, and arsenic. According to Belitz *et al.* (2004), the biological community in the basin is heavily altered as a result from these pollutants.

San Joaquin-Tulare Basin: NAWQA assessment

A study was conducted by the USGS in the mid-1990s on water quality within the San Joaquin-Tulare basins. USGS detected 49 of the 83 pesticides it tested for in the mainstem and three subbasins. Pesticides were detected in all but one of the 143 samples. The most common detections were of the herbicides simazine, dacthal, metolachlor, and EPTC (Eptam), and the insecticides diazinon and chlorpyrifos. Twenty-two pesticides were detected in 20% of the samples (Dubrovsky, Kratzer et al. 1998). Carbaryl and methomyl were detected in all three subbasins, despite land use differences. Carbaryl was detected in roughly 20% of samples from each subbasin, while methomyl detections ranged from 5% to 25%. Further, most samples contained mixtures of between 7 and 22 pesticides. Criteria for the protection of aquatic life were exceeded in 37% of samples of streams (Dubrovsky, Kratzer et al. 1998). Only seven pesticides exceeded their criteria: diuron, trifluralin, azinphos-methyl, carbaryl, chlorpyrifos, diazinon, and malathion. Forty percent of these exceedances were attributed solely to diazinon. However, criteria do not exist yet for over half of the detected compounds (Dubrovsky et al.1998).

Organochlorine insecticides in bed sediment and tissues of fish or clams were also detected. They include DDT and toxaphene. Levels at some sites were among the highest in the nation. Concentrations of trace elements in bed sediment generally were higher than concentrations found in other NAWQA study units (Dubrovsky, Kratzer et al. 1998).

Sacramento River Basin: NAWQA analysis

Another study conducted by the USGS from 1996-1998 within the Sacramento River Basin detected up to 24 out of 47 pesticides in surface waters (Domagalski 2000). Pesticides included thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Land use differences between sites are reflected in pesticide detections. Carbofuran was detected in 100% of samples from the agricultural site, but only 6.7% of urban samples (Domagalski 2000). Carbaryl, however, was detected in 100% of urban samples and 42.9% of agricultural samples. Some pesticides were detected at concentrations higher than criteria for the protection of aquatic life in the smaller streams, but were diluted to safer levels in the mainstem river. Intensive agricultural activities also impact water chemistry. In the Salinas River and in areas with intense agriculture use, water hardness, alkalinity, nutrients, and conductivity are also high.

Habitat Modification

The Central Valley area, including San Francisco Bay and the Sacramento and San Joaquin River Basins, has been drastically changed by development. Salmonid habitat has been reduced to 300 miles from historic estimates of 6,000 miles (CDFG 1993). In the San Joaquin Basin alone, the historic floodplain covered 1.5 million acres with 2 million acres of riparian vegetation (CDFG 1993). Roughly 5% of the Sacramento River Basin's riparian forests remain. Impacts of development include loss of LWD, increased bank erosion and bed scour, changes in sediment loadings, elevated stream temperature, and decreased base flow. Thus, lower quantity and quality of LWD and modified hydrology reduce and degrade salmonid rearing habitat.

The Klamath Basin in Northern California has been heavily modified as well. Water diversions have reduced spring flows to 10% of historical rates in the Shasta River, and dams block access to 22% of historical salmonid habitat. The Scott and Trinity Rivers have similar histories. Agricultural development has reduced riparian cover and diverted water for irrigation (NRC 2003). Riparian habitat has decreased due to extensive logging and grazing. Dams and water diversions are also common. These physical changes resulted in water temperatures too high to sustain salmonid populations. The Salmon River, however, is comparatively pristine; some reaches are designated as Wild and Scenic Rivers. The main cause of riparian loss in the Salmon River basin is likely wild fires – the effects of which have been exacerbated by salvage logging (NRC 2003).

Mining

Famous for the gold rush of the mid-1800s, California has a long history of mining. Extraction methods such as suction dredging, hydraulic mining, strip mining may cause water pollution problems. In 2004, California ranked top in the nation for non-fuel mineral production with 8.23% of total production (NMA 2007). Today, gold, silver, and iron ore comprise only 1% of the production value. Primary minerals include construction sand, gravel, cement, boron, and crushed stone. California is the only state to produce boron, rare-earth metals, and asbestos (NMA 2007).

California contains some 1,500 abandoned mines. Roughly 1% of these mines are suspected of discharging metal-rich waters into the basins. The Iron Metal Mine in the Sacramento Basin releases more than 1,100 lbs of copper and more than 770 lbs of zinc to the Keswick Reservoir below Shasta Dam. The Iron Metal Mine also released elevated levels of lead (Cain et al. 2000 in Carter and Resh 2005). Metal contamination reduces the biological productivity within a basin. Metal contamination can result in fish kills at high levels or sublethal effects at low levels. Sublethal effects include a reduction in feeding, overall activity levels, and growth. The Sacramento Basin and the San Francisco Bay watershed are two of the most heavily impacted basins within the state from mining activities. The basin drains some of the most productive mineral deposits in the region. Methylmercury contamination within San Francisco Bay, the result of 19^{th} century mining practices using mercury to amalgamate gold in the Sierra Nevada Mountains, remains a persistent problem today. Based on sediment cores, pre-mining concentrations were about five times lower than concentrations detected within San Francisco Bay today (Conaway, Squire et al. 2003).

Hydromodification Projects

Several of the rivers within the area have been modified by dams, water diversions, drainage systems for agriculture and drinking water, and some of the most drastic channelization projects in the nation (see species distribution maps). In all, there are about 1,400 dams within the State of California, more than 5,000 miles of levees, and more than 140 aqueducts (Mount 1995). In general, the southern basins have a warmer and drier climate and the more northern, coastal-influenced basins are cooler and wetter. About 75% of the runoff occurs in basins in the northern half of California, while 80% of the water demand is in the southern half. Two water diversion projects meet these demands—the federal Central Valley Project (CVP) and the California State Water Project (CSWP). The CVP is one of the world's largest water storage and transport systems. The CVP has more than 20 reservoirs and delivers about 7 million acre-ft per year to southern California. The CSWP has 20 major reservoirs and holds nearly 6 million acre-ft of water. The CSWP delivers about 3 million acre-ft of water for human use. Together, both diversions irrigate about 4 million acres of farmland and deliver drinking water to roughly 22 million residents.

Both the Sacramento and San Joaquin rivers are heavily modified, each with hundreds of dams. The Rogue, Russian, and Santa Ana rivers each have more than 50 dams, and the Eel, Salinas, and the Klamath Rivers have between 14 and 24 dams each. The Santa Margarita is considered one of the last free flowing rivers in coastal southern California. Nine dams occur in this watershed. All major tributaries of the San Joaquin River are impounded at least once and most have multiple dams or diversions. The Stanislaus River, a tributary of the San Joaquin River, has over 40 dams. As a result, the hydrograph of the San Joaquin River is seriously altered from its natural state. Alteration of the temperature and sediment transport regimes had profound influences on the biological community within the basin. These modifications generally result in a reduction of suitable habitat for native species and frequent increases in suitable habitat for nonnative species. The Friant Dam on the San Joaquin River is attributed with the extirpation of spring-run Chinook salmon within the basin. A run of the spring-run Chinook salmon once produced about 300,000 to 500,000 fish (Carter and Resh 2005).

Artificial Propagation

Anadromous fish hatcheries have existed in California since establishment of the McCloud River hatchery in 1872. There are nine state hatcheries: the Iron Gate (Klamath River), Mad River, Trinity (Trinity River), Feather (Feather River), Warm Springs (Russian River), Nimbus (American River), Mokelumne (Mokulumne River), and Merced (Merced River). The California Department of Fish and Game (CDFG) also manages artificial production programs on the Noyo and Eel rivers. The Coleman National Fish Hatchery, located on Battle Creek in the upper Sacramento River, is a federal hatchery operated by the USFWS. The USFWS also operates an artificial propagation program for Sacramento River winter run Chinook.

Of these, the Feather River, Nimbus, Mokelumne, and Merced River facilities comprise the Central Valley Hatcheries. Over the last ten years, the Central Valley Hatcheries have released over 30 million young salmon. State and the federal (Coleman hatchery) hatcheries work together to meet overall goals. State hatcheries are expected to release 18.6 million smolts in 2008 and Coleman is aiming for more than 12 million. There has been no significant change in hatchery practices over the year that would adversely affect the current year class of fish. A new program marking 25% of the 32 million Sacramento Fall-run Chinook smolts may provide data on hatchery fish contributions to the fisheries in the near future.

Commercial and Recreational Fishing

The region is home to many commercial fisheries. The largest in terms of total landings in 2006 were northern anchovy, Pacific sardine, Chinook salmon, sablefish, Dover sole, Pacific whiting, squid, red sea urchin, and Dungeness crab (CDFG 2007). Red abalone is also harvested. The commercial landings report does not include information on bycatch of listed salmonids (CDFG 2007). The first salmon cannery established along the west coast was located in the Sacramento River watershed in 1864. However, this cannery only operated for about two years because the sediment from hydraulic mining decimated the salmon runs in the basin (NRC 1996).

Alien Species

Plants and animals that are introduced into habitats in which they do not naturally occur are called non-native species. They are also known as non-indigenous, exotic, introduced, or invasive species, and have been known to affect ecosystems. Non-native species are introduced through infested stock for aquaculture and fishery enhancement, through ballast water discharge and from the pet and recreational fishing industries (http://biology.usgs.gov/s+t/noframe/x191.htm.). The Aquatic Nuisance Species (ANS) Task Force suggests that it is inevitable that cultured species will eventually escape confinement and enter U.S. waterways. Non-native species were cited as a contributing cause in the extinction of 27 species and 13 subspecies of North American fishes over the past 100 years (Miller, Williams et al. 1989). Wilcove, Rothstein et al. (1998) note that 25% of ESA listed fish are threatened by alien species. By competing with native species for food and habitat as well as preying on them, non-native species can reduce or eliminate populations of native species.

Surveys performed by CDFG state that at least 607 alien species are found in California coastal waterways (Foss *et al.* 2007). The majority of these species are representatives of four phyla: annelids (33%), arthropods (22%), chordates (13%), and mollusks (10%). Non-native chordate species are primarily fish and tunicates which inhabit fresh and brackish water habitats such as the Sacramento-San Joaquin Delta (Foss, Ode et al. 2007). The California Aquatic Invasive Species Management Plan (CAISMP) includes goals and strategies for reducing the introduction rate of new invasive species as well as removing those with established populations.

Atmospheric deposition

In 2002, chlopyrifos, diazinon, trifluralin, and other pesticides were detected in air samples collected from Sacramento, California (Majewski and Baston 2002).

Pesticide Reduction Programs

There are several measures in place in California that may reduce the levels of pesticides found in the aquatic environment beyond FIFRA label requirements. Monitoring of water resources is handled by California Environmental Protection Agency's Regional Water Boards. Each Regional Board makes water quality decisions for its region including setting standands and determining waste discharge requirements. The Central Valley Regional Water Quality Control Board (CVRWQCB) addresses issues in the Sacramento and San Joaquin River Basins. These river basins are characterized by crop land, specifically orchards, which historically rely heavily on organophosphates for pest control.

In 2003, the CVRWQCB adopted the Irrigated Lands Waiver Program (ILWP). Participation was required for all growers with irrigated lands that discharge waste which may degrade water quality. However, the ILWP allowed growers to select one of three methods for regulatory coverage (Markle, Kalman et al. 2005). These options included: 1) join a Coalition Group approved by the CVRWQCB, 2) file for an Individual Discharger Conditional Waiver, and 3) comply with zero discharge regulation (Markle, Kalman et al. 2005). Many growers opted to join a Coalition as the other options were more costly. Coalition Groups were charged with completing two reports – a Watershed Evaluation Report and a Monitoring and Reporting Plan. The Watershed Evaluation Report had to include information on crop patterns and pesticide/nutrient use, as well as mitigation measures that would prevent orchard run-off from impairing water quality. Similar programs are in development in other agricultural areas of California.

As a part of the Waiver program, the Central Valley Coalitions undertook monitoring of "agriculture dominated waterways". Some of the monitored waterways are small agricultural streams and sloughs that carry farm drainage to larger waterways. The coalition was also required to develop a management plan to address exceedance of State water quality standards. Currently, the Coalitions monitor toxcity to test organisms, stream parameters (*e.g.*, flow, temperature, etc.), nutrient levels, and pesticides used in the region, including diazinon and chlorpyrifos. Sampling diazinon exceedances within the Sacramento and Feather Rivers resulted in the development of a TMDL. The Coalitions were charged with developing and implementing management and monitoring plans to address the TMDL and reduce diazinon run-off.

The Coalition for Urban/Rural Environmental Stewardship (CURES) is a non-profit organization that was founded in 1997 to support educational efforts for agricultural and urban communities focusing on the proper and judicious use of pest control products. CURES educates growers on methods to decrease diazinon surface water contamination in the Sacramento River Basin. The organization has developed best-practice literature for pesticide use in both urban and agricultural settings (www.curesworks.org). CURES also works with California's Watershed Coalitions to standardize their Watershed Evaluation Reports and to keep the Coalitions informed. The organization has worked with local organizations, such as the California Dried Plum Board and the Almond Board of California, to address concerns about diazinon, pyrethroids, and sulfur. The CURES site discusses alternatives to oprganophosphate dormant spray applications. It lists pyrethroids and carbaryl as alternatives, but cautions that these compounds may impact non-target organisms. For example, carbaryl is highly toxic to honeybees, so bees must be removed from the area prior to application

In 2006, CDPR put limitations on dormant spay application of most insecticides in orchards, in part to adequately protect aquatic life in the Central Valley region. While the legislation was prompted by organophosphate use, limitations also apply to pyrethroids and carbamates.

The CDPR publishes voluntary interim measures for mitigating the potential impacts of pesticide useage to listed species. These measures are available online as county bulletins (http://www.cdpr.ca.gov/docs/endspec/colist.htm). Measures that apply to carbaryl, carbofuran, and methomyl use in salmonid habitat are:

- Do not use in currently occupied habitat
- Provide a 20 ft minimum strip of vegetation (on which pesticides should not be applied) along rivers, creeks, streams, wetlands, vernal pools and stock ponds, or on the downhill side of fields where runoff could occur. Prepare land around fields to contain runoff by proper leveling, etc. Contain as much water "on-site" as possible. The planting of legumes, or other cover crops for several rows adjacent to off-target water sites is recommended. Mix pesticides in areas not prone to runoff such as concrete mixing/loading pads, disked soil in flat terrain or graveled mix pads, or use a suitable method to contain spills and/or rinsate. Properly empty and triple-rinse pesticide

containers at time of use.

- Conduct irrigations efficiently to prevent excessive loss of irrigation waters through runoff. Schedule irrigations and pesticide applications to maximize the interval of time between the pesticide application and the first subsequent irrigation. Allow at least 24 hours between application of pesticides listed in this bulletin and any irrigation that results in surface runoff into natural waters. Time applications to allow sprays to dry prior to rain or sprinkler irrigations. Do not make aerial applications while irrigation water is on the field unless surface runoff is contained for 72 hours following the application.
- For sprayable or dust formulations: when the air is calm or moving away from habitat, commence applications on the side nearest the habitat and proceed away from the habitat. When air currents are moving toward habitat, do not make applications within 200 yards by air or 40 yards by ground upwind from occupied habitat. The county agricultural commissioner may reduce or waive buffer zones following a site inspection, if there is an adequate hedgerow, windbreak, riparian corridor or other physical barrier that substantially reduces the probability of drift.

Pacific Northwest Region

This region encompasses Idaho, Oregon, and Washington and includes parts of Nevada, Montana, Wyoming, and British Columbia. In this section we discuss three major areas that support salmonid populations within the action area. They include the Columbia River Basin and its tributaries, the Puget Sound Region, and the coastal drainages north of the Columbia River. Table 35, Table 36, and Table 37 show the types and areas of land use within each salmonid ESU/DPS.

Table 35. Area of land use categories within Chinook Salmon ESUs in km². Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type code		Chinook Salmon					
		Lower Columbia River	Upper Columbia River Spring Run	Puget Sound	Snake River Fall Run	Snake River Spring/ Summer Run	Upper Willamette River
Open Water	11	641	188	6,172	6,172	253	124
Perennial Snow/Ice	12	12	16	313	313	40	7
Developed, Open Space	21	649	203	1,601	1,601	328	632
Developed, Low Intensity	22	517	218	1,694	1,694	113	722
Developed, Medium Intensity	23	290	55	668	668	30	322
Developed, High Intensity	24	118	11	266	266	2	112
Barren Land	31	287	360	1,042	1,042	500	220
Deciduous Forest	41	551	21	999	999	10	248
Evergreen Forest	42	6,497	8,138	14,443	14,443	27,701	9,531
Mixed Forest	43	927	7	2,526	2,526	4	1,130
Shrub/Scrub	52	1,598	6,100	2,415	2,415	13,618	1,940
Herbaceous	71	520	1,737	957	957	11,053	801
Hay/Pasture	81	547	327	1,188	1,188	456	3,617
Cultivated Crops	82	278	636	258	258	3,860	2,355
Woody Wetlands	90	377	92	648	648	96	431
Emergent Herbaceous Wetlands	95	223	59	492	492	92	78
TOTAL (inc. open water)		14,031	18,168	35,683	35,683	58,157	22,269
TOTAL (w/o open water)		13,390	17,981	29,511	29,511	57,904	22,146

Table 36. Area of land use categories within chum and coho ESUs in km². Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type code		Chum Salmon		Coho Salmon	
		Columbia River	Hood Canal Summer Run	Lower Columbia River	Oregon Coast
Open Water	11	655	704	675	200
Perennial Snow/Ice	12	1	51	12	0
Developed, Open Space	21	605	134	708	1,107
Developed, Low Intensity	22	463	77	563	163
Developed, Medium Intensity	23	258	20	305	49
Developed, High Intensity	24	110	6	124	20
Barren Land	31	247	166	290	467
Deciduous Forest	41	548	97	575	418
Evergreen Forest	42	4,294	2,477	8,487	14,943
Mixed Forest	43	892	200	999	4,126
Shrub/Scrub	52	1,353	299	1,982	3,134
Herbaceous	71	526	133	600	1,478
Hay/Pasture	81	533	64	680	860
Cultivated Crops	82	213	2	348	64
Woody Wetlands	90	363	61	386	263
Emergent Herbaceous Wetlands	95	222	56	225	226
TOTAL (inc. open water)		11,284	4,548	16,959	27,520
TOTAL (w/o open water)		10,628	3,843	16,284	27,320

Table 37. Area of land use categories within sockeye ESUs and steelhead DPSs in km². Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type	code	Sockeye Salmon		Steelhead					
		Ozette Lake	Snake River	Lower Columbia River	Middle Columbia River	Puget Sound	Snake River	Upper Columbia River	Upper Willamette River
Open Water	11	30	19	250	575	6,172	285	359	62
Perennial Snow/Ice	12	0	18	12	13	313	42	16	0
Developed, Open Space	21	1	3	518	1,276	1,601	515	343	382
Developed, Low Intensity	22	0	2	506	627	1,694	144	294	513
Developed, Medium Intensity	23	0	0	287	192	668	40	80	231
Developed, High Intensity	24	0	0	116	25	266	3	13	75
Barren Land	31	2	9	174	183	1,042	504	361	77
Deciduous Forest	41	3	0	382	54	999	35	25	171
Evergreen Forest	42	158	755	7,023	18,347	14,443	39,556	8,223	4,133
Mixed Forest	43	3	0	611	41	2,526	17	7	791
Shrub/Scrub	52	14	185	1,589	32,089	2,415	15,644	9,351	994
Herbaceous	71	8	269	398	2,752	957	12,361	1,823	519
Hay/Pasture	81	0	12	605	863	1,188	463	448	2,529
Cultivated Crops	82	0	1	322	11,908	258	6,227	3,236	1,844
Woody Wetlands	90	8	16	244	217	648	116	109	292
Emergent Herbaceous Wetlands	95	1	34	93	291	492	111	81	43
TOTAL (inc. open water)		228	1,323	13,128	69,453	35,683	76,061	24,771	12,655
TOTAL (w/o open water)		199	1,304	12,878	68,878	29,511	75,777	24,411	12,593

Columbia River Basin

The most notable basin within the region is the Columbia River. The Columbia River is the largest river in the Pacific Northwest and the fourth largest river in terms of average discharge in the U.S. The Columbia River drains over 258,000 square miles, and is the sixth largest in terms of drainage area. Major tributaries include the Snake, Willamette, Salmon, Flathead, and Yakima rivers. Smaller rivers include the Owyhee, Grande Ronde, Clearwater, Spokane, Methow, Cowlitz, and the John Day Rivers (see Table 38 for a description of select Columbia River tributaries). The Snake River is the largest tributary at more than 1,000 miles long. The headwaters of the Snake River originate in Yellowstone National Park, Wyoming. The second largest tributary is the Willamette River in Oregon (Kammerer 1990; Hinck, Schmitt et al. 2004). The Willamette River is also the 19th largest river in the nation in terms of average annual discharge (Kammerer 1990). The basins drain portions of the Rocky Mountains, Bitteroot Range, and the Cascade Range.

Table 38. Select tributaries of the Columbia River (Carter and Resh 2005)

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Snake/Salmon rivers	870	108,495	CU, NR, MR, B/R	14	55,267	39 (19)	5 fish (4 T, 1 E), 6 (1 T, 5 E) snails, 1 plant (T)
Yakima River	214	6,139	CS, CU	7	3,602	50	2 fish (T)
Willamette River	143	11,478	CS, PB	60	32,384	61 (~31)	5 fish (4 T, 1 E),

* Physiographic Provinces: CU = Columbia-Snake River Plateaus, NR = Northern Rocky Mountains, MR = Middle Rocky Mountains, B/R = Basin & Range, CS = Cascade-Sierra Mountains, PB = Pacific Border

The Columbia river and estuary were once home to more than 200 distinct runs of Pacific salmon and steelhead with unique adaptations to local environments within a tributary (Stanford, Hauer et al. 2005). Salmonids within the basin include Chinook salmon, chum salmon, coho salmon, sockeye salmon, steelhead, redband trout, bull trout, and cutthroat trout.

Land Use

More than 50% of the U.S. portion of the Columbia River Basin is in federal ownership (most of which occurs in high desert and mountain areas). Approximately 39% is in private land ownership (most of which occurs in river valleys and plateaus). The remaining 11% is divided among the tribes, state, and local governments (Hinck, Schmitt et al. 2004). See Table 39 for a summary of land uses and population densities in several subbasins within the Columbia River watershed (data from (Stanford, Hauer et al. 2005).

Table 39. Land use and population density in select tributaries of the Columbia River (Stanford, Hauer et al. 2005).

Watershed	Land Use Categories (Percent)				Density (people/mi^2)
	Agriculture	Forest	Urban	Other	
Snake/Salmon rivers	30	10-15	1	54 scrub/rangeland/barren	39
Yakima River	16	36	1	47 shrub	80
Willamette River	19	68	5	--	171

The interior Columbia Basin has been altered substantially by humans causing dramatic changes and declines in native fish populations. In general, the basin supports a variety of mixed uses. Predominant human uses include logging, agriculture, ranching, hydroelectric power generation, mining, fishing, a variety of recreational activities, and urban uses. The decline of salmon runs in the Columbia River is attributed to loss of habitat, blocked migratory corridors, altered river flows, pollution, overharvest, and competition from hatchery fish. In the Yakima River, 72 stream and river segments are listed as impaired by the Washington Department of Ecology (DOE) and 83% exceed temperature standards. In the Willamette River, riparian vegetation was greatly reduced by land conversion. By 1990, only 37% of the riparian area within 120 m was forested, 30% was agricultural fields, and 16% was urban or suburban lands. In the Yakima River, non-native grasses and other plants are commonly found along the lower reaches of the river (Stanford, Hauer et al. 2005).

Agriculture and Ranching

Agriculture, ranching, and related services in the Pacific Northwest employ more than nine times the national average [19% of the households within the basin (NRC 2004)].

Ranching practices have led to increased soil erosion and sediment loads within adjacent tributaries. The worst of these effects may have occurred in the late 1800s and early 1900s from deliberate burning to increase grass production (NRC 2004). Several measures are currently in place to reduce the impacts of grazing. Measures include restricted grazing in degraded areas, reduced grazing allotments, and lowered stocking rates. Today, the agricultural industry impacts water quality within the basin. Agriculture is second only to the large-scale influences of hydromodification projects regarding power generation and irrigation. Water quality impacts from agricultural activities include alteration of the natural temperature regime, insecticide and herbicide contamination, and increased suspended sediments.

Roughly 6% of the annual flow from the Columbia River is diverted for the irrigation of 7.3 million acres of croplands within the basin. The vast majority of these agricultural lands are located along the lower Columbia River, the Willamette, Yakima, Hood, and Snake rivers, and the Columbia Plateau (Hinck, Schmitt et al. 2004).

Agriculture and ranching increased steadily within the Columbia River basin from the mid- to late-1800s. By the early 1900s, agricultural opportunities began increasing at a much more rapid pace with the creation of more irrigation canals and the passage of the Reclamation Act of 1902 (NRC 2004). Today, agriculture represents the largest water user within the basin (>90%).

The USGS has a number of fixed water quality sampling sites throughout various tributaries of the Columbia River. Many of the water quality sampling sites have been in place for decades. Water volumes, crop rotation patterns, croptype, and basin location are some of the variables that influence the distribution and frequency of pesticides within a tributary. Detection frequencies for a particular pesticide can vary widely. One study conducted by the USGS between May 1999 and January 2000 in the surface waters of Yakima Basin detected 25 pesticide compounds (Ebbert and Embry 2002). Atrazine was the most widely detected herbicide and azinphos-methyl was the most widely detected insecticide. Other detected compounds include simazine, terbacil, trifluralin;

deethylatrazine, carbaryl, diazinon, malathion, and DDE. In addition to current use-chemicals legacy chemicals continue to pose a serious problem to water quality and fish communities despite their ban in the 1970s and 1980s (Hinck, Schmitt et al. 2004).

Fish and macroinvertebrate communities exhibit an almost linear decline in condition as the level of agriculture intensity increases within a basin (Cuffney, Meador et al. 1997; Fuhrer, Morace et al. 2004). A study conducted in the late 1990s examined 11 species of fish, including anadromous and resident fish collected throughout the basin, for a suite of 132 contaminants. They included 51 semi-volatile chemicals, 26 pesticides, 18 metals, 7 PCBs, 20 dioxins, and 10 furans. Sampled fish tissues revealed PCBs, metals, chlorinated dioxins and furans (products of wood pulp bleaching operations), and other contaminants.

Yakima River Basin: NAWQA analysis

The Yakima River Basin is one of the most agriculturally productive areas in the U.S. (Fuhrer, Morace et al. 2004). Croplands within the Yakima Basin account for about 16% of the total basin area of which 77% is irrigated. The extensive irrigation-water delivery and drainage system in the Yakima River Basin greatly controls water quality conditions and aquatic health in agricultural streams, drains, and the Yakima River (Fuhrer, Morace et al. 2004). From 1999 to 2000, the USGS conducted a NAWQA study in the Yakima River Basin. Fuhrer *et al.* (2004) reported that nitrate and orthophosphate were the dominant forms of nitrogen and phosphorus found in the Yakima River and its agricultural tributaries. Arsenic, a known human carcinogen, was also detected in agricultural drains at elevated concentrations during the nonirrigation season when ground water is the primary source of streamflow.

The USGS also detected 76 pesticide compounds in the Yakima River Basin. They include 38 herbicides (including metribuzin), 17 insecticides (such as carbaryl, diazinon, and malathion), 15 breakdown products, and 6 others. Ninety-one percent of the samples collected from the small agricultural watersheds contained at least two pesticides or pesticide breakdown products. Carbaryl was detected in 29% of tributary samples and

17% of mainstem Yakima River samples at a screening level of 21 nanograms/liter (Fuhrer *et al.* 2004). Carbofuran was screened for, but not detected. The assessment did not screen for methomyl. The median and maximum number of chemicals in a mixture was 8 and 26, respectively (Fuhrer, Morace et al. 2004). The herbicide 2,4-D, occurred most often in the mixtures, along with azinphos-methyl, the most heavily applied pesticide, and atrazine, one of the most aquatic mobile pesticides (Fuhrer, Morace et al. 2004). However, the most frequently detected pesticides in the Yakima River Basin are total DDTs ,(DDT and its breakdown products, dichloro-diphenyl-dichloroethylene (DDE), dichloro-diphenyl-dichloroethane (DDD)), and dieldrin (Johnson and Newman 1983; Joy 2002; Fuhrer, Morace et al. 2004). Nevertheless, concentrations of total DDT in water have decreased since 1991. These reductions are attributed to erosion-controlling best management practices (BMPs).

Williamette Basin: NAWQA analysis

From 1991 to 1995, the USGS also sampled surface waters in the Willamette Basin, Oregon. Wentz *et al.* (1998) reported that, of the 86 tested for, 50 pesticides and pesticide degradates were detected in streams. Ten of the pesticides exceeded criteria established by the EPA for the protection of freshwater aquatic life from chronic toxicity. Carbaryl exceeded protective criteria in 17 of its 46 detections, while carbofuran exceeded limits in three of 51 detections (Wentz *et al.* 1998). Atrazine, simazine, metolachlor, deethylatrazine, diuron, and diazinon were detected in more than one-half of stream samples. Methomyl was tested for but not detected. Forty-nine pesticides were detected in streams draining predominantly agricultural land. About 25 pesticides were detected in streams draining mostly urban areas. The highest pesticide concentrations generally occurred in streams draining predominately agricultural land.

Snake River Basin: NAWQA assessment

The USGS conducted a water quality study from 1992-1995 in the upper Snake River basin, Idaho and Wyoming (Clark, Maret et al. 1998). In basin wide stream sampling in May and June 1994, Eptam [EPTC] (used on potatoes, beans, and sugar beets), atrazine and its breakdown product desethylatrazine (used on corn), metolachlor (used on potatoes and beans), and alachlor (used on beans and corn) were the most commonly detected pesticides. These same compounds accounted for 75% of all detections. Seventeen different pesticides were detected downstream from American Falls Reservoir. Carbaryl and carbofuran were each detected in only 1% of samples; methomyl was screened for but not detected (Clark, Maret et al. 1998).

Hood River Basin

The Hood River Basin ranks fourth in the state of Oregon in total agricultural pesticide usage (Jenkins, Jepson et al. 2004). The land in Hood River basin is used to grow five crops: alfalfa, apples, cherries, grapes, and pears. About 61 a.i.s, totaling 1.1 million lbs, are applied annually to roughly 21,000 acres. Of the top nine, three are carbamates and three are organophosphate insecticides (Table 40). These compounds will have a similar mode of action, though different toxicities, as carbaryl, carbofuran, and methomyl.

Table 40. Amount of most common a.i.s applied to crops in Hood River Basin 1990-1996 (Jenkins et al. 2004).

Active Ingredient	Class	Lbs applied
Oil	-	624,392
Lime Sulfur	-	121,703
Mancozeb	Carbamate	86,872
Sulfur	-	60,552
Ziram	Carbamate	45,965
Azinphos-methyl	Organo-phosphate	22,294
Metam-Sodium	Carbamate	17,114
Phosmet	Organo-phosphate	15,919
Chlorpyrifos	Organo-phosphate	14,833

The Hood River basin contains approximately 400 miles of perennial stream channel, of which an estimated 100 miles is accessible to anadromous fish. These channels are important rearing and spawning habitat for salmonids, making pesticide drift a major concern for the area.

Central Columbia Plateau: NAWQA Assessment

The USGS sampled 31 surface-water sites representing agricultural land use, with different crops, irrigation methods, and other agricultural practices for pesticides in Idaho and Washington from 1992-1995 (Williamson, Munn et al. 1998). Pesticides were detected in samples from all sites, except for the Palouse River at Laird Park (a headwaters site in a forested area). Many pesticides were detected in surface water at very low concentrations. Concentrations of six pesticides exceeded freshwater-chronic criteria for the protection of aquatic life in one or more surface-water samples. They include the herbicide triallate and five insecticides (azinphos-methyl, chlorpyrifos, diazinon, *gamma*-HCH, and parathion). Carbaryl and carbofuran were detected in 6% and 5% of samples, respectively. Methomyl was screened for, but not detected in any samples (Williamson *et al.* 1998).

Detections at four sites were high, ranging from 12 to 45 pesticides. The two sites with the highest detection frequencies are in the Quincy-Pasco subunit, where irrigation and high chemical use combine to increase transport of pesticides to surface waters. Pesticide detection frequencies at sites in the dryland farming (non-irrigated) areas of the North-Central and Palouse subunits are below the national median for NAWQA sites. All four of the sites had at least one pesticide concentration that exceeded a water-quality standard or guideline.

Concentrations of organochlorine pesticides and PCBs are higher than the national median (50^{th} percentile) at seven of 11 sites; four sites were in the upper 25% of all NAWQA sites. Although most of these compounds have been banned, they still persist in the environment. Elevated concentrations were observed in dryland farming areas as well as in irrigated areas.

Urban and Industrial Development

The largest urban area in the basin is the greater Portland metropolitan area, located at the mouth of the Willamette River. Portland's population exceeds 500,000 (Hinck, Schmitt et al. 2004). Although the basin's land cover is about 8% of the U.S. total land mass, its human population is one-third the national average (about 1.2% of the U.S. population) (Hinck, Schmitt et al. 2004).

Discharges from sewage treatment plants, paper manufacturing, and chemical and metal production represent the top three permitted sources of contaminants within the lower basin according to discharge volumes and concentrations (Rosetta and Borys 1996). Rosetta and Borys (1996) review of 1993 data indicate that 52% of the point source waste water discharge volume is from sewage treatment plants, 39% from paper and allied products, 5% from chemical and allied products, and 3% from primary metals. However, the paper and allied products industry are the primary sources of the suspended sediment load (71%). Additionally, 26% comes from sewage treatment plants and 1% is from the chemical and allied products industry. Nonpoint source discharges (urban stormwater runoff) account for significant pollutant loading to the lower basin, including most organics and over half of the metals. Although rural nonpoint sources contributions were not calculated, Rosetta and Borys (1996) surmised that in some areas and for some contaminants, rural areas may contribute a large portion of the load. This is particularly true for pesticide contamination in the upper river basin where agriculture is the predominant land use.

Water quality has been reduced by phosphorus loads and decreased water clarity, primarily along the lower and middle sections of the Columbia River Estuary. Although sediment quality is generally very good, benthic indices have not been established within the estuary. Fish tissue contaminant loads (PCBs, DDT, DDD, DDE, and mercury) are high and present a persistent and long lasting effect on estuary biology. Health advisories have been recently issued for people eating fish in the area that contain high levels of dioxins, PCBs, and pesticides.

Habitat Modification

Basin wide, critical ecological connectivity (mainstem to tributaries and riparian floodplains) has been disconnected by dams and associated activities such as floodplain deforestation and urbanization. Dams have flooded historical spawning and rearing habitat with the creation of massive water storage reservoirs. More than 55% of the Columbia River Basin that was accessible to salmon and steelhead before 1939 has been blocked by large dams (NWPPC 1986). Construction of the Grand Coulee Dam blocked 1,000 miles (1,609 km) of habitat from migrating salmon and steelhead (Wydoski and

Whitney 1979). Similarly, over one third (2,000 km) of coho salmon habitat is no longer accessible (Good, Waples et al. 2005). The mainstem habitats of the lower Columbia and Willamette rivers have been reduced primarily to a single channel. As a result, floodplain area is reduced, off-channel habitat features have been eliminated or disconnected from the main channel, and the amount of LWD in the mainstem has been reduced. Remaining areas are affected by flow fluctuations associated with reservoir management for power generation, flood control, and irrigation. Overbank flow events, important to habitat diversity, have become rare as a result of controlling peak flows and associated revetments. Portions of the basin are also subject to impacts from cattle grazing and irrigation withdrawals. Consequently, estuary dynamics have changed substantially.

Stream habitat degradation in Columbia Central Plateau is relatively high (Williamson, Munn et al. 1998). In the most recent NAWQA survey, a total of 16 sites were evaluated - all of which showed signs of degradation (Williamson, Munn et al. 1998). Streams in this area have an average of 20% canopy cover and 70% bank erosion. These factors have severely affected the quality of habitat available to salmonids. The Palouse subunit of the Lower Snake River exceeds temperature levels for the protection of aquatic life (Williamson, Munn et al. 1998).

Habitat loss has fragmented habitat and human density increase has created additional loads of pollutants and contaminants within the Columbia River Estuary (Anderson, Dugger et al. 2007). About 77% of swamps, 57% of marshes, and over 20% of tree cover have been lost to development and industry. Twenty four threatened and endangered species occur in the estuary, some of which are recovering and others (*i.e.*, Chinook salmon) are not.

The Willamette Basin Valley has been dramatically changed by modern settlement. The complexity of the mainstem river and extent of riparian forest have both been reduced by 80% (PNERC 2002). About 75% of what was formerly prairie and 60% of what was wetland has been converted to agricultural purposes. These actions, combined with urban development, extensive (96 miles) bank stabilization, and in-river and near-shore gravel mining, have resulted in a loss of floodplain connectivity and off-channel habitat (PNERC 2002).

Habitat Restoration

Since 2000, land management practices included improving access by replacing culverts and fish habitat restoration activities at Federal Energy Regulatory Commission (FERC)-licensed dams. Habitat restoration in the upper (reducing excess sediment loads) and lower Grays River watersheds may benefit the Grays River chum salmon population as it has a subyearling juvenile life history type and rears in such habitats. Short-term daily flow fluctuations at Bonneville Dam sometimes create a barrier (*i.e.*, entrapment on shallow sand flats) for fry moving into the mainstem rearing and migration corridor. Some chum fry have been stranded on shallow water flats on Pierce Island from daily flow fluctuations. Coho salmon are likely to be affected by flow and sediment delivery changes in the Columbia River plume. Steelhead may be affected by flow and sediment delivery changes in the plume (Casillas 1999).

In 2006, NOAA Fisheries completed consultation on issuance of a 50-year incidental take permit to the State of Washington for its Washington State Forest Practices Habitat Conservation Plan (HCP). The HCP is expected to improve habitat conditions on state forest lands within the action area. Improvements include removing barriers to migration, restoring hydrologic processes, increasing the number of large trees in riparian zones, improving stream bank integrity, and reducing fine sediment inputs (FCRPS 2008).

Mining

Most of the mining in the basin is focused on minerals such as phosphate, limestone, dolomite, perlite, or metals such as gold, silver, copper, iron, and zinc. Mining in the region is conducted in a variety of methods and places within the basin. Alluvial or glacial deposits are often mined for gold or aggregate. Ores are often excavated from the hard bedrocks of the Idaho batholiths. Eleven percent of the nation's output of gold has come from mining operations in Washington, Montana, and Idaho. More than half of the nation's silver output has come from a few select silver deposits.

Many of the streams and river reaches in the basin are impaired from mining. Several abandoned and former mining sites are also designated as superfund cleanup areas (Stanford, Hauer et al. 2005; Anderson, Dugger et al. 2007). According to the U.S. Bureau of Mines, there are about 14,000 inactive or abandoned mines within the Columbia River Basin. Of these, nearly 200 pose a potential hazard to the environment (Quigley, Arbelbide et al. 1997 in Hincke et al. 2004). Contaminants detected in the water include lead and other trace metals.

Hydromodification Projects

More than 400 dams exist in the basin, ranging from mega dams that store large amounts of water to small diversion dams for irrigation. Every major tributary of the Columbia River except the Salmon River is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects. Of these, 18 dams are located on the mainstem Columbia River and its major tributary, the Snake River. The FCRPS encompasses the operations of 14 major dams and reservoirs on the Columbia and Snake rivers. These dams and reservoirs operate as a coordinated system. The Corps operates 9 of 10 major federal projects on the Columbia and Snake rivers, and the Dworshak, Libby and Albeni Falls dams. The BOR operates the Grand Coulee and Hungry Horse dams. These federal projects are a major source of power in the region. These same projects provide flood control, navigation, recreation, fish and wildlife, municipal and industrial water supply, and irrigation benefits.

BOR has operated irrigation projects within the basin since 1904. The irrigation system delivers water to about 2.9 million acres of agricultural lands. About 1.1 million acres of land are irrigated using water delivered by two structures, the Columbia River Project (Grand Coulee Dam) and the Yakima Project. The Grand Coulee Dam delivers water for the irrigation of over 670,000 acres of croplands and the Yakima Project delivers water to nearly 500,000 acres of croplands (Bouldin, Farris et al. 2007).

The Bonneville Power Administration (BPA), an agency of the U.S. Department of Energy, wholesales electric power produced at 31 federal dams (67% of its production) and non-hydropower facilities in the Columbia-Snake Basin. The BPA sells about half the electric power consumed in the Pacific Northwest. The federal dams were developed over a 37-year period starting in 1938 with Bonneville Dam and Grand Coulee in 1941, and ending with construction of Libby Dam in 1973 and Lower Granite Dam in 1975.

Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on the ecosystems of the Columbia River Basin (ISG 1996). These effects have been especially adverse to the survival of anadromous salmonids. The construction of the FCRPS modified migratory habitat of adult and juvenile salmonids. In many cases, the FCRPS presented a complete barrier to habitat access for salmonids. Approximately 80% of historical spawning and rearing habitat of Snake River fall-run Chinook salmon is now inaccessible due to dams. The Snake River spring/summer run has been limited to the Salmon, Grande Ronde, Imnaha, and Tuscanon rivers. Damming has cut off access to the majority of Snake River Chinook salmon spawning habitat. The Sunbeam Dam on the Salmon River is believed to have limited the range of Snake River sockeye salmon as well.

Both upstream and downstream migrating fish are impeded by the dams. Additionally, a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delays in passage, exposure to high concentrations of dissolved gases, warm water, and increased predation. Non-federal hydropower facilities on Columbia River tributaries have also partially or completely blocked higher elevation spawning.

Qualitatively, several hydromodification projects have improved the productivity of naturally produced Snake River fall Chinook salmon. Improvements include flow augmentation to enhance water flows through the lower Snake and Columbia Rivers

(USBR 1998 *in* (FCRPS 2008); providing stable outflows at Hells Canyon Dam during the fall Chinook salmon spawning season and maintaining these flows as minimums throughout the incubation period to enhance survival of incubating fall-run Chinook salmon; and reduced summer temperatures and enhanced summer flow in the lower Snake River (see (Corps, BPA et al. 2007), Appendix 1 *in* (FCRPS 2008)). Providing suitable water temperatures for over-summer rearing within the Snake River reservoirs allows the expression of productive “yearling” life history strategy that was previously unavailable to Snake River fall-run Chinook salmon.

The mainstem FCRPS corridor has also improved safe passage through the hydrosystem for juvenile steelhead and yearling Chinook salmon with the construction and operation of surface bypass routes at Lower Granite, Ice Harbor, and Bonneville dams and other configuration improvements (Corps, BPA et al. 2007).

For salmon, with a stream-type juvenile life history, projects that have protected or restored riparian areas and breached or lowered dikes and levees in the tidally influenced zone of the estuary have improved the function of the juvenile migration corridor. The FCRPS action agencies recently implemented 18 estuary habitat projects that removed passage barriers. These activities provide fish access to good quality habitat.

The Corps et al. (2007) estimated that hydropower configuration and operational improvements implemented from 2000 to 2006 have resulted in an 11.3% increase in survival for yearling juvenile LCR Chinook salmon from populations that pass Bonneville Dam. Improvements during this period included the installation of a corner collector at Powerhouse II (PH2) and the partial installation of minimum gap runners at Powerhouse 1 (PH1) and of structures that improve fish guidance efficiency at PH2. Spill operations have been improved and PH2 is used as the first priority powerhouse for power production because bypass survival is higher than at PH1. Additionally, drawing water towards PH2 moves fish toward the corner collector. The bypass system screen was removed from PH1 because tests showed that turbine survival was higher than through the bypass system at that location.

Artificial Propagation

There are several artificial propagation programs for salmon production within the Columbia River Basin. These programs were instituted under federal law to lessen the effects of lost natural salmon production within the basin from the dams. The hatcheries are operated by federal, state, and tribal managers. For more than 100 years, hatcheries in the Pacific Northwest have been used to produce fish for harvest and replace natural production lost to dam construction. Hatcheries have only minimally been used to protect and rebuild naturally produced salmonid population (*e.g.*, Redfish Lake sockeye salmon). In 1987, 95% of the coho salmon, 70% of the spring Chinook salmon, 80% of the summer Chinook salmon, 50% of the fall-run Chinook salmon, and 70% of the steelhead returning to the Columbia River Basin originated in hatcheries (CBFWA 1990). More recent estimates suggest that almost half of the total number of smolts produced in the basin come from hatcheries (Beechie, Liermann et al. 2005).

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive (Hard, Jones et al. 1992). Hatchery practices, among other factors, are a contributing factor to the 90% reduction in natural coho salmon runs in the lower Columbia River over the past 30 years (Flagg, Waknitz et al. 1995). Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from non-native basins. The impacts of these hatchery practices are largely unknown. Adverse effects of these practices likely included: loss of genetic variability within and among populations (Busack 1990; Riggs 1990; Hard, Jones et al. 1992; Reisenbichler 1997), disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and the displacement of natural fish (Steward and Bjornn 1990; Hard, Jones et al. 1992; Fresh 1997). Species with extended freshwater residence may face higher risk of domestication, predation, or altered migration than species that spend only a brief time in freshwater (Hard, Jones et al. 1992). Nonetheless, artificial propagation may also contribute to the conservation of listed salmon and steelhead. However, it is unclear whether or how much artificial propagation during the recovery process will compromise the distinctiveness of natural populations (Hard, Jones et al. 1992).

The States of Oregon and Wasington and other fisheries co-managers are engaged in a substantial review of hatchery management practices through the Hatchery Scientific Review Group (HSRG). The HSRG was established and funded by Congress to provide an independent review of current hatchery program in the Columbia River Basin. The HSRG has completed its work on LCR populations and provided its recommendations. A general conclusion is that the current production programs are inconsistent with practices that reduce impacts on naturally-spawning populations, and will have to be modified to reduce adverse effects on key natural populations identified in the Interim Recovery Plan. The adverse effects are caused by hatchery-origin adults spawning with natural-origin fish or competing with natural-origin fish for spawning sites (FCRPS 2008). Oregon and Washington initiated a comprehensive program of hatchery and associated harvest reforms (WDFW 2005; ODFW 2007). The program is designed to achieve HSRG objectives related to controlling the number of hatchery-origin fish on the spawning grounds and in the hatchery broodstock.

Coho salmon hatchery programs in the lower Columbia have been tasked to compensate for impacts of fisheries. However, hatchery programs in the LCR have not operated specifically to conserve LCR coho salmon. These programs threaten the viability of natural populations. The long-term domestication of hatchery fish has eroded the fitness of these fish in the wild and has reduced the productivity of wild stocks where significant numbers of hatchery fish spawn with wild fish. Large numbers of hatchery fish have also contributed to more intensive mixed stock fisheries. These programs largely overexploited wild populations weakened by habitat degradation. Most LCR coho salmon populations have been heavily influenced by hatchery production over the years.

Commercial, Recreational, and Subsistence Fishing

Archeological records indicate that indigenous people caught salmon in the Columbia River more than 7,000 years ago. One of the most well known tribal fishing sites within the basin was located near Celilo Falls, an area in the lower river that has been occupied by Dalles Dam since 1957. Salmon fishing increased with better fishing methods and preservation techniques, such as drying and smoking. Salmon harvest substantially increased in the mid-1800s with canning techniques. Harvest techniques also changed over time, from early use of hand-held spears and dip nets, to riverboats using seines and gill-nets. Harvest techniques eventually transitioned to large ocean-going vessels with trolling gear and nets and the harvest of Columbia River salmon and steelhead from California to Alaska (Beechie, Liermann et al. 2005).

During the mid-1800s, an estimated 10 to 16 million adult salmon of all species entered the Columbia River each year. Large annual harvests of returning adult salmon during the late 1800s ranging from 20 million to 40 million lbs of salmon and steelhead significantly reduced population productivity (Beechie, Liermann et al. 2005). The largest known harvest of Chinook salmon occurred in 1883 when Columbia River canneries processed 43 million lbs of salmon (Lichatowich 1999). Commercial landings declined steadily from the 1920s to a low in 1993. At that time, just over one million lbs of Chinook salmon were harvested (Beechie, Liermann et al. 2005).

Harvested and spawning adults reached 2.8 million in the early 2000s, of which almost half are hatchery produced (Beechie, Liermann et al. 2005). Most of the fish caught in the river are steelhead and spring/summer Chinook salmon. Ocean harvest consists largely of coho and fall Chinook salmon. Most ocean catches are made north of Cape Falcon, Oregon. Over the past five years, the number of spring and fall salmon commercially harvested in tribal fisheries has averaged between 25,000 and 110,000 fish (Beechie, Liermann et al. 2005). Recreational catch in both ocean and in-river fisheries varies from 140,000 to 150,000 individuals (Beechie, Liermann et al. 2005).

Non-Indian fisheries in the lower Columbia River are limited to a harvest rate of 1%. Treaty Indian fisheries are limited to a harvest rate of 5 to 7%, depending on the run size of upriver Snake River sockeye stocks. Actual harvest rates over the last 10 years have ranged from 0 to 0.9%, and 2.8 to 6.1%, respectively (see TAC 2008, Table 15 *in* FCRPS (2008).

Columbia River chum salmon are not caught incidentally in tribal fisheries above Bonneville Dam. However, Columbia River chum salmon are incidentally caught occasionally in non-Indian fall season fisheries below Bonneville Dam. There are no fisheries in the Columbia River that target hatchery or natural-origin chum salmon. The species' later fall return timing make them vulnerable to relatively little potential harvest in fisheries that target Chinook salmon and coho salmon. Columbia River chum salmon rarely take the sport gear used to target other species. Incidental catch of chum amounts to a few tens of fish per year (TAC 2008). The harvest rate of Columbia River chum salmon in proposed state fisheries in the lower river is estimated to be 1.6% per year and is less than 5%.

LCR coho salmon are harvested in the ocean and in the Columbia River and tributary freshwater fisheries of Oregon and Washington. Incidental take of coho salmon prior to the 1990s fluctuated from approximately 60 to 90%. However, this number has been reduced since its listing to 15 to 25% (LCFRB 2004). The exploitation of hatchery coho salmon has remained approximately 50% through the use of selective fisheries.

LCR steelhead are harvested in Columbia River and tributary freshwater fisheries of Oregon and Washington. Fishery impacts of LCR steelhead have been limited to less than 10% since implementation of mark-selective fisheries during the 1980s. Recent harvest rates on UCR steelhead in non-Treaty and treaty Indian fisheries ranged from 1% to 2%, and 4.1% to 12.4%, respectively (FCRPS 2008).

Alien Species

Many non-native species have been introduced to the Columbia River Basin since the 1880s. At least 81 invasive species have currently been identified, composing one-fifth of all species in some areas. New non-native species are discovered in the basin regularly; a new aquatic invertebrate is discovered approximately every 5 months (Sytsma, Cordell et al. 2004). It is clear that the introduction of non-native species has changed the environment, though whether these changes will impact salmonid populations is uncertain (Sytsma, Cordell et al. 2004).

Puget Sound Region

Puget Sound is the second largest estuary in the U.S. It has about 1,330 miles of shoreline and extends from the mouth of the Strait of Juan de Fuca east. Puget Sound includes the San Juan Islands and south to Olympia, and is fed by more than 10,000 rivers and streams.

Puget Sound is generally divided into four major geographic marine basins: Hood Canal, South Sound, Whidbey Basin, and the Main Basin. The Main Basin has been further subdivided into two subbasins: Admiralty Inlet and Central Basin. About 43% of the Puget Sound's tideland is located in the Whidbey Island Basin. This reflects the large influence of the Skagit River, which is the largest river in the Puget Sound system and whose sediments are responsible for the extensive mudflats and tidelands of Skagit Bay.

Habitat types that occur within the nearshore environment include eelgrass meadows, kelp forest, mud flats, tidal marshes, sub-estuaries (tidally influenced portions of river and stream mouths), sand spits, beaches and backshore, banks and bluffs, and marine riparian vegetation. These habitats provide critical functions such as primary food production and support habitat for invertebrates, fish, birds, and other wildlife.

Major rivers draining to Puget Sound from the Cascade Mountains include the Skagit, Snohomish, Nooksack, Puyallup, and Green rivers, as well as the Lake Washington/Cedar River watershed. Major rivers from the Olympic Mountains include the Hamma Hamma, the Duckabush, the Quilcene, and the Skokomish rivers. Numerous other smaller rivers drain to the Sound, many of which are significant salmonid production areas despite their small size.

The Puget Sound basin is home to more than 200 fish and 140 mammalian species. Salmonids within the region include coho, Chinook, sockeye, chum, and pink salmon, kokanee, steelhead, rainbow, cutthroat, and bull trout (Wydoski and Whitney 1979; Kruckeberg 1991). Important commercial fishes include the five Pacific salmon and several rockfish species. A number of introduced species occur within the region, including brown and brook trout, Atlantic salmon, bass, tunicates (sea squirts), and a saltmarsh grass (*Spartina* spp.). Estimates suggest that over 90 species have been intentionally or accidentally introduced in the region (Ruckelshaus and McClure 2007). At present, over 40 species in the region are listed as threatened and endangered under the ESA.

Puget Sound is unique among the nation's estuaries as it is a deep fjord-like structure that contains many urban areas within its drainage basin (Collier, O'Neill et al. 2006). Because of the several sills that limit entry of oceanic water into Puget Sound, it is relatively poorly flushed compared to other urbanized estuaries of North America. Thus, toxic chemicals that enter Puget Sound have longer residence times within the system. This entrainment of toxics can result in biota exposure to increased levels of contaminant for a given input, compared to other large estuaries. This hydrologic isolation puts the Puget Sound ecosystem at higher risk from other types of populations that enter the system, such as nutrients and pathogens.

Because Puget Sound is a deep, almost oceanic habitat, the tendency of a number of species to migrate outside of Puget Sound is limited relative to similar species in other large urban estuaries. This high degree of residency for many marine species, combined with the poor flushing of Puget Sound, results in a more protracted exposure to contaminants. The combination of hydrologic and biological isolation makes the Puget Sound ecosystem highly susceptible to inputs of toxic chemicals compared to other major estuarine ecosystems (Collier, O'Neill et al. 2006).

An indication of this sensitivity occurs in Pacific herring, one of Puget Sound's keystone forage fish species (Collier, O'Neill et al. 2006). These fish spend almost all of their lives in pelagic waters and feed at the lower end of the food chain. Pacific herring should be among the least contaminated of fish species. However, monitoring has shown that herring from the main basins of Puget Sound have higher body burdens of persistent chemicals (*e.g.*, PCBs) compared to herring from the severely contaminated Baltic Sea. Thus, the pelagic food web of Puget Sound appears to be more seriously contaminated than previously anticipated.

Chinook salmon that are resident in Puget Sound (a result of hatchery practices and natural migration patterns) are several times more contaminated with persistent bioaccumulative contaminants than other salmon populations along the West Coast (Collier, O'Neill et al. 2006). Because of associated human health concerns, fish consumption guidelines for Puget Sound salmon are under review by the Washington State Department of Health.

Extremely high levels of chemical contaminants are also found in Puget Sound's top predators, including harbor seals and ESA-listed southern resident killer whales (Collier, O'Neill et al. 2006). In addition to carrying elevated loads of toxic chemicals in their tissues, Puget Sound's biota also show a wide range of adverse health outcomes associated with exposure to chemical contaminants. They include widespread cancer and reproductive impairment in bottom fish, increased susceptibility to disease in juvenile salmon, acute die-offs of adult salmon returning to spawn in urban watersheds, and egg and larval mortality in a variety of fish. Given current regional projections for population growth and coastal development, the loadings of chemical contaminants into Puget Sound will increase dramatically in future years.

Land Use

The Puget Sound Lowland contains the most densely populated area of Washington. The regional population in 2003 was an estimated 3.8 million people, with 86% residing in King, Pierce, and Snohomish counties (Snohomish, Cedar-Sammamish Basin, Green-Duwamish, and Puyallup River watersheds). The area is expected to attract 4 to 6 million new human residents in the next 20 years (Ruckelshaus and McClure 2007). The Snohomish River watershed, one of the fastest growing watersheds in the region, increased about 16% in the same period.

Land use in the Puget Sound lowland is composed of agricultural areas (including forests for timber production), urban areas (industrial and residential use), and rural areas (low density residential with some agricultural activity). Pesticides are regularly applied to agricultural and non-agricultural lands and are found virtually in every land use area. Pesticides and other contaminants drain into ditches in agricultural areas and eventually to stream systems. Roads bring surface water runoff to stream systems from industrial, residential, and landscaped areas in the urban environment. Pesticides are also typically found in the right-of-ways of infrastructure that connect the major landscape types. Right-of-ways are associated with roads, railways, utility lines, and pipelines.

In the 1930s, all of western Washington contained about 15.5 million acres of "harvestable" forestland. By 2004, the total acreage was nearly half that originally surveyed (PSAT 2007). Forest cover in Puget Sound alone was about 5.4 million acres in the early 1990s. About a decade later, the region had lost another 200,000 acres of forest cover with some watersheds losing more than half the total forested acreage. The most intensive loss of forest cover occurred in the Urban Growth Boundary, which encompasses specific parts of the Puget Lowland. In this area, forest cover declined by 11% between 1991 and 1999 (Ruckelshaus and McClure 2007). Projected land cover changes indicate that trends are likely to continue over the next several decades with population changes (Ruckelshaus and McClure 2007). Coniferous forests are also projected to decline at an alarming rate as urban uses increase.

According to the 2001 State of the Sound report (PSAT 2007), impervious surfaces covered 3.3% of the region, with 7.3% of lowland areas (below 1,000 ft elevation) covered by impervious surfaces. From 1991 to 2001, the amount of impervious surfaces increased 10.4% region wide. Consequently, changes in rainfall delivery to streams alter stream flow regimes. Peak flows are increased and subsequent base flows are decreased and alter in-stream habitat. Stream channels are widened and deepened and riparian vegetation is typically removed which can cause increases in water temperature and will reduce the amounts of woody debris and organic matter to the stream system.

Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, polybrominated diphenyl ethers (PBDEs) compounds, PAHs, nutrients (phosphorus and nitrogen), and sediment (Table 41). Other ions generally elevated in urban streams include calcium, sodium, potassium, magnesium, and chloride ions where sodium chloride is used as the principal road deicing salt (Paul and Meyer 2001). The combined effect of increased concentrations of ions in streams is the elevated conductivity observed in most urban streams.

Table 41. Examples of Water Quality Contaminants in Residential and Urban Areas

Contaminant groups	Select constituents	Select example(s)	Source and Use Information
Fertilizers	Nutrients	Phosphorus Nitrogen	lawns, golf courses, urban landscaping
Heavy Metals	Pb, Zn, Cr, Cu, Cd, Ni, Hg, Mg	Cu	brake pad dust, highway and parking lot runoff, rooftops
Pesticides including- Insecticides (I) Herbicides (H) Fungicides (F) Wood Treatment chemicals (WT) Legacy Pesticides (LP) Other ingredients in pesticide formulations (OI)	Organophosphates (I) Carbamates (I) Organochlorines (I) Pyrethroids (I) Triazines (H) Chloroacetanilides (H) Chlorophenoxy acids (H) Triazoles (F) Copper containing fungicides (F) Organochlorines (LP) Surfactants/adjuvants (OI)		golf courses, right of ways, lawn and plant care products, pilings, bulkheads, fences
Pharmaceuticals and personal care products	Natural and synthetic hormones soaps and detergents	Ethinyl estradiol Nonylphenol	hospitals, dental facilities, residences, municipal and industrial waste water discharges
Polyaromatic hydrocarbons (PAHs)	Tricyclic PAHs	Phenanthrene	fossil fuel combustion, oil and gasoline leaks, highway runoff, creosote-treated wood
Industrial chemicals	PCBs PBDEs Dioxins	Penta-PBDE	utility infrastructure, flame retardants, electronic equipment

Many other metals have been found in elevated concentrations in urban stream sediments including arsenic, iron, boron, cobalt, silver, strontium, rubidium, antimony, scandium, molybdenum, lithium, and tin (Wheeler, Angermeier et al. 2005). The concentration, storage, and transport of metals in urban streams are connected to particulate organic matter content and sediment characteristics. Organic matter has a high binding capacity for metals and both bed and suspended sediments with high organic matter content frequently exhibit 50-7,500 times higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium than sediments with lower organic matter content.

Although urban areas occupy only 2% of the Pacific Northwest land base, the impacts of urbanization on aquatic ecosystems are severe and long lasting (Spence, Lomnicky et al. 1996). O'Neill *et al.* (2006) found that Chinook salmon returning to Puget Sound had significantly higher concentrations of PCBs and PBDEs compared to other Pacific coast salmon populations. Furthermore, Chinook salmon that resided in Puget Sound in the winter rather than migrate to the Pacific Ocean (residents) had the highest concentrations of persistent organic pollutants (POPs), followed by Puget Sound fish populations believed to be more ocean-reared. Fall Chinook salmon from Puget Sound have a more localized marine distribution in Puget Sound and the Georgia Basin than other populations of Chinook salmon from the west coast of North America. This ESU is more contaminated with PCBs (2 to 6 times) and PBDEs (5 to 17 times). O'Neill *et al.* (2006) concluded that regional body burdens of contaminants in Pacific salmon, and Chinook salmon in particular, could contribute to the higher levels of contaminants in federally-listed endangered southern resident killer whales.

Endocrine disrupting compounds (EDCs) are chemicals that mimic natural hormones, inhibit the action of hormones and/or alter normal regulatory functions of the immune, nervous and endocrine systems and can be discharged with treated effluent (King County 2002d). Endocrine disruption has been attributed to DDT and other organochlorine pesticides, dioxins, PAHs, alkylphenolic compounds, phthalate plasticizers, naturally occurring compounds, synthetic hormones and metals. Natural mammalian hormones such as 17β-estradiol, are also classified as endocrine disruptors. Both natural and synthetic mammalian hormones are excreted through the urine and are known to be present in wastewater discharges.

Jobling *et al.* (1995) reported that ten chemicals known to occur in sewage effluent interacted with the fish estrogen receptor by reducing binding of 17β-estradiol to its receptor, stimulating transcriptional activity of the estrogen receptor or inhibiting transcription activity. Binding of the ten chemicals with the fish endocrine receptor indicates that the chemicals could be endocrine disruptors and forms the basis of concern about WWTP effluent and fish endocrine disruption.

Fish communities are impacted by urbanization (Wheeler, Angermeier et al. 2005). Urban stream fish communities have lower overall abundance, diversity, taxa richness and are dominated by pollution tolerant species. Lead content in fish tissue is higher in urban areas. Furthermore, the proximity of urban streams to humans increases the risk of non-native species introduction and establishment. Thirty-nine non-native species were collected in Puget Sound during the 1998 Puget Sound Expedition Rapid Assessment Survey (Brennan, Higgins et al. 2004). Lake Washington, located within a highly urban area, has 15 non-native species identified (Ajawani 1956).

PAH compounds also have distinct and specific effects on fish at early life history stages (Incardona, Collier et al. 2004). PAHs tend to adsorb to organic or inorganic matter in sediments, where they can be trapped in long-term reservoirs (Johnson, Collier et al. 2002). Only a portion of sediment-adsorbed PAHs are readily bioavailable to marine organisms, but there is substantial uptake of these compounds by resident benthic fish through the diet, through exposure to contaminated water in the benthic boundary layer, and through direct contact with sediment. Benthic invertebrate prey are a particularly important source of PAH exposure for marine fishes, as PAHs are bioaccumulated in many invertebrate species (Varanasi, Stein et al. 1989; Varanasi, Stein et al. 1992; Meador, Stein et al. 1995).

PAHs and their metabolites in invertebrate prey can be passed on to consuming fish species, PAHs are metabolized extensively in vertebrates, including fishes (Johnson, Collier et al. 2002). Although PAHs do not bioaccumulate in vertebrate tissues, PAHs cause a variety of deleterious effects in exposed animals. Some PAHs are known to be immunotoxic and to have adverse effects on reproduction and development. Studies show that PAHs exhibit many of the same toxic effects in fish as they do in mammals (Johnson, Collier et al. 2002).

Habitat Modification

Much of the region's estuarine wetlands have been heavily modified, primarily from agricultural land conversion and urban development (NRC 1996). Although most estuarine wetland losses result from conversions to agricultural land by ditching, draining, or diking, these wetlands also experience increasing effects from industrial and urban causes. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at 11 deltas in Puget Sound (Bortleson, Chrzastowski et al. 1980). Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50-90%. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at eleven deltas in Puget Sound (Bortleson, Chrzastowski et al. 1980). More recently, tidal wetlands in Puget Sound amount to about 17-19% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50-90% common for individual estuaries. Salmon use freshwater and estuarine wetlands for physiological transition to and from saltwater and rearing habitat. The land conversions and losses of Pacific Northwest wetlands constitute a major impact. Salmon use marine nearshore areas for rearing and migration, with juveniles using shallow shoreline habitats (Brennan, Higgins et al. 2004).

About 800 miles of Puget Sound's shorelines are hardened or dredged (PSAT 2004; Ruckelshaus and McClure 2007). The area most intensely modified is the urban corridor (eastern shores of Puget Sound from Mukilteo to Tacoma). Here, nearly 80% of the shoreline has been altered, mostly from shoreline armoring associated with the Burlington Northern Railroad tracks (Ruckelshaus and McClure 2007). Levee development within the rivers and their deltas has isolated significant portions of former floodplain habitat that was historically used by salmon and trout during rising flood waters.

Urbanization has caused direct loss of riparian vegetation and soils and significantly altered hydrologic and erosion rates. Watershed development and associated urbanization throughout the Puget Sound, Hood Canal, and Strait of Juan de Fuca regions have increased sedimentation, raised water temperatures, decreased LWD recruitment, decreased gravel recruitment, reduced river pools and spawning areas, and dredged and filled estuarine rearing areas (Bishop and Morgan 1996 in (NMFS 2008)). Large areas of the lower rivers have been channelized and diked for flood control and to protect agricultural, industrial, and residential development.

The NMFS' 2005 Report to Congress on implementation of the Pacific Coastal Salmon Recovery Fund listed habitat-related factors as the leading limits to Puget Sound Chinook Salmon and Hood-Canal Summer Run Chum recovery (PCSRF 2006). Similarly, the principal factor for decline of Puget Sound steelhead is the destruction, modification, and curtailment of its habitat and range. Barriers to fish passage and adverse effects on water quality and quantity resulting from dams, the loss of wetland and riparian habitats, and agricultural and urban development activities have contributed and continue to contribute to the loss and degradation of steelhead habitats in Puget Sound (NMFS 2008).

Industrial Development

More than 100 years of industrial pollution and urban development have affected water quality and sediments in Puget Sound. Many different kinds of activities and substances release contamination into Puget Sound and the contributing waters. According to the State of the Sound Report (PSAT 2007) in 2004, more than 1,400 fresh and marine waters in the region were listed as “impaired.” Almost two-thirds of these water bodies were listed as impaired due to contaminants, such as toxics, pathogens, and low dissolved oxygen or high temperatures, and less than one-third had established cleanup plans. More than 5,000 acres of submerged lands (primarily in urban areas; 1% of the study area) are contaminated with high levels of toxic substances, including polybrominated diphenyl ethers (PBDEs; flame retardants), and roughly one-third (180,000 acres) of submerged lands within Puget Sound are considered moderately contaminated. In 2005 the Puget Sound Action Team (PSAT) identified the primary pollutants of concern in Puget Sound and their sources listed below in Table 42.

Table 42. Pollutants of Concern in Puget Sound (PSAT 2005)

Pollutant	Sources
Heavy Metals: Pb, Hg, Cu, and others	vehicles, batteries, paints, dyes, stormwater runoff, spills, pipes.
Organic Compounds: Polycyclic aromatic hydrocarbons (PAHs)	Burning of petroleum, coal, oil spills, leaking underground fuel tanks, creosote, asphalt.
Polychlorinated biphenyls (PCBs)	Solvents electrical coolants and lubricants, pesticides, herbicides, treated wood.
Dioxins, Furans	Byproducts of industrial processes.
Dichloro-diphenyl-trichloroethane (DDTs)	Chlorinated pesticides.
Phthalates	Plastic materials, soaps, and other personal care products. Many of these compounds are in wastewater from sewage treatment plants.
Polybrominated diphenyl ethers (PBDEs)	PBDEs are added to a wide range of textiles and plastics as a flame retardant. They easily leach from these materials and have been found throughout the environment and in human breast milk.

Puget Sound Basin: NAWQA analysis

The USGS sampled waters in the Puget Sound Basin between 1996 and 1998. (Ebbert, Embrey et al. 2000) reported that 26 of 47 analyzed pesticides were detected. A total of 74 manmade organic chemicals were detected in streams and rivers, with different mixtures of chemicals linked to agricultural and urban settings. NAWQA results reported that the herbicides atrazine, prometon, simazine and tebuthiuron were the most frequently detected herbicides in surface and ground water (Bortleson and Ebbert 2000). Herbicides were the most common type of pesticide found in an agricultural stream (Fishtrap Creek) and the only type of pesticide found in shallow ground water underlying agricultural land (Bortleson and Ebbert 2000). The most commonly detected VOC in the agricultural land use study area was associated with the application of fumigants to soils prior to planting (Bortleson and Ebbert 2000). One or more fumigant-related compound (1,2-dichloropropane, 1,2,2-trichloropropane, and 1,2,3-trichloropropane) were detected in over half of the samples. Insecticides, in addition to herbicides, were detected frequently in urban streams (Bortleson and Ebbert 2000). Sampled urban streams showed the highest detection rate for the three insecticides: carbaryl, diazinon, and malathion. Carbaryl was detected at over 60% of urban sample sites (Ebbert, Embrey et al. 2000). The insecticide diazinon was also frequently detected in urban streams at concentrations that exceeded EPA guidelines for protecting aquatic life (Bortleson and Ebbert 2000). Insecticides screened for included both carbofuran and methomyl. Carbofuran was detected, while methomyl was not. No insecticides were found in shallow ground water below urban residential land (Bortleson and Ebbert 2000).

Habitat Restoration

Positive changes in water quality in the region are evident. One of the most notable improvements was the elimination of sewage effluent to Lake Washington in the mid-1960s. This significantly reduced problems within the lake from phosphorus pollution and triggered a concomitant reduction in cyanobacteria (Ruckelshaus and McClure 2007). Even so, as the population and industry has risen in the region a number of new and legacy pollutants are of concern.

Mining

Mining has a long history in Washington. In 2004, the state was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano, Ellis et al. 1993; NMA 2007). Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 (Palmisano, Ellis et al. 1993). Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Artificial Propagation

The artificial propagation of late-returning Chinook salmon is widespread throughout Puget Sound (Good, Waples et al. 2005). Summer/fall Chinook salmon transfers between watersheds within and outside the region have been commonplace throughout this century. Therefore, the purity of naturally spawning stocks varies from river to river. Nearly 2 billion Chinook salmon have been released into Puget Sound tributaries since the 1950s. The vast majority of these have been derived from local late-returning adults.

Returns to hatcheries have accounted for 57% of the total spawning escapement. However, the hatchery contribution to spawner escapement is probably much higher than that due to hatchery-derived strays on the spawning grounds. The genetic similarity between Green River late-returning Chinook salmon and several other late-returning Chinook salmon in Puget Sound suggests that there may have been a significant and lasting effect from some hatchery transplants (Marshall, Smith et al. 1995).

Overall, the use of Green River stock throughout much of the extensive hatchery network in this ESU may reduce the genetic diversity and fitness of naturally spawning populations (Good, Waples et al. 2005).

Hydromodification Projects

More than 20 dams occur within the region's rivers and overlap with the distribution of salmonids. A number of basins contain water withdrawal projects or small impoundments that can impede migrating salmon. The resultant impact of these and land use changes (forest cover loss and impervious surface increases) has been a significant modification in the seasonal flow patterns of area rivers and streams, and the volume and quality of water delivered to Puget Sound waters. Several rivers have been hydromodified by other means including levees and revetments, bank hardening for erosion control, and agriculture uses. Since the first dike on the Skagit River delta was built in 1863 for agricultural development (Ruckelshaus and McClure 2007), other basins like the Snohomish River are diked and have active drainage systems to drain water after high flows that top the dikes. Dams were also built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

Over the next few years, however, a highly publicized and long discussed dam removal project is expected to begin in the Elwha River. The removal of two dams in the Elwha River, a short but formerly very productive salmon river, is expected to open up more than 70 miles of high quality salmon habitat (Wunderlich, Winter et al. 1994; Ruckelshaus and McClure 2007). Estimates suggest that nearly 400,000 salmon could begin using the basin within 30 years after the dams are removed (PSAT 2007).

In 1990, only one-third of the water withdrawn in the Pacific Northwest was returned to the streams and lakes (NRC 1996). Water that returns to a stream from an agricultural irrigation is often substantially degraded. Problems associated with return flows include increased water temperature, which can alter patterns of adult and smolt migration; increased toxicant concentrations associated with pesticides and fertilizers; increased salinity; increased pathogen populations; decreased dissolved oxygen concentration; and increased sedimentation (NRC 1996). Water-level fluctuations and flow alterations due to water storage and withdrawal can affect substrate availability and quality, temperature, and other habitat requirements of salmon. Indirect effects include reduction of food sources; loss of spawning, rearing, and adult habitat; increased susceptibility of juveniles to predation; delay in adult spawning migration; increased egg and alevin mortalities; stranding of fry; and delays in downstream migration of smolts (NRC 1996).

Commercial and Recreational Fishing

Most of the commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries and by charter and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries. Recreational anglers typically use hook and line, and may fish from boat, river bank, or docks. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Pesticides are used in commercial oyster-producing areas in Willapa Bay and Grays Harbor. Currently, under a statewide SLN or Section 24 (c) registration, carbaryl can be applied to aerially applied to intertidal areas when exposed at low tide. Pesticide use is intended to control ghost shrimp (*Neotrypaea californiensis)* and mud shrimp *(Upogebia* *pugettensis).* An NPDES permit is required for application, and although the registration allows application anywhere in the state, these locations are the only ones covered under active permits. There has been some discussion of phasing out carbaryl use and replacing it with alternative pesticides.

Harvest impacts on Puget Sound Chinook salmon populations average 75% in the earliest five years of data availability and have dropped to an average of 44% in the most recent five-year period (Good, Waples et al. 2005). Populations in Puget Sound have not experienced the strong increases in numbers seen in the late 1990s in many other ESUs. Although more populations have increased than decreased since the last BRT assessment, after adjusting for changes in harvest rates, trends in productivity are less favorable. Most populations are relatively small, and recent abundance within the ESU is only a small fraction of estimated historic run size.

Atmospheric deposition

Pesticides were detected in wet deposition (rain) (Capel, Ma et al. 1998), and snow samples from Mount Rainier National Park, Washington (Hageman, Simonich et al. 2006). Three of the four most frequently detected pesticides were found in the Mount Rainier snow (dacthal, chlorpyrifos, and endosulfan).

Oregon-Washington-Northern California Coastal Drainages

This region encompasses drainages originating in the Klamath Mountains, the Oregon Coast Mountains, and the Olympic Mountains. More than 15 watersheds drain the region's steep slopes including the Umpqua, Alsea, Yaquina, Nehalem, Chehalis, Quillayute, Queets, and Hoh rivers. Numerous other small to moderately sized streams dot the coastline. Many of the basins in this region are relatively small. The Umpqua River drains a basin of 4,685 square miles and is slightly over 110 miles long. The Nehalem River drains a basin of 855 square miles and is almost 120 miles long. However, systems here represent some of the most biologically diverse basins in the Pacific Northwest (Kagan, Hak et al. 1999; Belitz, Hamlin et al. 2004; Carter and Resh 2005).

Land Use

The rugged topography of the western Olympic Peninsula and the Oregon Coastal Range has limited the development of dense population centers. For instance, the Nehalem River and the Umpqua River basins consist of less than 1% urban land uses. Most basins in this region have long been exploited for timber production, and are still dominated by forest lands. In Washington State, roughly 90% of the coastal region is forested (Palmisano, Ellis et al. 1993). Roughly 80% of the Oregon Coastal Range is forested as well (Gregory 2000). Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural (Belitz, Hamlin et al. 2004). Similarly, in the Umpqua River basin, about 86% is forested land, 5% agriculture, and 0.5% is considered urban lands. Roughly half the basin is under federal management (Carter and Resh 2005).

Habitat Modification

While much of the coastal region is forested, it has still been impacted by land use practices. Less than 3% of the Oregon coastal forest is old growth conifers (Gregory 2000). The lack of mature conifers indicates high levels of habitat modification. As such, overall salmonid habitat quality is poor, though it varies by watershed. The amount of remaining high quality habitat ranges from 0% in the Sixes to 74% in the Siltcoos (ODFW 2005). Approximately 14% of freshwater winter habitat available to juvenile coho is of high quality. Much of the winter habitat is unsuitable due to high temperatures. For example, 77% of coho salmon habitat in the Umpqua basin exceeds temperature standards.

Reduction in stream complexity is the most significant limiting factor in the Oregon coastal region. An analysis of the Oregon coastal range determined the primary and secondary lifecycle bottlenecks for the 21 populations of coastal coho salmon (Nicholas, McIntosh et al. 2005). Nicholas et al. (2005) determined that stream complexity is either the primary (13) or secondary (7) bottleneck for every population. Stream complexity has been reduced through past practices such as splash damming, removing riparian vegetation, removing LWD, diking tidelands, filling floodplains, and channelizing rivers.

Habitat loss through wetland fills is also a significant factor. Table 43 summarizes the change in area of tidal wetlands for several Oregon estuaries (Good 2000).

Table 43. Change in total area (acres^2) of tidal wetlands (tidal marshes and swamps) due to filling and diking between 1870 and 1970 (Good 2000).

Estuary	Diked or Filled Tidal Wetland	Percent of 1870 Habitat Lost
Necanicum	15	10
Nehalem	1,571	75
Tillamook	3,274	79
Netarts	16	7
Sand Lake	9	2
Nestucca	2,160	91
Salmon	313	57
Siletz	401	59
Yaquina	1,493	71
Alsea	665	59
Siuslaw	1,256	63
Umpqua	1,218	50
Coos Bay	3,360	66
Coquille	4,600	94
Rogue	30	41
Chetco	5	56
Total	20,386	72%

The only listed salmonid population in coastal Washington is the Ozette Lake Sockeye. The range of this ESU is small, including only one lake (31 km^2) and 71 km of stream. Like the Oregon Coastal drainages, the Ozette Lake area has been heavily managed for logging. Logging resulted in road building and the removal of LWD, which affected the nearshore ecosystem (NMFS 2008). LWD along the shore offered both shelter from predators and a barrier to encroaching vegetation (NMFS 2008). Aerial photograph analysis shows near-shore vegetation has increased significantly over the past 50 years (Ritchie 2005). Further, there is strong evidence that water levels in Ozette Lake have dropped between 1.5 and 3.3 ft from historic levels (Herrera 2005 in (NMFS 2008)). The impact of this water level drop is unknown. Possible effects include increased desiccation of sockeye redds and loss of spawning habitat. Loss of LWD has also contributed to an increase in silt deposition, which impairs the quality and quantity of spawning habitat.

Very little is known about the relative health of the Ozette Lake tributaries and their impact on the sockeye salmon population.

Mining

Oregon is ranked 35th nationally in total nonfuel mineral production value in 2004. In that same year, Washington was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano, Ellis et al. 1993; NMA 2007). Metal mining for all metals (*e.g.*, zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 (Palmisano, Ellis et al. 1993). Today, construction sand, gravel, Portland cement, and crushed stone are the predominant materials mined in both Oregon and Washington. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) changes in channel elevations and patterns, instream sediment loads, may result and alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment. Additionally, the severity of the effects is influenced by flood and precipitation conditions during or after the mining operations.

Hydromodification Projects

Compared to other areas in the greater Northwest Region, the coastal region has fewer dams and several rivers remain free flowing (*e.g.*, Clearwater River). The Umpqua River is fragmented by 64 dams, the fewest number of dams on any large river basin in Oregon (Carter and Resh 2005). According to Palmisano *et al.* (1993) dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat. In the past, temporary splash dams were constructed throughout the region to transport logs out of mountainous reaches. The general practice involved building a temporary dam in the creek adjacent to the area being logged, and filling the pond with logs. When the dam broke the floodwater would carry the logs to downstream reaches where they could be rafted and moved to market or downstream mills. Thousands of splash dams were constructed across the Northwest in the late 1800s and early 1900s. While the dams typically only temporarily blocked salmon habitat, in some cases dams remained long enough to wipe out entire salmon runs. The effects of the channel scouring and loss of channel complexity resulted in the long-term loss of salmon habitat (NRC 1996).

Commercial and Recreational Fishing

Most commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries, as well as by charter, and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries. Recreational anglers typically use hook and line and may fish from boat, river bank, or docks.

Atmospheric deposition

Pesticides and other chemicals may be transported through the air and later deposited on land and into waterways. For example, orthophosphate insecticides were detected in two Oregon streams, Hood River and Mill Creek (tributaries of the Columbia River). Detection occurred following periods of chemical applications on orchard crops, and may be related to atmospheric drift, mixing operations, or other aspects of pesticide use.

Environmental Protection Programs

When using carbaryl, carbofuran, and methomyl, growers must adhere to the court-ordered injunctive relief, requiring buffers of 20 yards for ground application and 100 yards for any aerial application. These measures are mandatory in all four states, pending completion of consultation.

California and Oregon both have Pesticide Use Reporting System (PURS) legislation. California PURS requires all agricultural uses of registered pesticides be reported. In this case “agricultural” use includes applications to parks, golf courses, and most livestock uses. Oregon requires reporting if application is part of a business, for a government agency, or in a public place. However, the Governor of Oregon has suggested suspending the PURS program for 2009 – 2011 due to budget shortages. A final decision will be made during the summer. If suspension occurs, PURS will resume for the 2012 growing season.

Washington State has a Surface Water Monitoring Program that looks at pesticide concentrations in some salmonid bearing streams and rivers. The program was initiated in 2003 and now monitors four areas. Three of these were chosen due to high overlap with agriculture: the Skagit-Samish watershed, the Lower Yakima Watershed, and the Wenatchee and Entiat watersheds. The final area, in the Cedar-Sammamish watershed, is an urban location, intended to look at runoff in a non-agriculture setting. It was chosen due to detection of pesticides coincident with pre-spawning mortality in Coho salmon. The Surface Water Monitoring program is relatively new and will continue to add watersheds and testing for additional pesticides over time.

Washington State also has a voluntary program that assists growers in addressing water rights issues within a watershed. Several watersheds have elected to participate, forming Comprehensive Irrigation District Management Plans (CIDMPs). The CIDMP is a collaborative process between government and landowners and growers; the parties determine how they will ensure growers get the necessary volume of water while also guarding water quality. This structure allows for greater flexibility in implementing mitigation measures to comply with both the CWA and the ESA.

Oregon has also implemented a voluntary program. The Pesticide Stewardship Partnerships (PSP) program began in 1999 through the Oregon Department of Environmental Quality. Like the CIDMP program, the goal is to involve growers and other stakeholders in water quality management at a local level. Effectiveness monitoring is used to provide feedback on the success of mitigation measures. As of 2006, there were six pilot PSPs planned or in place. Early results from the first PSPs in the Columbia Gorge Hood River and in Mill Creek demonstrate reductions in chlorpyrifos and diazinon levels and detection frequencies. DEQ’s pilot programs suggest that PSPs can help reduce contamination of surface waters.

Oregon is in the process of developing a Pesticide Management Plan for Water Quality Protection, as required under FIFRA. This plan describes how government agencies and stakeholders will collaboratively reduce pesticides in Oregon water supplies. The PSP program is a component of this Plan, and will provide information on the effectiveness of mitigation measures.

The Columbia Gorge Fruit Growers Association is a non-profit organization dedicated to the needs of growers in the mid-Columbia area. The association brings together over 440 growers and 20 shippers of fruit from Oregon and Washington. It has issued a BMP handbook for OPs, including information on alternative methods of pest control. However, their website does not mention carbamate pesticides. The mid-Columbia area is of particular concern, as many orchards are in close proximity to streams.

Idaho State Department of Agriculture has published a BMP guide for pesticide use. The BMPs include eight "core" voluntary measures that will prevent pesticides from leaching into soil and groundwater. These measures include applying pest-specific controls, being aware of the depth to ground water, and developing an Irrigation Water Management Plan.

Integration of the Environmental Baseline on Listed Resources

Collectively, the components of the environmental baseline for the action area include sources of natural mortality as well as influences from natural oceanographic and climatic features in the action area. Climatic variability may affect the growth, reproductive success, and survival of listed Pacific salmonids in the action area. Temperature and water level changes may lead to: (1) Reduced summer and fall stream flow, leading to loss of spawning habitat and difficulty reaching spawning beds; (2) increased winter flooding and disturbance of eggs; (3) changes in peak stream flow timing affecting juvenile migration; and (4) rising water temperature may exceed the upper temperature limit for salmonids at 64ºF (18ºC) (JISAO 2007). Additional indirect impacts include changes in the distribution and abundance of the prey and the distribution and abundance of competitors or predators for salmonids. These conditions will influence the population structure and abundance for all listed Pacific salmonids.

The baseline also includes human activities resulting in disturbance, injury, or mortality of individual salmon. These activities include hydropower, hatcheries, harvest, and habitat degradation, including poor water quality and reduced availability of spawning and rearing habitat for all 28 ESUs/DPSs. As such, these activities degrade salmonid habitat, including all designated critical habitat and their PCEs. While each area is affected by a unique combination of stressors, the two major impacts to listed Pacific salmonid critical habitat are habitat loss and decreased prey abundance. Although habitat restoration and hydropower modification measures are ongoing, the long-term beneficial effects of these actions on Pacific salmonids, although anticipated, remain to be realized. Thus, we are unable to quantify these potential beneficial effects at this time.

Listed Pacific salmonids and designated critical habitat may be affected by the proposed registration of carbaryl, carbofuran, and methomyl in California, Idaho, Oregon, and Washington. These salmonids are and have been exposed to the components of the environmental baseline for decades. The activities discussed above have some level of effect on all 28 ESUs/DPSs in the proposed action area. They have also eroded the quality and quantity of salmonid habitat – including designated critical habitat. We expect the combined consequences of those effects, including impaired water quality, temperature, and reduced prey abundance, may increase the vulnerability and susceptibility of overall fish health to disease, predation, and competition for available suitable habitat and prey items. The continued trend of anthropogenic impairment of water quality and quantity on Pacific salmonids and their habitats may further compound the declining status and trends of listed salmonids, unless measures are implemented to reverse this trend.

Effects of the Proposed Action

The analysis includes three primary components: exposure, response, and risk characterization. We analyze exposure and response, and integrate the two in the risk characterization phase where we address support for risk hypotheses. These risk hypotheses are predicted on effects to salmonids and designated critical habitats' PCEs. The combined analysis evaluated effects to listed Pacific salmonids and their designated critical habitat (see *Approach to the Assessment).*

Exposure Analysis

In this section, we identify and evaluate exposure information from the stressors of the action (Figure 1). We begin by presenting a general discussion of the physical and chemical properties of carbaryl, carbofuran, and methomyl that influence the distribution and persistence of action stressors in the environment and exposure of listed species and designated critical habitat (structures shown in Figure 36). Next we present general life history information of Pacific salmon and steelhead and evaluate the likely co-occurrence of action stressors with the listed Pacific salmonids. We then summarize exposure estimates presented in the three BEs and present other sources of information, including other modeling estimates and monitoring data to further characterize exposure to listed species and designated critical habitat. Finally, we conclude with a summary of expected ranges of exposure and the uncertainty contained in the exposure analysis. Because the ESA section 7 consultation process is intended to ensure that the agency action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS considers a variety of exposure scenarios in addition to those presented in EPA's BEs. These scenarios provide exposure estimates for the range of habitats utilized by listed salmonids.

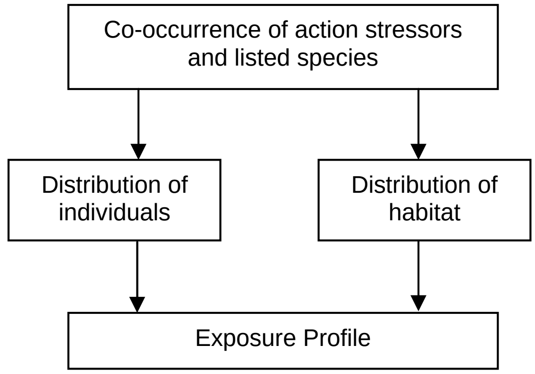
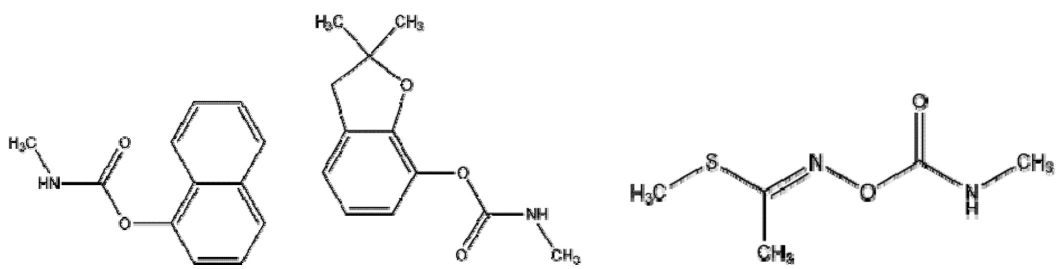


Figure 35. Exposure analysis

Summary of Chemical Fate of A.I.s



Carbaryl Carbofuran Methomyl

Figure 36. Chemical structures of carbaryl, carbofuran, and methomyl.

Carbaryl

“Carbaryl is a widely used pesticide that is commonly detected in the environment from its application in agricultural and non-agricultural settings (EPA 2003).” Carbaryl is primarily applied to terrestrial habitats, although a 24(c) registration in Washington State allows for application to commercial oyster beds to control native ghost shrimp and mud shrimp. Carbaryl can contaminate surface waters via runoff, erosion, leaching, and spray drift from application at terrestrial sites, or direct application to aquatic habitats. Carbaryl and its primary degradate, 1-naphthol, are fairly mobile and slightly persistent in the environment. Although they are not likely to persist or accumulate under most conditions, they may do so under acidic conditions with limited microbial activity.

Carbaryl dissipates in the environment by abiotic and microbially mediated degradation. The environmental fate characteristics for carbaryl are listed below (Table 44).

Table 44. Environmental fate characteristics of carbaryl (EPA 2003).

Parameter	Value
Water solubility	32 mg/L at 20 deg C
Vapor pressure	1.36 10^{-7} torrs
Henry's law constant	1.28 x 10^{-8} atm m^3 /mol
Octanol/Water partition	K_{ow} = 229
Hydrolysis ($t_{1/2}$) pH 5, pH 7, and pH 9	Stable, 12 days, 3.2 hours
Aqueous photolysis ($t_{½}$)	21 days
Soil photolysis($t_{½}$)	assumed stable
Aerobic soil metabolism ($t_{½}$)	4 days - sandy loam soil
Anaerobic soil metabolism ($t_{½}$)	72 days
Aerobic aquatic metabolism ($t_{½}$)	4.9 days
Anaerobic aquatic metabolism ($t_{½}$)	72 days
K_{oc}	177- 249 ml/g

Fish are most likely to be exposed to carbaryl through direct uptake of the chemical from the water column and across the gills, although other routes of exposure may also be important. Potential exposure routes for aquatic organisms include direct uptake from the water column or pore water of sediment, incidental ingestion of the chemical in sediment, or ingestion of the chemical in food items. EPA asserts that "carbaryl is not expected to bioaccumulate" and reports a bioacculation factor of 45 (EPA 2003). Other sources suggest accumulation of carbaryl can occur in fish, invertebrates, algae and plants. Residue levels in fish can be 140 fold greater than the concentration of carbaryl in water (http://pmep.cce.cornell.edu/profiles/extoxnet/carbaryl-dicrotophos/carbaryl-ext.html#14). In general, due to its rapid metabolism and rapid degradation, carbaryl should not pose a significant bioaccumulation risk in alkaline waters. However, under conditions below neutrality accumulation of carbaryl may be significant (http://pmep.cce.cornell.edu/profiles/extoxnet/carbaryl-dicrotophos/carbaryl-ext.html#14).

EPA reports the major degradation products of carbaryl are CO_2 and 1-naphthol, which is further degraded to CO_2 (EPA 2003). Carbaryl is stable to hydrolysis in acidic conditions but hydrolyzes in neutral environments. Hydrolysis rates increase with increasing alkalinity. Carbaryl is degraded by photolysis in water with a half-life of 21 days (d).

Under aerobic conditions, it degrades rapidly by microbial metabolism in soil and aquatic environments. Metabolism is much slower in anaerobic environments, with half-lives on the order of 2 to 3 months. Carbaryl is mobile in the environment. Sorption onto soils is positively correlated with increasing soil organic content. Because of its low octanol/water partition coefficient (K_{ow} values range from 65 to 229), carbaryl is not expected to significantly bioaccumulate (EPA 2003).

The major metabolite of carbaryl degradation by both abiotic and microbially mediated processes is 1-naphthol. This degradate represented up to 67% of the applied carbaryl in degradation studies. It is also formed in the environment by degradation of naphthalene and other PAH compounds. Data suggest 1-naphthol “is less persistent and less mobile than parent carbaryl (EPA 2003).

In a field dissipation study, carbaryl was applied on 3- to 8- ft tall pine trees in an Oregon forest. Maximum measured concentrations were 264 mg/kg on foliage at 2 d post-treatment, 28.7 mg/kg in leaf litter after 92 days, 0.16 mg/kg in the upper 15 cm of litter-covered soil at 62 d, and 1.14 mg/kg in the upper 15 cm of exposed soil at 2 d. Carbaryl was detected in the leaf litter up to 365 d after treatment and in litter-covered soil up to 302 d after treatment. Half-lives were 21 d on foliage, 75 d in leaf litter, and 65 days in soil. Carbaryl was detected at <0.003 mg/L in water and <0.003 mg/kg in sediment from a pond and stream located approximately 50 ft from the treated area (EPA 2003).

Carbofuran

Carbofuran is an *N*-methyl carbamate and is used as a broad spectrum insecticide and nematicide. Carbofuran can contaminate surface waters via runoff, erosion, leaching, and spray drift from application at upland sites.

Potential exposure routes for aquatic organisms include direct uptake of the chemical from the water column or pore water of sediment, incidental ingestion of the chemical in sediment, or ingestion of the chemical in food items. Considering that carbofuran has high water solubility, low K_{ow}, and a relatively low bioconcentration factor (2-12), fish are most likely to be exposed to carbofuran through direct uptake from the water column (Table 45). Exposure via the food chain, pore water, and sediment pathways are also possible but are less likely to be relevant for most life stages of fish (EPA 2004).

Table 45. Environmental fate characteristics of carbofuran (EPA 2004).

Parameter	Value
Water solubility	700 mg/L
Vapor pressure	6 x 10^{-7} torrs
Henry's law constant	No data
Octanol/Water partition	K_{ow} = 0.03
Hydrolysis ($t_{1/2}$) pH 6, pH 7, pH 7, & pH 9	Stable, 28 days, 3 days, and 0.8-15 hrs
Aqueous photolysis ($t_{½}$)	6 days
Soil photolysis ($t_{½}$)	78 days
Aerobic soil metabolism ($t_{½}$)	321 days
Anaerobic soil metabolism ($t_{½}$)	624 days
Aerobic aquatic metabolism ($t_{½}$)	No data
Anaerobic aquatic metabolism ($t_{½}$)	No data
K_{oc}	9 to 62 ml/g

Carbofuran is highly mobile and can leach to ground water in many soils or reach surface waters via runoff. The median K_{oc} of carbofuran is 30 and the Freundlich coefficient (K_f) ranges from 0.10 to 30.3 (EPA 2004). Major factors influencing the fate and persistence of carbofuran are water and soil pH. Carbofuran is very mobile and persistent in acidic environments, but dissipates more rapidly in pHs that are basic. Carbofuran is stable to hydrolysis at pHs < 6, but becomes increasingly susceptible to hydrolysis as the pH increases, hydrolyzing rapidly in alkaline aquatic environments (EPA 2004). A study evaluating its persistence in natural surface waters found it took 3 weeks to degrade by 50% (Sharom, Miles et al. 1980). Carbofuran phenol (7-phenol) was the only degradate detected in hydrolysis studies (EPA 2004). The rate of carbofuran degradation in soils is also pH dependent. In an acidic soil (pH 5.7), carbofuran dissipated with a half-life of 321 d, but when the soil was limed to a pH of 7.7, the half-life dropped to 149 d. The major identified degradate was 3-keto carbofuran, which peaked at 12% of the amount applied after 181 d. The other degradation products formed during photolysis, soil, and aquatic metabolism studies are 3-hydroxycarbofuran, 3-hydroxy-7-phenol, and 3-keto-7-phenol (EPA 2004).

In an aqueous photolysis study carbofuran photodegraded in neutral water (buffered [pH 7] solution) at 25ºC with a half-life of 6 d. In a soil photolysis study, carbofuran photodegraded with a half-life of 78 d on a sandy loam soil. Carbofuran is moderately persistent to microbial degradation, with half-lives on the order of a year. Near-surface photolysis is significant under laboratory conditions in aqueous solution, with a half-life on the order of days (EPA 2004).

Methomyl

Methomyl is a broad spectrum insecticide approved for a variety of terrestrial use sites. Use in aquatic habitats is not permitted. EPA characterized methomyl as moderately persistent and highly mobile suggesting methomyl may contaminate surface waters through runoff, erosion, leaching, and spray drift from application at upland sites (EPA 2003).

Potential exposure routes for fish and other aquatic organisms include direct uptake of the chemical from the water column or pore water of sediment, incidental ingestion of the chemical in sediment, or ingestion of the chemical in food items. Methomyl bioaccumulation has not been studied in fish, although it is expected to be relatively low given low octanol/water partition coefficients (K_{ow} ranges from 1.29 to 1.33, Table 46).

Methomyl degradation occurs through metabolism and photolysis. Site-specific factors affecting the persistence of methomyl include aerobicity, organic matter and soil moisture content, exposure to sunlight, and pH. Climate, crop management, soil type and other site-specific factors also influence leaching and runoff.

Table 46. Environmental fate characteristics of methomyl (EPA 2003).

Parameter	Value
Water solubility	58,000 mg/L
Vapor pressure	$1x10^{-5}$ torrs
Henry's law constant	$1.8x10^{-10}$ atm-m^3/mol
Octanol/Water partition[1]	K_{ow} =1.29 – 1.33
Hydrolysis ($t_{1/2}$) pH 5, pH 7, & pH 9	Stable, stable, & 30 days
Aqueous photolysis ($t_{½}$)	1 day
Soil photolysis ($t_{½}$)	36 days
Aerobic soil metabolism ($t_{½}$)	11 to 45 days
Anaerobic soil metabolism ($t_{½}$)	No data
Aerobic aquatic metabolism ($t_{½}$)	3.5-4.5 days[1]
Anaerobic aquatic metabolism ($t_{½}$)	<7 to 14 days
K_{oc}	24 ml/g

1Reported in (EPA 2007)

Methomyl photolysis occurs relatively quickly in water but slowly in soils. It is moderately stable to aerobic soil metabolism but degrades more rapidly under anaerobic conditions. While methomyl becomes more susceptible to hydrolysis as the pH increases above neutral, this is not expected to be a major route of dissipation under most circumstances. Laboratory studies show that methomyl does not readily adsorb to soil and has the potential to be very mobile. Dissipation from the soil surface occurs by a combination of chemical breakdown and movement. Field studies show that the varying dissipation rates for methomyl were related primarily to differences in soil moisture content, which may affect the microbial activity, and rainfall/irrigation, which could influence leaching (EPA 2003).

Several degradates of methomyl have been identified. CO_2 is a major degradate of methomyl. Methomyl oxime, is a minor degradation product in soil and water that degrades rapidly and has negligible toxicity (DuPont 2009). Another degradate, S-methyl-N-hydroxythioacetamidate, which is highly mobile, appears to be primarily a product of alkaline hydrolysis. In an aquatic metabolism study, methomyl degraded with estimated half-lives of 4-5 d. After 7 d, acetonitrile comprised a maximum of 17% and acetamide up to 14% of the amount of methomyl applied. After 102 d, volatilized acetonitrile totaled up to 27% of the parent methomyl applied and CO_2 up to 46% of the applied material (EPA 2003).

Habitats Occupied by Listed Salmonids

Listed salmonids occupy habitats that range from shallow, low flow freshwaters to open reaches of the Pacific Ocean. All listed Pacific salmonid species use freshwater, estuarine, and marine habitats. The temporal and spatial use of habitats by salmonids depends on the species and the individuals' life history and life stage (Table 47). Many migrate hundreds or thousands of miles during their lifetime, increasing the likelihood that they will come in contact with aquatic habitats contaminated with pesticides. Given that all listed Pacific salmonid ESUs/DPSs use watersheds where the use of carbaryl, carbofuran, and methomyl products is authorized, and these compounds are frequently detected in watersheds where they are used (Gilliom, Barbash et al. 2006) we expect all listed Pacific salmonid ESUs/DPSs will continue to be exposed to these compounds and other stressors of the action.

Table 47. General life histories of Pacific salmonids.

Species	General Life History Descriptions		
(number of listed ESUs or DPSs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Chinook (9)	Mature adults (usually four to five years old) enter rivers (spring through fall, depending on run). Adults migrate and spawn in river reaches extending from above the tidewater to as far as 1,200 miles from the sea. Chinook salmon migrate and spawn in four distinct runs (spring, fall, summer, and winter). Chinook salmon are semelparous (can spawn only once).	Generally spawn in the middle and upper reaches of main stem rivers and larger tributary streams.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, fry swim-up and distribute to habitats that provide refuge from fast currents and predators. Juveniles exhibit two general life history types: Ocean-type fish migrate to sea in their first year, usually within six months of hatching. Ocean-type juveniles may rear in the estuary for extended periods. Stream-type fish migrate to the sea in the spring of their second year.

Species	General Life History Descriptions		
(number of listed ESUs or DPSs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Coho (4)	Mature adults (usually two to four years old) enter the rivers in the fall. The timing varies depending on location and other variables. Coho salmon are semelparous (can spawn only once).	Spawn throughout smaller coastal tributaries, usually penetrating to the upper reaches to spawn. Spawning takes place from October to March.	Following emergence, fry move to shallow areas near stream banks. As fry grow they distribute up and downstream and establish territories in small streams, lakes, and off-channel ponds. Here they rear for 12-18 months. In the spring of their second year juveniles rapidly migrate to sea. Initially, they remain in nearshore waters of the estuary close to the natal stream following downstream migration.
Chum (2)	Mature adults (usually three to four years old) enter rivers as early as July, with arrival on the spawning grounds occurring from September to January. Chum salmon are semelparous (can spawn only once).	Generally spawn from just above tidewater in the lower reaches of mainstem rivers, tributary stream, or side channels to 100 km upstream.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry migrate downstream to estuarine areas. They reside in estuaries near the shoreline for one or more weeks before migrating for extended distances, usually in a narrow band along the Pacific Ocean's coast.
Sockeye (2)	Mature adults (usually four to five years old) begin entering rivers from May to October. Sockeye are semelparous (can spawn only once).	Spawn along lakeshores where springs occur and in outlet or inlet streams to lakes.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry migrate to nursery lakes or intermediate feeding areas along the banks of rivers. Populations that migrate directly to nursery lakes typically occupy shallow beach areas of the lake's littoral zone; a few cm in depth. As they grow larger they disperse into deeper habitats. Juveniles usually reside in the lakes for one to three years before migrating to off shore habitats in the ocean. Some are residual, and complete their entire lifecycle in freshwater.

General life histories of Pacific salmonids (continued)			
(number of listed ESUs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Steelhead (11)	Mature adults (three to five years old) may enter rivers any month of the year, and spawn in late winter or spring. Migration in the Columbia River system extends up to 900 miles from the ocean in the Snake River. Steelhead are iteroparous (can spawn more than once).	Usually spawn in fine gravel in a riffle above a pool.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry usually inhabit shallow water along banks of stream or aquatic habitats on streams margins. Steelhead rear in a wide variety of freshwater habitats, generally for two to three years, but up to six or seven years is possible. They smolt and migrate to sea in the spring.

Modeling: Estimates of Exposure to Carbaryl, Carbofuran, and Methomyl

Exposure estimates for non-crop pesticide applications

EPA's BEs indicate that pesticides containing carbaryl, carbofuran, and methomyl are registered for over 100 use sites (EPA 2003; EPA 2003; EPA 2004). As previously indicated, many of the uses identified in the BEs have since been modified, or are scheduled to be phased out or cancelled. The BEs provided relatively few estimates of exposure given the number and variety of uses authorized (Table 48). All modeled exposure estimates assumed pesticide use in agricultural crops. No estimates were provided for other uses of carbaryl, carbofuran, or methomyl.

Table 48. Examples of registered uses of carbaryl, carbofuran, methomyl and the exposure method used by EPA in BEs.

a.i.	Examples of Registered Use	Exposure Characterization in BE
Carbaryl	Crops: cranberries, cucumbers, beans, eggplant, grapefruit, grapes, hay, lemons, lettuce, nectarines, olives, onions, oranges, parsley, peaches, peanuts, pears, pecans, peppers, pistachios, plums, potatoes, prunes, pumpkins, rice, sod, spinach, squash, strawberries, sugar beets, sunflowers, sweet corn, sweet potatoes, tangelos, tangerines, tomatoes, walnuts, watermelons, and wheat.	PRZM-EXAMS 9 crops
	Targeted pests: adult mosquitoes, ticks, fleas, fire ants, and grasshoppers.	No estimates provided
	Other use sites: home and commercial lawns, flower beds, around buildings, recreation areas, golf courses, sod farms, parks, rights-of-way, hedgerows, Christmas tree plantations, oyster beds, and rural shelter belts.	No estimates provided
Carbofuran	Crops: All uses will be cancelled. Examples of current uses include Alfalfa, artichoke, banana, barley, coffee, field corn, sweet corn, pop corn, cotton, cucumber, melons, squash, grapes, oats, pepper, plantain, potato, sorghum, soybean, sugar beet, sugarcane, sunflower, wheat, cotton, spinach grown for seed, and tobacco.	PRZM-EXAMS Estimates for 5 crops
	Other use sites: All uses will be cancelled. Examples of current uses include agricultural fallow land, ornamental and/or shade trees, ornamental herbaceous plants, ornamental non-flowering plants, pine, ornamental woody shrubs, and vines.	No estimates provided
Methomyl	Crops: alfalfa, anise, asparagus, barley, beans (succulent and dry), beets, Bermuda grass (pasture), blueberries, broccoli, broccoli raab, brussels sprouts, cabbage, carrot, cauliflower, celery, chicory, Chinese broccoli, Chinese cabbage, collards (fresh market), corn, cotton, cucumber, eggplant, endive, garlic, horseradish, leafy green vegetables, lentils, lettuce (head and leaf), lupine, melons, mint, oats, onions (dry and green), peas, peppers, potato, pumpkin, radishes, rye, sorghum, soybeans, spinach, strawberry, sugar beet, summer squash, sweet potato, tomatillo, tomato, wheat, and orchards including apple, avocado, grapes, grapefruit, lemon, nectarines, oranges, peaches, pomegranates, tangelo, and tangerine.	PRZM-EXAMS Estimates for 4 crops
	Other use sites: sod farms, bakeries, beverage plants, broiler houses, canneries, commercial dumpsters which are enclosed, commercial use sites (unspecified), commissaries, dairies, dumpsters, fast food establishments, feedlots, food processing establishments, hog houses, kennels, livestock barns, meat processing establishments, poultry houses, poultry processing establishments, restaurants, supermarkets, stables, and warehouses.	No estimates provided

Exposure estimates for crop applications

The BEs provide estimated environmental concentrations (EECs) for carbaryl, carbofuran, and methomyl in surface water (Table 49). EECs were generated using the PRZM-EXAMS model and used to estimate exposure of the three a.i.s to listed salmonids and their prey (EPA 2004). PRZM-EXAMS generates pesticide concentrations for a generic “farm pond”. The pond is assumed to represent all aquatic habitats including rivers, streams, off-channel habitats, estuaries, and near shore ocean environments. EPA indicated that the PRZM-EXAMS scenarios provide “worst-case” estimates of salmonid exposure and it “believes that the EECs from the farm pond model do represent first order streams, such as those in headwaters areas” used by listed salmon (EPA 2003; EPA 2003; EPA 2004). However, listed salmonids use aquatic habitats with physical characteristics that would be expected to yield higher pesticide concentrations than would be predicted with the “farm pond” based model. Juvenile salmonids rely upon a variety of off-channel habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle (Table 47). Below we discuss the utility of the EECs for the current consultation. NMFS presents information that indicates the EECs do not represent worst-case environmental concentrations that listed Pacific salmonids may be exposed to. Finally, NMFS provides additional modeling estimates to evaluate potential exposure in vulnerable off-channel habitats used by salmonids.

Table 49. PRZM-EXAMS exposure estimates from EPA's BEs.

Scenario: crop, state	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC (μg/L)	Chronic EEC 60-d average (μg/L)[1]
Carbaryl			
Sweet corn, OH	2/aerial/8; 3.4/aerial/2	53; 46	19; 13
Field corn, OH	2/aerial/4; 1/aerial/2	47; 13	14; 4
Apples, PA	2/aerial/5; 1.2/spray blast/2	31; 12	7; 2
Sugar Beets, MN	1.5/aerial/2; 1.5/aerial/1	23; 7	6; 2
Citrus, FL	5/aerial/4; 3.4/aerial/2	153; 100	41; 23
Peaches, CA	7/aerial/2; 3.5/air blast/1	57; 14	12; 3
Citrus, CA	5/aerial/4; 3.4/aerial/2	20; 7	11; 2
Tomatoes, CA	2/aerial/4; 0.66/air blast/1	17; 2	7; 1
Apples, OR	2/aerial/5; 1.2/aerial/2	19; 3	6; 1
Blackberries, OR	2/aerial/5; 1.9/air blast/1	12; 8	6; 3
Snap beans, OR	1.5/aerial/4; 0.8/ground/1	10; 1.2	1; 0.3
Carbofuran			
Alfalfa, CA	1/foliar/1	6	3.0
Alfalfa, PA	1/foliar/1	7.9	4.1
Cotton, MS	1/in-furrow/1	11	5.5
Grapes, CA	10/soil surface/1	5.5	2.7
Potatoes, ME	1/not reported/2	26	14
Artichokes, CA	2/ground/1	35	19
Cotton, CA	1/in-furrow/1	0.8	0.4
Potatoes, ID	2/foliar/1; 3/in-furrow/1; 6/chemigation/1	6.2; 0.2; 10.4	4.0; 0.1; 6.2
Methomyl			
Lettuce	0.9/aerial/10; 0.225/aerial/15	88; 30	81; 26
Sweet corn	0.45/aerial/16	60	54
Peaches	1.8/aerial/3	99	85
Cotton	0.6/aerial/3	55	47

[1]The chronic values reported for methomyl are 56-day average concentrations rather than 60-day average concentrations.

Utility of EECs for consultation

As described in the *Approach to the Assessment* section, our exposure analysis begins at the organism (individual) level of biological organization. We consider the number, age (or life stage), gender, and life histories of the individuals likely to be exposed. This scale of assessment is essential as adverse effects to individuals may result in population-level consequences, particularly for populations of extremely low abundance. Characterization of impacts to individuals provides necessary information to assess potential impacts to populations, and ultimately to the species. To assess risk to individuals, we must consider the highest concentrations to which any individuals of the population may be exposed. Several lines of evidence discussed below suggest that

EECs in the BEs may underestimate exposure of some listed organisms and designated critical habitat.

Although EPA characterized these exposure estimates as “worst case” in the BEs, it has also acknowledged that measured concentrations in the environment sometimes exceed PRZM-EXAMs EECs (EPA 2007). EPA has subsequently clarified that rather than providing worst case estimates, PRZM-EXAMS estimates are protective for the vast majority of applications and aquatic habitats (EPA 2007). NMFS agrees that the model is designed to produce estimates of exposure that are protective for a large number of aquatic habitats.

Recent formal consultation and reviews of EPA informal consultations by the Services found that concentrations measured in surface water sometimes exceed peak concentrations predicted with PRZM/EXAMS modeling (NMFS 2007; NMFS 2008; USFWS 2008). Concentrations of carbaryl, carbofuran, and methomyl were generally less than concentrations predicted in the BEs using PRZM-EXAMs although there were examples where measured concentrations in surface water exceeded these estimates (*e.g.,* (Hurlburt 1986; Creekman and Hurlburt 1987; Tufts 1989; Tufts 1990; Beyers, Farmer et al. 1995). These findings demonstrate that the EECs generated using PRZM-EXAMS can underestimate peak concentrations that actually occur in some aquatic habitats, and therefore, peak exposure experienced by some individuals of listed species may be underestimated.

Exposure values derived by EPA using PRZM-EXAMS may underestimate or overestimate salmonid exposure to carbaryl, carbofuran, and methomyl. Two assumptions are discussed below that show salmonids may be exposed to higher concentrations than predicted with PRZM-EXAMS modeling:

*Assumption 1: Model outputs are 90th percentile time-weighted averages***.** It is important to recognize that the model predicts concentrations based on site-specific assumptions (*e.g.,* rainfall) and that environmental concentrations provided for the estimate do not represent the highest aquatic concentrations predicted given the assumptions. Rather, the exposure estimates provided in the BEs are time-weighted average concentrations for one day (*i.e.,* peak), 21 d, and 60 d. Although EPA refers to the 1 d-averages as peak concentrations, they do not represent the maximum concentration predicted. Rather, they represent the average concentration over a 24-hour (h) period. Additionally, the concentrations reported represent the 90th percentile of the estimates derived using PRZM-EXAMS (Lin 1998). Although NMFS agrees this is a relatively protective approach for evaluating exposure in most aquatic habitats, it does not represent the possible “worst case” exposure.

Assumption 2: Model inputs used the highest use rates and greatest number of applications. NMFS compiled information on the maximum use rates permitted (single and seasonal), number of applications allowed, and minimum application intervals required through EPA product labeling (Tables 1-3).

Several of the PRZM-EXAMS scenarios did not match up well with current use restrictions and did not account for maximum permitted application rates. For example, PRZM-EXAMS scenarios for carbaryl evaluated a maximum single application rate of 3.5 lb a.i./acre versus higher single application rates authorized by EPA for oyster beds, home lawn, golf course, and sod farms (8-9.1 lb a.i./acre), nut trees (5 lbs a.i./acre), and citrus (7.5 – 12 lbs a.i./acre). Some of the simulations included multiple applications that exceed single application rates. For example, there were simulations for 4 applications of 5 lbs a.i./acre in citrus and 2 applications of 7 lbs a.i./acre in peaches. However, these scenarios do not cover the maximum rate of carbaryl allowed per applications (12 lbs a.i./acre in California citrus) or annually (24 lbs a.i./acre/year in apple, pear, crabapple, and others). The maximum single application rate for methomyl is 0.9 lbs a.i./acre. One scenario assumed 3 applications of 1.8 lbs a.i./acre (California peaches). This scenario may overestimate aquatic concentrations for single applications at comparable use sites. However, considering multiple applications, the maximum application rate of methomyl evaluated in the BE was 9 lbs a.i./acre (lettuce) versus allowable application rates of 10-

32 lbs a.i./acre/year for alfalfa, broccoli, cabbage, corn, lettuce, onions, and spinach (EPA 2007).

Few crop scenarios were assessed relative to the number of approved uses. The BEs provided pesticide exposure estimates from uses in relatively few crops considering the number of registered uses of carbofuran, carbaryl, and methomyl. For example, estimates of carbaryl exposure were provided for nine agricultural crops. An evaluation of currently registered uses of a single carbaryl product label (Sevin brand XLR plus carbaryl insecticide) revealed the product can be applied to more than 75 agricultural crops in California alone (CDPR 2009). Similarly, the product Furadan 4-F Insecticide - Nematicide (carbofuran) can be applied to more than 20 use sites in California while exposure estimates evaluated only 5 agricultural crops. Finally, DuPont Lannate SP Insecticides (methomyl) is authorized for use on more than 80 agricultural crops in California while the BE provided exposure estimates for only 4 crops for all methomyl products. NMFS also reviewed aquatic exposure estimates developed by EPA to assess the risk of carbaryl and methomyl to the red legged frog (EPA 2007). Although these estimates were specific to registered uses in California only, they represent a more thorough evaluation of uses than was provided in the BEs. Estimates for agricultural crops were relatively comparable to estimates in the BEs with most scenarios predicting concentrations of carbaryl and methomyl within an order of magnitude (10-100 µg/L). Exposure estimates associated with carbaryl use in rice (2,579 µg/L) were a notable exception and this use was not evaluated in the BE. The acute EECs for carbaryl ranged from 6 – 167 µg/L for other crops assessed versus 1-153 µg/L in the BE. Acute EECs for methomyl ranged from 3 – 183 µg/L versus 30-99 µg/L in the BE.

Crop scenarios are likely not representative of the entire action area. The regional scale that the modeled scenarios are intended to represent is unclear. Many of the scenarios were conducted for states outside the distribution of listed salmonids. The methomyl BE did not provide information on geographic locations simulated (*e.g.*, county, state, region, etc.). The assumed rainfall and other site-specific input assumptions can have large impacts on predicted exposure. For example, the carbofuran

BE provides a peak EEC of 5.5 μg/L in water following an application rate of 10 lbs a.i./acre. Yet three simulations conducted at a 10-fold lower application rate produced greater estimates of exposure due to differences in site-specific assumptions (Table 49). NMFS also questions whether input assumptions were adequate to represent the range in variability among sites throughout the action area. Site-specific meteorological and soil conditions vary greatly throughout the four states where listed salmonids are distributed and crops are grown. The BEs did not indicate site-specific input assumptions of each scenario nor did they put these assumptions into perspective with regard to the range of conditions throughout the four states. This makes it difficult to determine the representativeness of scenario estimates for the complete range of crop uses.

Crop scenarios do not consider application of more than one pesticide. The pesticide labels NMFS reviewed had few restrictions regarding the co-application (*i.e.*, tank mixture applications) or sequential applications of other pesticide products containing different a.i.s. Also, there were few restrictions for those pesticides containing ingredients that share a common mode of action (*e.g.*, cholinesterase-inhibiting insecticides). For example, we saw no restrictions that would prevent either co-application or sequential application of products containing carbofuran, carbaryl, and methomyl. Examples of fish kill incidents discussed in the *Risk Characterization* section of the Opinion indicate combinations of cholinesterase-inhibiting insecticides are sometimes applied on the same day or over a short interval, increasing the likelihood of salmonid exposure to chemical mixtures that may have additive or synergistic effects. Some labels encourage the use of more than one product. The Sevin Brand XLR Plus Carbaryl Insecticide (EPA registration No. 264-333) advises that 8 applications of carbaryl at 3 d intervals "may not provide adequate levels of protection under conditions of rapid growth or severe pest pressure. The use of an alternative product should be considered in conjunction with this product." Multiple applications of pesticides increase the likelihood of cumulative exposure. We considered cumulative exposure based on generated 60 d time-weighted average concentrations to simulate situations where pesticide products containing the three a.i.s were applied at separate times during the growing season (Table 49). To address potential variability between sites, we generated exposure values for a few labeled uses using the GENEEC model, which is intended to provide screening estimates over large geographic regions (Table 50)[1]. The input parameters used were consistent with recent EPA model inputs (EPA 2006; EPA 2007; EPA 2007).

Table 50. GENEEC estimated concentrations of carbaryl, carbofuran, and methomyl in surface water adjacent to sweet corn, potatoes, sweet corn, potatoes, and citrus.

Chemical use	Rate	No.*	Interval	Buffer	EEC (μg/L)				
Foliar/ ground application	lbs/ acre		days	ft	24-h avg	4-d avg	21-d avg	60-d avg	90-d avg
Sweet Corn									
Carbaryl	2	8	3	0	397	390	348	272	229
Carbofuran	0.5	2	7	0	56	56	55	54	53
Methomyl	0.45 0.225	14 28	1 1	25 25	307 288	276 259	163 153	72 68	49 46
Potatoes									
Carbaryl	2	3	7	0	175	172	153	120	101
Carbofuran	1	2	5	0	112	112	111	108	106
Methomyl	0.9	5	5	25	212	191	113	50	34
Citrus									
Carbaryl	12	1	-	0	485	476	424	332	280
Carbofuran	-	-	-	-	-	-	-	-	-
Methomyl	0.9	3	5	25	133	120	71	31	21

*Number of applications
-Not applicable

The EECs for EPA's effect determinations were derived primarily using the PRZM-EXAMS model. This model predicts runoff to a "farm pond" based on application specifications (rate and method), properties of the a.i. (solubility, soil adsorption coefficient, soil metabolisms rate, etc.,), assumed meteorological conditions (amount of rainfall), and other site-specific assumptions [soil type, slope, etc. (EPA 2004)]. The farm pond scenario is likely a poor surrogate of certain habitats used by salmonids.

In particular, listed salmonids rely extensively upon a variety of off-channel habitats that would be expected to yield higher pesticide concentrations than would be predicted with the "farm pond" based PRZM/EXAMS model. Examples of off-channel habitats include

[1] EPA characterizes GENEEC as a tier-1 screening model (EPA 2004c). GENEEC is a meta-model of the PRZM-EXAMS model that incorporates assumptions that are intended to model exposure estimates on a site vulnerable to runoff. The size of the treated area and aquatic habitat (farm pond) are the same as described for PRZM-EXAMS.

alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, off-channel ponds, and braids (Swift III 1979; Anderson 1999). Diverse, abundant communities of invertebrates (many of which are salmonid prey items) also populate these habitats and, in part, are responsible for juvenile salmonids reliance on off-channel habitats. Juvenile coho salmon, stream-type Chinook salmon, and steelhead use off-channel habitats for extended durations (several months). Although these habitats typically vary in surface area, volume, and flow, they are frequently shallow, low to no-flow systems protected from a river's or a stream's primary flow. Thus, rearing and migrating juvenile salmonids use these habitats extensively (Beechie T. and Bolton 1999, Beechie T. J. et al. 2005, Caffrey 1996, Henning 2006, Montgomery 1999, Morley S. A. et al. 2005, Opperman and Merenlender 2004, Roni 2002).

Small streams and some off-channel habitats represent examples of habitats used by salmonids that can have a lower capacity to dilute pesticide inputs than the farm pond. The PRZM-EXAM estimates assume that a 10-hectare (approximately 25 acres) drainage area is treated and the aquatic habitat is assumed to be static (no inflow or outflow). Pesticide treatment areas of 10-hectares and larger occur frequently in agricultural crops, particularly under pest eradication programs. Additionally, aquatic habitats used by salmon vary in volume and recharge rates and consequently have different dilution capacities to spray drift and runoff events. The assumed drainage area to water volume ratio (100,000 m^2:20,000 m^3) is easily exceeded for small water bodies. For example, a one acre pond with an average depth of 1 m would exceed this ratio for treated drainage areas of approximately five acres in size and larger. The assumed aquatic habitat and size of the treated area for the PRZM-EXAMS scenarios suggest that exposure is underestimated for listed salmonids that utilize relatively small aquatic habitats with low dilution capacities.

NMFS estimates of potential exposure in shallow water habitats used by salmonids

Direct over-spray

To estimate potential exposure of salmon to pesticides in off-channel and other shallow-water habitats we first determined the initial average concentrations that will result from a direct overspray of shallow surface water. NMFS is unaware of any circumstances where EPA has authorized direct application of methomyl to aquatic habitats. However, both carbaryl and carbofuran may be applied to aquatic habitats. For example, carbaryl is registered for use in rice crops to control tadpole shrimp and other rice pests. The resulting concentrations in surface water are a function of the amount applied and the volume of the water body when pesticides are applied directly to aquatic habitats (Table 51). Carbaryl can be applied twice in rice at a rate of 1.5 lbs a.i./acre. A single application at that rate would result in an average initial carbaryl concentration of 1,682 μg/L in 10 cm of water. Similarly, EPA estimated aquatic concentrations of 2,579 μg/L associated with 2 applications of carbaryl at 1.5 lbs a.i./acre in a recent assessment for the red legged frog (EPA 2007). Specimen labels for carbaryl do not place any restrictions on rice paddy discharges, yet they warn "discharge from rice fields may kill aquatic and estuarine invertebrates (EPA Reg. No. 264-333 and 264-349)." Although the section 3 registration for rice was canceled in 1997, the BE indicates carbofuran may still be applied in rice from use of existing stock or in connection with emergency exemption requests. The BE indicates it is applied to rice fields prior to flooding at a rate of 0.5 lbs a.i./acre. Assuming this rate in 10 cm of water results in an average initial average concentration of 561 μg/L.

Carbaryl is registered for use at a rate of 2 lbs a.i./acre in cranberries. Five applications can be made at 7 d intervals. The label for Sevin brand XLR Plus warns that the "use in cranberries may kill shrimp and crabs. Do not use in areas where these are important resources (EPA Reg. No. 264-333)." However, there are no restrictions regarding applications to standing water, or label requirements to ensure bog water does not contaminate other surface waters. A single application of 2 lbs a.i./acre to water 10 cm of water results in an initial average concentrations of 2,242 μg/L.

Carbaryl can also be applied at 8 lbs a.i./acre in estuarine areas in Washington state to kill ghost shrimp and mud shrimp in commercial oyster beds. Application of 8 lbs of carbaryl to a static water body of 10 cm deep would result in an initial average concentration of approximately 9 mg/L (8,968 μg/L). This estimate does not consider tidal influences. The 24(c) label for the use of carbaryl on oyster beds does not specify when applications are to be made with relation to incoming or outgoing tides. The label does specify applicators must obtain an NPDES permit for this use. A current NPDES permit specifies that applications to these tidal areas are made when the beds are uncovered by the outgoing tide. Peak concentrations in the water column are expected as the treated area is re-flooded by the incoming tide. These concentrations are expected to decrease as the volume of water over the treated area increases. The area contaminated by the carbaryl application will increase beyond the treatment site, and water column concentrations will simultaneously decrease due to transport and dilution from the incoming tide. This use is discussed in more detail in the *Risk Characterization* portion of the Opinion (see *Field studies in ESA-listed salmonid habitat*).

Table 51. Average initial concentration of any a.i. in surface water resulting from a direct overspray of aquatic habitat.

Application Rate	Water Depth	A.i. Concentration in Surface Water
(lbs a.i. / acre)	(meters)	(μg/L)
0.25	2	14
0.5	2	28
1	2	56
3	2	168
10	2	560
0.25	1	28
0.5	1	56
1	1	112
3	1	336
10	1	1121
0.25	0.5	56
0.5	0.5	112
1	0.5	224
3	0.5	673
10	0.5	2242
0.25	0.3	93
0.5	0.3	187
1	0.3	374
3	0.3	1121
10	0.3	3736

Application Rate	Water Depth	A.i. Concentration in Surface Water
(lbs a.i. / acre)	(meters)	(μg/L)
0.25	0.1	280
0.5	0.1	560
1	0.1	1121
3	0.1	3363
10	0.1	11208

Pesticide drift

We also provide estimated pesticide concentrations in shallow off-channel habitats associated with drift from terrestrial applications of pesticides (Table 52). These estimates were derived using the AgDrift model and estimate downwind deposition from pesticide drift (Teske 2001). Additional deposition from runoff was not considered. The drift estimates derived represent mean projected drift. Although AgDrift adequately predicts drift, its field validations studies and other research show drift is highly variable and influenced by site-specific conditions and application equipment (Bird, Perry et al. 2002). No-spray buffer zones (or setbacks) may significantly reduce pesticide exposure to salmonids by reducing runoff and drift inputs.

The methomyl RED requires label statements for methomyl products that specify not to "apply by ground equipment within 25 ft, or by air within 100 ft of lakes, reservoirs, rivers, estuaries, commercial fish ponds and natural, permanent streams, marshes or natural, permanent ponds. Increase the buffer zone to 450 ft from the above aquatic areas when ultra low volume application is made (EPA 1998)." This label requirement appears to address many, but not all aquatic habitats used by listed salmonids. For example, labels specify buffers to natural and permanent streams, marshes, and ponds but there is no mention of important intermittent aquatic habitats or manmade watercourses such as floodplain restoration sites and irrigation systems that either contain listed species or drain to such habitats.

Specific buffer zones that correspond to current label requirements for carbaryl, carbofuran, and methomyl were assessed (Table 52). No buffers to aquatic habitats are required by EPA for carbaryl or carbofuran products. However, we did consider interim "voluntary" buffers recommended by CDPR to protect federally listed endangered species (http://www.cdpr.ca.gov/docs/endspec/prescint.htm). Our simulations assumed the off-channel habitat had a downwind width of 10 m. Pesticide concentrations were predicted for habitats that ranged in depths from 0.1 to 2 m. These dimensions were assumed based on research of salmonid use of off-channel habitats (Beechie et al. 2005, Henning 2006, Montgomery 1999, Morley S. A. et al. 2005, Roni 2002). Average initial concentration estimates derived from the simulations ranged from 0.2-447 μg/L for each lb of a.i. applied. These simulations indicate that applications of several lbs a.i. per acre adjacent to some off-channel habitats could result in aquatic concentrations exceeding 1 mg/L, a value that would result in substantial toxicity to aquatic life, including deaths of exposed salmonids.

Maximum rates of carbaryl permitted for most vegetable crops range from 1-2 lbs a.i./acre. Several tree crops allow much higher application rates (3-8 lbs a.i./acre), with a maximum single application rate of 12 lbs a.i./acre approved for use in California on citrus crops. Carbaryl may be applied by ground boom, chemigation, spray blast, and aerial spray applications. Considering application rates, methods, and no requirements for buffers to aquatic habitat, the estimated initial concentrations of carbaryl in surface waters range from 3 μg/L to 7 mg/L in the modeled habitats. Voluntary buffers recommended by CDPR for protection of federally listed fish would considerably reduce the deposition. For example, drift from aerial applications with the 600 ft buffer to a 10 m wide stream are predicted to be approximately equivalent to 2-6% of the applied rate versus drift equivalent to 30-39% of the applied rate with no buffer. Additionally, drift from ground application with the 120 ft buffer predict drift equivalent to 0.3 – 2% of the applied rate versus 7-23% of the applied rate predicted with no buffer.

Current carbofuran labels reviewed by NMFS specify application rates of 1 lb a.i./acre or less for most crops. Carbofuran can be applied by both ground and aerial methods. Simulations assuming no aquatic buffers at these rates provide estimates of initial average carbofuran concentrations from the low μg/L range to several hundred μg/L. The carbofuran BE indicates several 24(c) uses have previously allowed application of carbofuran at rates as high as 10 lbs a.i./acre. Simulations indicate that application rates in this range, with no buffer restrictions, would result in initial surface water concentrations of 2 μg/L to over 4,000 μg/L.

The maximum single application rate for methomyl is 0.9 lbs a.i./acre. Considering label-required buffer zones, simulations at the maximum labeled application rate predict initial average concentrations of 1-57 μg/L for ground applications and 4-84 μg/L for aerial applications. Incorporating voluntary buffers listed by CDPR predict concentrations ranging from 0.2 – 58 μg/L.

Table 52. Average initial pesticide concentration in 10 m wide off-channel habitat per lb of pesticide applied based on AgDrift simulations.

Depth of aquatic habitat (meters)	Buffer to Aquatic Habitat (ft)	Average Initial Concentration in Surface Water (μg/L)
Aerial Applications, EPA default (ASAE fine-medium droplet size distribution)		
2	0[1]	17
1	0[1]	34
0.5	0[1]	67
0.1	0[1]	333
2	100[2]	5
1	100[2]	9
0.5	100[2]	19
0.1	100[2]	93
2	600[3]	1
1	600[3]	2
0.5	600[3]	4
0.1	600[3]	18
Aerial Applications, (ASAE very fine – fine droplet distribution)		
2	0[1]	22
1	0[1]	45
0.5	0[1]	89
0.1	0[1]	447
2	450[2]	4
1	450[2]	8
0.5	450[2]	16
0.1	450[2]	79
2	600[3]	3
1	600[3]	6
0.5	600[3]	13
0.1	600[3]	64
Air Blast Applications, Dormant Spray		
2	0[1]	11
1	0[1]	21
0.5	0[1]	43
0.1	0[1]	214
2	25[2]	3
1	25[2]	6
0.5	25[2]	12
0.1	25[2]	62
2	120[3]	0.3
1	120[3]	1

Depth of aquatic habitat (meters)	Buffer to Aquatic Habitat (ft)	Average Initial Concentration in Surface Water (μg/L)
0.5	120[3]	1
0.1	120[3]	6
Ground Application, Low Boom, ASAE very fine-fine distribution, 50th percentile		
2	0[1]	4
1	0[1]	8
0.5	0[1]	15
0.1	0[1]	76
2	25[2]	1
1	25[2]	1
0.5	25[2]	2
0.1	25[2]	11
2	120[3]	0.2
1	120[3]	0.4
0.5	120[3]	1
0.1	120[3]	4
Ground Application, High Boom, ASAE very fine-fine distribution, 50th percentile		
2	0[1]	13
1	0[1]	26
0.5	0[1]	52
0.1	0[1]	261
2	25[2]	3
1	25[2]	6
0.5	25[2]	13
0.1	25[2]	64
2	120[3]	1
1	120[3]	2
0.5	120[3]	3
0.1	120[3]	17

[1]Carbaryl and carbofuran labels do not require buffers to aquatic habitats.
[2]Buffer to aquatic habitats specified on current methomyl product labels, as required by the 1998 methomyl RED.
[3]Voluntary “interim bulletin” measures developed by California Department of Pesticide Regulation to protect threatened and endangered species.

Monitoring Data: Measured Concentrations of Carbaryl, Carbofuran, Methomyl, 1-Napthol and 3-Hydroxyfuran

Data Described in USEPA’s Biological Evaluations

The BE for carbofuran (EPA 2004) summarized national scale surface water monitoring data available from the USGS NAWQA program and also provided state level summaries based on NAWQA and CDPR data. It should be noted that some USGS data are also reported in the CDPR database. National scale NAWQA data (1991-2001) were presented by land use category, and included information on reporting limits (0.02 μg/L), frequency of detections, and maximum concentrations. Maximum concentrations reported for the various land categories were: agricultural land (7 μg/L), mixed land use (0.678 μg/L), urban land use (0.034 μg/L), and undeveloped land use (0.034 μg/L). An additional table summarizing NAWQA data (1984-2004) by state (California, Idaho, Oregon, and Washington) reported a maximum concentration of 32.2 μg/L. This concentration occurred in surface water sampled from Zollner Creek, near Mt. Angel, Oregon on April 17, 2002, and was also the maximum residue found in the database on a national scale. CDPR data (1990-2003) were presented by county. Zollner Creek is a tributary to the Willamette River and within the distribution of threatened Upper Willamette River Chinook salmon and Upper Willamette River steelhead. The maximum concentration measured in California was 5.5 μg/L, in Imperial County. Date and specific location were not provided.

The BEs for carbaryl (EPA 2003) and methomyl (EPA 2003) provided less detailed information about monitoring data on a national basis. The BE for carbaryl provided a national maximum reported concentration of 5.5 μg/L, and noted that carbaryl was the second most commonly detected insecticide in the NAWQA database. It also noted there were more frequent detections and higher concentrations in urban streams than in streams draining agricultural or mixed land use areas. The BE for methomyl reported that few national level monitoring data were available, and reported data located in EPA's STORET database. The highest reported concentration nationally was 1 μg/L (in Texas).

Table 53. National Maximum Concentrations of Carbaryl, Carbofuran, and Methomyl as Reported in EPA BEs (EPA 2003a, EPA 2003b, and EPA 2004)

Pesticide	Maximum Concentration (μg/L)	Data Source	Years Reported
Carbaryl	5.5	USGS NAWQA	1991-1998
Carbofuran	32.2	USGS NAWQA	1984-2004
Methomyl	1.0	EPA STORET	Not Reported
1-Napthol	Not Reported		

USGS NAWQA Data for California, Idaho, Oregon, and Washington

We obtained updated data from the USGS NAWQA database to evaluate the occurrence of carbofuran, carbaryl, methomyl, and 1-napthol (a degradate of carbaryl) in surface waters monitored in California, Idaho, Oregon, and Washington. The database query resulted in approximately 5,000 samples in which one or more of the compounds was an analyte Approximately 350 unique sampling locations were represented. Available data covered a range of 15 years, from 1992-2007. Land uses associated with the sampling stations included agriculture, forest, rangeland, urban, and mixed use. Some stations occurred only once in the data set, others appeared multiple times across a span of years. The frequency of detection is a combination of the actual occurrence of pesticides in the water and the sampling intensity. Values reported are for a filtered water sample (0.7 micron glass fiber filter), representing the dissolved phase. Chemicals transported primarily in the particulate phase would be underreported in this data set. No sediment or tissue data were available from USGS for these compounds. Because the USGS monitoring program does not generally coordinate sampling efforts with specific pesticide applications or runoff events, detected concentrations are likely to be lower than actual peak concentrations that occur.

Summary information for quantifiable concentrations of carbaryl, carbofuran, methomyl, and 1-napthol is reported below (Table 54). Non-detects in this data set are reported as less than (“<”) the laboratory reporting level (LRL) for that sample. Summary statistics were calculated on samples not designated as (“<”). The LRL ranges reported were estimated based on “<”-qualified data. Many of the concentrations that could be quantified were designated as “E,” meaning the concentrations were estimated. These data are included in the summary statistics. Quantifiable concentrations ranged from 0.0001 – 33.5 µg/L for carbaryl, 0.0015 – 32.2 µg/L for carbofuran, 0.0039 – 0.8222 µg/L for methomyl, and 0.0007 – 1.6 µg/L for 1-napthol.

Table 54. Summary of Occurrences of Carbaryl, Carbofuran, Methomyl, and 1-Napthol in USGS NAWQA Database (1992-2007) for California, Idaho, Oregon, and Washington

Statistic	Carbaryl	Carbofuran	Methomyl	1-Napthol
Samples	4938	4545	1297	1256
Percent Detections	30%	8.5%	2.3%	8.5%
LRL range (µg/L)	0.0005-5.2	0.002-0.150	0.0044-1.22	0.007-0.09
Minimum concentration (µg/L)	0.0001	0.0015	0.0039	0.0007
Maximum concentration (µg/L)	33.5	32.2	0.8222	1.6
Arithmetic mean concentration (µg/L)	0.0943	0.3846	0.1171	0.0258
Standard deviation (µg/L)	1.11	2.13	0.19	0.15
Median concentration	0.0314	0.0318	0.0440	0.0072

Monitoring Data from California Department of Pesticide Regulation

We evaluated monitoring data available from the CDPR, which maintains a public database of pesticide monitoring data for surface waters in California (CDPR 2008). Data were available for carbaryl, carbofuran, methomyl and 3-hydroxycarborfuran (a degradate of carbofuran), but not 1-napthol. Data in the database (http://www.cdpr.ca.gov/docs/emon/surfwtr/surfdata.htm) are from multiple sources, including monitoring conducted by CDPR, USGS (data from the NAWQA program as well as other studies), state, city, and county water resource agencies; and some non-governmental or inter-governmental groups such as Deltakeeper. The CDPR requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included in the surface water database. Unlike the USGS NAWQA data set, the CDPR database may contain whole water samples as well as filtered samples. If whole water concentrations are reported for compounds that sorb significantly to the particulate phase, concentrations would appear higher than in a filtered sample, which represents only the dissolved phase. The majority of the studies, which are described in metadata available from CDPR, are not targeted at correlating water concentrations with specific application practices, with the exception of some studies evaluating rice pesticides.

Summary information for carbaryl, carbofuran, methomyl, and 3-hydroxycarbofuran is reported below (Table 55). No monitoring for 1-napthol was reported in the database. The database, last updated in June 2008, consists of approximately 270,000 data records. Each record reports a specific sampling site, date, and analyte. The number of records associated with a particular compound is indicative of monitoring intensity rather than actual occurrence in surface waters. In this database, detections below the LOQ are reported as 0 μg/L. Summary statistics were calculated on samples with values above the LOQ.

Carbaryl (5,265 records) and carbofuran (5,864 records) were the subject of greater monitoring intensity than either methomyl (1,982 records) or 3-hydroxycarbofuran (1,149 records). Quantifiable amounts of methomyl appeared in 11.0% of the samples. Carbaryl (3.6%) and carbofuran (5.9%) were quantifiable less often, and 3-hydroxycarbofuran (0.3%) was only rarely quantifiable. Quantifiable concentrations ranged from 0.0030 – 8.4 μg/L for carbaryl, 0.0066 –

5.2 μg/L for carbofuran, 0.0150 – 5.4 μg/L for methomyl, and 0.0600 – 0.1800 μg/L for 3-hydroxycarbofuran.

Table 55. Summary of Occurrences of Carbofuran, Carbaryl, Methomyl, and 3-Hydroxycarbofuran in CDPR Database (1991-2006)

Statistic	Carbaryl	Carbofuran	Methomyl	3-Hydroxycarbofuran
Samples in data set	5265	5864	1982	1149
Quantifiable samples	191	347	218	3
% of quantifications in data set	3.6%	5.9%	11.0%	0.3%
LOQ range (μg/L)	0.003-1.0	0.002	0.002-1.0	0.0058-0.69
Minimum concentration (μg/L)	0.0030	0.0066	0.0150	0.0600
Maximum concentration (μg/L)	8.4	5.2	5.4	0.18
Arithmetic mean concentration (μg/L)	0.3733	0.3197	0.2984	0.1367
Standard deviation (μg/L)	0.8999	0.5521	0.5223	0.0666
Median concentration	0.1200	0.1100	0.1550	0.1700

Monitoring Data from Washington State

Data from ~30 pesticide monitoring studies conducted in the state of Washington are included in Department of Ecology Environmental Information Management (EIM) database (http://www.ecy.wa.gov/eim/). Data in the database are from multiple sources, including state agencies, and may contain whole water samples as well as filtered samples. If whole water concentrations are reported for compounds that sorb significantly to the particulate phase, concentrations would appear higher than in a filtered sample, which represents only the dissolved phase. The EIM requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included. Some of the studies contained in this database are targeted with respect to specific pesticide uses, while others are more generalized water quality surveys. Results from a database query on carbaryl, carbofuran, methomyl, 1-napthol, and 3-hydroxycarbofuran, conducted by NMFS, are provided below (Table 56).

Included in the EIM database are monitoring efforts conducted jointly by the Washington Departments of Agriculture and of Ecology in some salmon-bearing streams. Final reports for 2003-2007 seasons are publically available on their website (http://agr.wa.gov/PestFert/natresources/SWM/default.htm). A separate summary of data from those investigations is provided below (Table 57). Water samples are not filtered, and thus concentrations reported include pesticides in both dissolved and particulate phases, although the sampling protocol specifies an attempt to avoid collection of excessive particulates (Johnson and Cowles 2003). Whole water concentrations for compounds that sorb significantly to the particulate phase will appear higher than those for a filtered sample, which represents only the dissolved phase.

The procedure for reporting in the EIM database includes reporting non-detects as the reporting limit for that particular sample, and adding a “U” data qualifier. The reporting limit was not specified in the data accessed by NMFS, thus LOQ ranges (Table 56) were estimated based on “U”-qualified data. Summary statistics were calculated on samples with values above the LOQ (*i.e.*, not qualified with a “U”).

Summary information for carbaryl, carbofuran, methomyl, 1-napthol, and 3-hydroxycarbofuran based on all data in the database is reported below (Table 56). Sampling intensity was approximately the same for the three parent compounds considered in this Opinion (~1,400 samples each), with slightly fewer sampling events for the degradates. Based on the method of reporting in the database, we cannot determine how many samples may have contained detection of any of the compounds that were below LOQs. Carbaryl appeared most often in the samples (8.5%), followed by carbofuran (4.0%), and methomyl (3.5%). The degradates were detected slightly less often when they were analytes, with 1-napthol quantifiable in 1.8% of the samples, and 3-hydroxycarbofuran quantifiable in 3.3%. Quantifiable concentrations ranged from 0.002 – 10.0 μg/L for carbaryl, 0.01 – 2.3 μg/L for carbofuran, 0.0150 – 0.17 μg/L for methomyl, 0.01-0.64 for 1-napthol, and 0.05 – 0.42 μg/L for 3-hydroxycarbofuran.

Table 56. Summary of Occurrences of Carbofuran, Carbaryl, Methomyl, 1-Napthol and 3-Hydroxycarbofuran in Surface Water - Washington EIM Database (1988-2007)

Statistic	Carbaryl	Carbofuran	Methomyl	1-Napthol	3-Hydroxy carbofuran
Samples in data set	1477	1410	1402	1173	1396
Quantifiable samples	126	57	49	23	46
% of quantifications in data set	8.5%	4.0%	3.5%	2.0%	3.3%
LOQ range (μg/L)	0.002-25.0	0.01-25.0	0.01-5.0	0.01-5.0	0.01-10.0
Minimum concentration (μg/L)	0.01	0.028	0.02	0.01	0.10
Maximum concentration (μg/L)	10.0	0.16	0.17	0.64	0.15
Arithmetic mean concentration (μg/L)	0.30	0.09	0.04	0.13	0.12
Standard deviation (μg/L)	1.40	0.07	0.05	0.19	0.04
Median concentration	0.03	0.08	0.02	0.08	0.12

Summary information for carbaryl, carbofuran, methomyl, 1-napthol, and 3-hydroxycarbofuran based on the recent studies (2003-2007) conducted by Washington Departments of Agriculture and Ecology are presented below (Table 57). All sampling was done in streams that contain listed Pacific salmonids. These data are a subset of the data listed in Table 56. Sampling intensity was approximately the same for the three parent compounds considered in this Opinion (~1,200 samples each), with slightly fewer sampling events for the degradates. Based on the method of reporting in the database, we cannot determine how many samples may have contained detection of any of the compounds that were below LOQs. Carbaryl appeared most often in the samples (4.3%), followed by 1-napthol (1.0%), and methomyl (0.7%). Carbofuran and 3-hydroxycarbofuran appeared less often (0.2% and 0.2% respectively). Quantifiable concentrations ranged from 0.01 – 10.0 μg/L for carbaryl, 0.028 – 0.16 μg/L for carbofuran, 0.015 – 0.17 μg/L for methomyl, 0.01 – 0.641 for 1-napthol, and 0.095 – 0.15 μg/L for 3-hydroxycarbofuran.

Table 57. Summary of Occurrences of Carbofuran, Carbaryl, Methomyl, 1-Napthol and 3-Hydroxycarbofuran in recent studies by Washington Department of Ecology (2003-2007)

Statistic	Carbaryl	Carbofuran	Methomyl	1-Napthol	3-Hydroxy carbofuran
Samples in data set	1,223	1,208	1,207	1,003	102
Quantifiable samples	52	3	9	10	2
% of quantifications in data set	4.3%	0.2%	0.7%	1.0%	0.2%
LOQ range (μg/L)	0.01-0.25	0.01-0.25	0.01-0.25	0.03-0.25	0.02-0.25
Minimum concentration (μg/L)	0.0100	0.0280	0.015	0.01	0.0950
Maximum concentration (μg/L)	10.0	0.16	0.17	0.641	0.15
Arithmetic mean concentration (μg/L)	0.30	0.09	0.043	0.1338	0.12
Standard deviation (μg/L)	1.40	0.07	0.050	0.189	0.04
Median concentration	0.03	0.08	0.019	0.078	0.12

Three studies in the EIM database reported sediment concentrations. One was conducted to determine the concentrations of carbaryl and 1-napthol in the sediment of Willapa Bay, and two were sediment toxicity assessments. The sediment toxicity assessments, which provided the reported values (Table 58) for carbofuran, methomyl, and 3-hydoxycarbofuran as well as some data on carbaryl and 1-napthol, had an LOQ of 100 μg/kg. The LOQs for the Willapa Bay analysis were lower, ranging from 22-37 μg/kg. Neither of the sediment toxicity studies reported detectable concentration of any of the chemicals. In the Willapa Bay study, both carbaryl (range 29-3,400 μg/kg) and 1-napthol (range 34-280 μg/kg) were detected. These data represent a very small subset of potentially contaminated sediments, and the negative results should not necessarily be interpreted to mean that these chemicals are not in the sediment.

Table 58. Summary of Occurrences of Carbofuran, Carbaryl, Methomyl, 1-Napthol and 3-Hydroxycarbofuran in Sediments in Washington EIM Database (1988-2007)

Statistic	Carbaryl	Carbofuran	Methomyl	1-Napthol	3-Hydroxy carbofuran
Samples in data set	26	26	26	69	26
Quantifiable samples	17	0	0	9	0
% of quantifications in data set	65.32%	0%	0%	13.0%	0%
LOQ range (g/kg)	22-160	100	100	22-58	100
Minimum concentration (g/kg)	29	NA	NA	34	NA
Maximum concentration (g/kg)	3,400	NA	NA	280	NA
Arithmetic mean concentration (g/kg)	762	NA	NA	122	NA
Standard deviation (g/kg)	1,166	NA	NA	75	NA
Median concentration (g/kg)	200	NA	NA	120	NA

Summary of National and State Monitoring Databases

Overall, data from the three sets of monitoring data examined by NMFS are relatively consistent, with carbaryl generally being the most frequently quantifiable parent compound. No state monitoring data were available from Oregon or Idaho. Carbaryl and carbofuran were measured in concentrations ranging from 0.0001-33.5 μg/L. Methomyl generally was measured at slightly lower concentrations, ranging from 0.004-5.4 μg/L. Both 1-napthol and 3-hydroxycarbofuran were detected in slightly lower concentrations, ranging from 0.0007-0.64 μg/L, than any of the parent compounds. Water concentrations based on these monitoring data, as anticipated, were generally lower than those based on modeling estimates.

Targeted Monitoring Studies

Numerous studies have been conducted in coordination with applications of carbaryl in Washington State to control burrowing shrimp in commercial oyster beds. Several investigators have documented water column concentrations exceeding several mg/L, with a peak detection of 27,800 μg/L (27.8 mg/L) in one estuary. Those monitoring studies are discussed in greater detail in the *Risk Characterization* portion of the Opinion.

The BEs for carbofuran (EPA 2004) and methomyl (EPA 2003) report some targeted studies evaluating pesticide concentrations in field runoff and receiving water bodies. The carbofuran BE reports concentrations of 16-28 μg/L of carbofuran in runoff from treated rice fields 26 d after flooding (Nicosia, Carr et al. 1991) as cited in (EPA 2004)). Concentrations declined to ≤5 μg/L by 37 d after flooding. The discussion does not describe when or how the carbofuran was applied prior to flooding. Other studies described merely note that carbofuran does move into surface water, and that concentrations decline over time. Ground water monitoring data for carbofuran reported in the BE included concentrations of up 4.3 μg/L in wells on Long Island, NY 20 years after use was prohibited, and a maximum reported concentration of 176 μg/L in Suffolk County, NY.

The methomyl BE reports data from monitoring studies for specific crops that appear to have been designed to approximate PRZM-EXAMS scenarios. These studies evaluated application rates ranging from 0.3 – 1.35 lb ai/acre on crops such as cantaloupe, sweet corn, apples, lettuce, and tomatoes. All of the studies included multiple applications (from 5 to 29) and short re-application intervals (1-5 d). Some studies measured concentrations in field runoff (96-1,320 μg/L), and all measured concentrations in various receiving waterbodies (4.6 – 175 μg/L).

Carbaryl is one of the pesticides used by the U.S. Department of Agriculture's Animal and Plant Health Inspection Service (APHIS) to control grasshopper infestations. Research was conducted to evaluate the effects of rangeland aerial applications of a carbaryl formulation known as Sevin-4-Oil (Beyers, Farmer et al. 1995). Fixed-wing aircraft were used to apply the carbaryl to rangeland on both sides of the Little Missouri River. Pesticide applicators were instructed to observe a 152 m no-spray buffer around the Little Missouri River. Carbaryl was applied at a rate of 0.5 lbs a.i./acre in 1991 and 0.4 lbs per acre in 1993. Surface water concentrations were monitored during a drought and non-drought year. During the study, discharge in the river ranged from 0.026 to 0.057 m^3/s in 1991 and 37.7 to 49.8 m^3/s in 1993. Concentrations of carbaryl in the river were monitored for 3 d following application (Table 59). Maximum values in surface water were measured on the day of application and peaked at 85.1 and 12.6 μg/L in 1991 and 1993. Concentrations declined but were detectable when the study was terminated 96 h post treatment. Peak values were approximately 7 fold during the drought year when flows were low, demonstrating the influence of dilution capacity on receiving water concentrations. These values represent concentrations that might be observed in similar habitats used by listed salmonids. The 85.1 μg/L represents one of the higher values observed in the monitoring data despite the low use rate (0.5 lbs a.i./acre) and no-spray buffer of almost 500 ft.

Table 59. Mean concentration (μg/L) of carbaryl in the Little Missouri River following rangeland application of Sevin-4-Oil

	Hours after application						
	1	2	4	8	12	24	96
1991	85.1	-	12.3	3.08	5.30	10.3	0.100
1993	12.0	12.6	3.84	4.01	-	4.51	5.14

-No Data

The CDPR database contained data from several studies conducted by CDPR in the 1990s evaluating concentrations of rice chemicals in the Colusa Basin Drain, Butte Slough, and the Sacramento River (referenced in their metadata as studies 17, 30, 34, and 40). Carbofuran, along with molinate, malathion, methyl parathion, and thiobencarb, was one of the analytes in all years. Some years, analytes also included other pesticides such as 2,4-D, triclopyr, and propanil (Norberg-King, Durhan et al. 1991). Sampling was conducted in March-July. Carbofuran was quantifiable in 100% of the samples for which it was an analyte. Concentrations across all years ranged from 0.087 – 2.97 μg/L (Table 60). Another study located in open literature, conducted in 1991-1992 (Crepeau and Kuivila 2000), measured peak concentrations in the Colusa River Basin of 0.6-1.1 μg/L. This study also measured concentrations in the Sacramento River at Sacramento and the Sacramento River at Rio Vista. Carbofuran concentrations in the river were typically an order of magnitude lower than the concentrations in the Colusa Basin Drain. The CDPR studies included acute biotoxicity studies (test organism not specified in metadata) and generally found no significant toxicity. However, a study on water from the Colusa Basin Drain using *Ceriodaphnia dubia* as the test organism in a Toxicity Indicator Evaluation (TIE) procedure found that an extract of Colusa Basin Drain water caused toxicity in laboratory tests (Norberg-King, Durhan et al. 1991) The procedure indicated that carbofuran and methyl parathion accounted for the toxicity of the sample, although other chemicals such as molinate and thiobencarb were present as well. The concentrations observed in surface water monitoring are, as expected; much lower than those predicted for the flooded rice paddies themselves. It should be noted that California requires rice growers to obtain discharge permits or get conditional waivers before discharging water from treated rice fields. Obtaining the permits or waivers typically requires holding periods following pesticide applications, which helps reduce pesticide concentrations in the discharge. Those conditions are not a requirement of the federal label.

Table 60. Carbofuran Concentrations in CDPR Studies of Rice Effluents (1995-1998)

Statistic	All years	1995	1996	1997	1998
Samples in data set	97	22	29	25	21
Quantifiable samples	97	22	29	25	21
% of quantifications in data set	100%	100%	100%	100%	100%
LOQ range (μg/L)	0.1-0.35	0.05-0.35	0.05-0.35	0.05-0.35	0.05-0.35
Minimum concentration (μg/L)	0.0872	0.1240	0.1620	0.1400	0.0872
Maximum concentration (μg/L)	2.97	0.7000	2.97	0.6600	1.35
Arithmetic mean concentration (μg/L)	0.5661	0.3907	0.8995	0.3998	0.4800
Standard deviation (μg/L)	0.5098	0.1465	0.7862	0.1472	0.3276
Median concentration	0.4125	0.3700	0.6100	0.4080	0.3800

Based on these studies, other targeted monitoring studies described in EPA BEs, and open literature evaluated for the Opinion rendered on chlorpyrifos, diazinon, and malathion (NMFS 2008), and the general state of knowledge regarding field runoff from pesticide applications, we anticipate the following:

- edge-of-field runoff concentrations will be higher than concentrations measured in waterbodies with substantial diluting volume,
- low-flow or runoff-dominated systems may contain the highest concentrations (approaching or exceeding modeled concentrations), and
- measured concentrations are likely to be lower than peak runoff concentrations, as sampling may not coincide with initial application and/or runoff events.

Monitoring data considerations

Surface water monitoring can provide useful information regarding real-time exposure and the occurrence of environmental mixtures. A primary consideration in evaluating monitoring data is whether the study design is sufficient to address exposure in a qualitative, quantitative, or probabilistic manner. The available monitoring studies were conducted under a variety of protocols and for varying purposes. None of the datasets discussed above were designed to capture peak concentrations of exposure for salmon, or define exposure distributions.

Of the monitoring programs discussed, only the studies conducted by the Washington State DOE were designed to evaluate exposure to listed Pacific salmonid habitats in several Washington State watersheds. This sampling program was intended to evaluate pesticide occurrence in some salmonid-bearing streams during the pesticide application seasons (Johnson and Cowles 2003).

Sample sites for this study are best characterized as integration sites selected based on the presence of the listed Yakima salmonid population (one of 17 independent populations that comprise the Middle Columbia River steelhead DPS) and high diversity and intensity of agriculture. The study design included sampling during the pesticide application season but did not target specific applications of pesticides nor did it target salmonid habitats that would be expected to produce the highest concentrations of pesticides (*e.g.*, shallow off-channel habitat in close proximity to pesticide application sites). Sampling was generally conducted on a weekly basis, so it is likely peak concentrations associated with drift and runoff events were not captured. Sampling stations included both agricultural- and urban-dominated watersheds, and some storm events are captured in the sampling. Sampling favored the detection of multiple pesticides, rather than peak concentrations in some habitats used by Middle Columbia River steelhead.

Other available monitoring data are also applicable to assessing exposure in listed salmon, but to varying degrees. Common aspects that limit the utility of the available monitoring data as accurate depictions of exposure within listed salmonid habitats include: 1) protocols were not designed to capture peak concentrations or durations of exposure in habitats occupied by listed species; 2) limited utility as a surrogate for other non-sampled surface waters; 3) lack of representativeness of current and future pesticide uses and conditions; and 4) lack of information on actual pesticide use to correlate with observed surface water concentrations.

Protocols not designed to capture peak exposure. The NAWQA monitoring studies contain the largest data set evaluated. However, these studies were designed to evaluate trends in water quality and were not designed to characterize exposure of pesticides to listed salmonids (Hirsch 1988). The NAWQA design does not result in an unbiased representation of surface waters. For example, some agricultural activities and related pesticide uses that may be very important in a particular region may not be represented in the locations sampled. Sampling from the NAWQA studies and other studies reviewed was typically not conducted in coordination with specific applications of carbaryl, carbofuran, and methomyl. Similarly, sampling was not designed with consideration to salmon distribution or to target the salmonid habitats most likely to contain the greatest concentrations of pesticides. Given the relatively rapid dissipation of these pesticides in flowing water habitats, it is not surprising that pesticide concentrations from these datasets were generally much lower than predicted by modeling efforts.

Limited applicability to other locations. Pesticide runoff and drift are influenced by a variety of site-specific variables such as meteorological conditions, soil type, slope, and physical barriers to runoff and drift. Additionally, surface water variables such as volume, flow, and pH influence both initial concentrations and persistence of pesticides in aquatic habitats. Finally, cropping patterns and pesticide use have high spatial variability. Given these and other site-specific factors, caution should be used when extrapolating monitoring data to other sites.

Representativeness of current and future uses. Pesticide use varies annually depending on regulatory changes, market forces, cropping patterns, and pest pressure. The use of cholinesterase inhibiting insecticides has declined in California over recent decades. However, pesticide use patterns change annually and may result in either increases or decreases in use of pesticide products for specific uses. There is considerable uncertainty regarding the representativeness of monitoring conditions to forecast future use of products containing the three a.i.s.

Lack of information on actual use to correlate with observed concentrations. A common constraint in the monitoring data was lack of information on actual use of pesticides containing the three a.i.s. For example, the ability to relate surface water monitoring data to the proposed action was severely hampered because information on application rates, setbacks/buffers, and applications methods associated with the monitoring were generally not reported. In most cases, the temporal and spatial aspect of pesticide use relative to sampling was not reported, further limiting the utility of the information.

Exposure to Other Action Stressors

Stressors of the action also include the metabolites and degradates of the a.i.s, other active and inert ingredients included in their product formulations, and tank mixtures and adjuvants authorized on their product labels. Below we summarize information presented in the BEs and provide additional information to characterize exposure to these stressors.

Metabolites and degradates of carbaryl, carbofuran, and methomyl

Available monitoring data for some metabolites and degrades of the three a.i.s in surface water are presented above. The BE indicates that degradation studies show 1-naphthol is found at up to 67% of the applied carbaryl. It is also formed in the environment by degradation of naphthalene and other polyaromatic hydrocarbon compounds. EPA suggests that salmonid exposure to 1-naphthol is expected, particularly in alkaline waters, but indicates exposure to aquatic organisms cannot be calculated due to lack of environmental fate and transport data for this degradate.

The major transformation product of carbofuran in water is 7-phenol (EPA 2004). The BE indicates other transformation products have the potential to reach the aquatic environment, including 3-hydroxycarbofuran and 3-ketocarbofuran, and that these typically occur in small amounts and are relatively short lived as compared to the parent. The major degradate in a soil metabolism study was 3-ketocarbofuran, which peaked at 12% of the amount applied after 181 days.

Several transformation products were also identified in the methomyl BE (EPA 2003). CO_2 was identified as the major degradate in most metabolism studies. S-methyl-N-hydroxythioacetamidate was identified as a highly mobile product of alkaline hydrolysis. In an aquatic metabolism study, after 7 d, acetonitrile comprised a maximum of 17% and acetamide up to 14% of the amount of methomyl applied. After 102 d, volatilized acetonitrile totaled up to 27% of the applied methomyl.

The BEs recognized that listed salmonids are likely exposed to several metabolites and degradates of the a.i.s. However, estimates quantifying exposure to these transformation products were not provided and remain a considerable source of uncertainty.

Other ingredients in formulated products

Registered pesticide products containing carbaryl, carbofuran, and methomyl generally include other ingredients such as carriers and surfactants. NMFS searched NPIRS and located several pesticide products that contain multiple a.i.s. NMFS located several labels of currently registered products that contain more than one a.i. (Table 61). Carbaryl is a common component in several fertilizer products used on turf. It is also commonly formulated with other pesticidal ingredients including rotenone, a botanical extract used to eradicate fish and as an insecticide. Carbaryl is formulated with malathion (another cholinesterase-inhibiting insecticide), bifenthrin (a neurotoxic synthetic pyrethroid) and copper sulfate (registered for use as an insecticide, algicide, and fungicide). Methomyl is formulated with muscalure, a fly attractant used in fly bait formulations. NMFS is not aware of any carbofuran products that contain other a.i.s.

Table 61. Examples of listed a.i.s on pesticide products containing carbaryl and methomyl.

EPA Product Registration Number	A.i.s	Other Ingredients
4-29	Basic cupric sulfate 7%, carbaryl 1.25%, rotenone 0.5%, cube resins other than rotenone 1%	90.25%
4-59	Carbaryl 0.5%, malathion 3%, captan 5.87%	90. 63%
4-122	Carbaryl 0.3%, malathion 6%, captan 11.7%	82%
4-458	Basic cupric sulfate 7%, carbaryl 2%	19%
4-333, 239-2514, 8119-5, 71096-15	Metaldehyde 2%, carbaryl 5%	93%
9198-233, 9198-234, 9198-235	Carbaryl 2.3%, Bifenthrin 0.058%	97.642%
270-255	Methomyl 1%, component of muscalure 0.025%	98.975%
2724-274	Methomyl 1%, component of muscalure 0.049%	98.951%
7319-6	Methomyl 1%, component of muscalure 0.026%	98.974%
53871-3	Methomyl 1%, component of muscalure 0.04%	98.96%

Nonylphenol (NP) and nonylphenol polyethoxylates are inert ingredients that may be part of a pesticide product formulation and are common adjuvant ingredients added during pesticide applications. NP and nonylphenol polyethoxylates are also ingredients in detergents, cosmetics, and other industrial products and are a common wastewater contaminant from industrial and municipal sources. NP has been linked to endocrine disrupting effects in aquatic systems. A national survey of streams found that NP was among the most ubiquitous organic wastewater contaminants in the U.S., detected in more than 50% of the samples tested. The median concentration of NP in streams surveyed was 0.8 μg/L and the maximum concentration detected was 40.0 μg/L (Table 62). Related compounds were also detected at a relatively high frequency (Koplin, Furlong et al. 2002).

Table 62. Detection of nonionic detergent degradates in streams of the U.S. (Koplin, Furlong et al. 2002)

Chemical	Frequency Detected	Maximum (μg/L)	Median (μg/L)
4-nonylphenol	50.6	40	0.8
4-nonylphenol monoethoxylate	45.9	20	1
4-nonylphenol diethoxylate	36.5	9	1
4-octylphenol monoethoxylate	43.5	2	0.2
4-octylphenol diethoxylate	23.5	1	0.1

We are uncertain to what degree NP and NP-ethoxylates may or may not occur in carbaryl, carbofuran, and methomyl product formulations and/or are added prior to application. Inert ingredients are often not specified on product labels. Additionally, NP and NP-ethoxylates represent a very small portion of the more than 4,000 inert ingredients that EPA permits for use in pesticide formulations (EPA 2008). Many of these inerts are known to be hazardous in their own right (*e.g.*, xylene is a neurotoxin and coal tar is a known carcinogen). Several permitted inerts are also registered a.i.s (*e.g.*, copper, zinc, chloropictrin, chlorothalonil). Inerts can be more than 50% of the mass of pesticide products, and millions of lbs of these products are applied to the landscape each year (CDPR 2007). This equates to large contaminant loads of inerts that may adversely affect salmon or their habitat. Uncertainty regarding exposure to these ingredients will be qualitatively incorporated into our analysis.

Tank Mixtures

Several pesticide labels authorize the co-application of other pesticide products and other materials in tank mixes, thereby increasing the likelihood of exposure to multiple chemical stressors. For example, Sevin XLR Plus (EPA Reg. No. 264-333), which contains carbaryl, specifies the product is compatible with a wide range of pesticides. The label indicates that when used as a fruit thinner, Sevin XLR Plus may be mixed with other fruit thinners. Although mixtures with other pesticides products are not specifically recommended on the Furadan LFR label (EPA Reg. No. 279-3310, containing carbofuran), they are not prohibited. Furadan LFR does specifically indicate that the product may be mixed with liquid fertilizer. Lannate SP (EPA Reg. No. 352-342, containing methomyl) also provides instructions for tank mixing with other products and identifies some tank mixes that are not compatible. These ingredients and the other inert ingredients in these products are considered part of the action because they are authorized by EPA's approval of the FIFRA label. Exposure, and consequently risk associated with potential ingredients in tank mixtures were not addressed in EPA's BEs and remain a significant source of uncertainty.

Environmental Mixtures

As described in the *Approach to the Assessment*, we analyze the status of listed species, in conjuction with the *Environmental Baseline* in evaluating the likelihood that action stressors will reduce the viability of populations of listed salmonids. This involves considering interactions between the stressors of the action and the *Environmental Baseline*. For example, we consider that listed salmonids may be exposed to the wide array of chemical stressors that occur in the various marine, estuarine, and freshwater habitats they occupy throughout their life cycle. Exposure to multiple pesticide ingredients is most likely in freshwater habitats and nearshore environments adjacent to areas where pesticides are used. As of 1997, about 900 a.i.s were registered in the U.S. for use in more than 20,000 different pesticide products (Aspelin and Grube 1999). Typically 10 to 20 new a.i.s are registered each year (Aspelin and Grube 1999). In a typical year in the U.S., pesticides are applied at a rate of approximately five billion lbs of a.i. per year (Kiely, Donaldson et al. 2004). Pesticide contamination in the nation's freshwater habitats is ubiquitous and pesticides usually occur in the environment as mixtures (Gilliom, Barbash et al. 2006). "More than 90% of the time, water from streams with agricultural, urban, or mixed-land-use watersheds had detections of two or more pesticides or degradates, and about 20% of the time they had detections of 10 or more (Gilliom, Barbash et al. 2006)." The likelihood of exposure to multiple pesticides throughout a listed salmonids' lifetime is great considering their migration routes and habitats occupied for spawning and rearing. In a three-year monitoring study conducted by the Washington DOE, pesticide mixtures were found to be common in both urban and agricultural watersheds (Burke, Anderson et al. 2006). An average of three pesticides was found in each sample collected on urban sampling sites with as many as nine pesticides found in a single sample. Agricultural sites averaged three to five pesticides per sample with as many as 14 pesticides being detected in a single sample (Burke, Anderson et al. 2006). Mixtures of chemicals that share a common mode or mechanism of action are of particular concern. Six to 11 million lbs of cholinesterase-inhibiting insecticides are used annually in California (CDPR 2007). One a.i., thiodicarb, degrades into methomyl. Potential effects of thiodicarb and other pesticide mixtures containing carbaryl, carbofuran, and methomyl were not addressed in the BEs.

Cholinesterase inhibiting insecticides, including carbaryl, diazinon, chlorpyrifos, and malathion are the most frequently detected mixtures in urban streams across the U.S. (Gilliom, Barbash et al. 2006).

Gilliom and others (2006) suggested that assessment of pesticide mixture toxicity to aquatic life is needed given the widespread and common occurrence of pesticide mixtures, particularly in streams, because the total combined toxicity of pesticides in water is often greater than that of any single pesticide compound. Exposure to multiple pesticide ingredients can result in additive and synergistic responses as described below in the *Risk Characterization* section. It is reasonable to conclude that compounds sharing a common mode of action cause additive effects and in some cases synergistic effects. CDPR's most recent pesticide use report indicates 6,857,530 lbs of cholinesterase-inhibiting insecticides were applied in California during 2006. Over 60 cholinesterase-inhibiting a.i.s are currently registered in California (CDPR 2007). Exposure to these compounds and other baseline stressors (*e.g.,* thermal stress) was not a consideration in the BEs. Therefore, risk to listed species may be underestimated.

Exposure Conclusions

Pacific salmon and steelhead use a wide range of freshwater, estuarine, and marine habitats and many migrate hundreds of miles to complete their life cycle. Carbaryl, carbofuran, and methomyl are commonly detected in freshwater habitats within the four western states where listed Pacific salmonids are distributed. Because the proposed action of registration of the three a.i.s for the next 15 years authorizes many of the same uses, these three a.i.s will continue to be present in the action area. Therefore, we expect some individuals within all the listed Pacific salmon and steelhead ESUs/DPSs will be exposed to these chemicals and other stressors of the action. Carbaryl can exceed several mg/L in coastal estuaries based on measured environmental concentrations. Carbaryl and carbofuran concentrations in off-channel habitat can also exceed several mg/L where buffers are lacking. Peak concentrations of methomyl are expected to be significantly lower given label restrictions that require buffers to aquatic habitats and a relatively low maximum application rate. However, model estimates indicate methomyl can reach concentrations of several hundred μg/L in surface waters given some uses that allow repeated applications at short re-application intervals. Given variable use of these pesticides across the landscape, and variable temporal and spatial distributions of listed salmonids, we expect exposure is also highly variable among individuals and populations of listed salmon. However, defining exposure and distributions of exposure among differing life stages of each independent population is complicated by several factors. Paramount among these is the uncertainty associated with the use of pesticide products containing these a.i.s. More specifically:

- Although the BEs and RED documents provide information on EPA regulatory decisions, they lack a full characterization of label-specific information needed to assess exposure (*e.g.,* application restrictions including application methods, rates, and intervals are lacking for many non-agricultural uses);

- EPA-authorized labels contain language that frequently does not provide clear boundaries on product use (*e.g.*, the maximum number of applications is commonly not specified and labels often instruct applicators to repeat applications "as necessary");

- Product labels authorize the application of chemical mixtures that are not specified or not clearly defined (*e.g.,* the ingredients of pesticide formulations are not fully disclosed, labels recommend tank mixture applications with other pesticides and adjuvants and tank mixtures with other pesticides are permitted unless specifically stated otherwise);

- Defining use of these products is highly uncertain because products are not likely to be used to the full extent permitted on the labels and historical use information is limited and may not reflect future use.

A major limitation of these assessments is that the majority of monitoring data used were not designed to determine exposure to listed salmonids, with the exception of specific studies conducted in Washington. Therefore, caution should be exercised in using these data for that purpose. Additionally, the assessments lack uncertainty analyses of the monitoring and toxicity data used, which limit the confidence in the given estimates (Warren-Hicks and Moore 1998). Given the complexity and scale of this action we are unable to accurately define exposure distributions for the chemical stressors. We assume the highest probability of exposure occurs in freshwater, and nearshore estuarine/marine environments with close proximity to areas where pesticide products containing carbaryl, carbofuran, and methomyl are applied. We considered several sources of information to define the range of potential exposure to action stressors. EPA provided a number of exposure estimates with maximum concentrations of 153, 35, and 99 μg/L predicted for registered uses of carbaryl, carbofuran, and methomyl, respectively. We generated additional exposure estimates for shallow off-channel habitats with predicted concentrations exceeding 1,000 μg/L for carbaryl and carbofuran, and a peak concentration of 58 μg/L for methomyl. Additionally, we considered monitoring data presented by EPA and from other sources which indicate comparable concentrations of carbaryl, carbofuran, and methomyl have been detected in surface waters within the four states where the listed salmon and steelhead are distributed (85, 32, and 175 μg/L, respectively).

We assume that the exposure estimates provided by EPA in the BEs and additional modeling and monitoring information provided above represent realistic exposure levels for some individuals of the listed species. Further, we assume the distribution within the range of exposures is a function of pesticide use and the duration of time listed salmonids spend in these habitats. All listed Pacific salmon and steelhead occupy habitats that could contain high concentrations of these pesticides at one or more life stages. However, the time spent in these habitats varies among species. Adult salmon and steelhead spend weeks to several months in freshwater habitats during their migration and spawning activities. Immediately after emerging from the gravel substrate and transitioning from alevins to fry, salmonids move to habitats where they can swim freely and forage. At this point in their development most salmon occupy freshwater habitats. Chum salmon are an exception. They immediately migrate downstream following emergence to nearshore environments in estuaries near the mouth of their parent stream. Upon arrival in the estuary the chum salmon fry inhabit nearshore areas at a preferred depth of 1.5-5 m. In Puget Sound, WA, surveys indicate chum salmon fry are distributed extremely close to the shoreline and concentrated in the top 6 inches of water. Therefore, chum salmon fry are less likely to be exposed to high concentrations of pesticides than other salmonids given the duration of time spent in shallow water habitats and the habitat they occupy. They may reside immediately next to the shore in estuaries for as little as one or two weeks before moving offshore or into deeper-water habitats within the nearshore environment. Sockeye salmon fry most frequently distribute to shallow beach areas in the littoral zones of lakes. They initially occupy shoreline habitats of only a few centimeters in depth before moving further off-shore and taking on a more pelagic existence. Coho salmon, Chinook salmon, and steelhead fry typically select off-channel habitats associated with their natal rivers and streams. These species are most likely to experience higher pesticide exposures given their utilization of shallow freshwater habitats as juveniles for rearing. Coho salmon and steelhead have a greater preference for the shallow habitats and rear in freshwater for more than a year.

Substantial data gaps in EPA's exposure characterization include exposure estimates associated with product uses on many crops and particularly, on non-crop uses. The highest concentrations detected in surface waters were those associated with applications to aquatic habitats. Those types of applications although mentioned, were not evaluated in EPA's BEs. Additionally, exposure estimates for other chemical stressors including other ingredients in pesticide formulations, other pesticide products authorized for co-application, adjuvants, degradates, and metabolites are not available or are non-existent. Although NMFS is unable to comprehensively quantify exposure to these chemical stressors, we are aware that exposure to these stressors is likely. We assume these chemical stressors may pose additional risk to listed Pacific salmonids. However, in order to ensure that EPA's action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS analyzes exposure based on all stressors that could result from all uses authorized by EPA's action.

Response Analysis

In this section, we identify and evaluate toxicity information from the stressors of the action and organize the information under assessment endpoints (Figure 37). The endpoints target potential effects from the stressors of the action to individual salmonids and their supporting habitats. The assessment endpoints represent biological attributes that, when adversely affected, lead to reduced fitness of individual salmonids or degrade PCEs (*e.g.*, prey abundance and water quality). We constructed a visual conceptual model to guide development of risk hypotheses and assessment endpoints to highlight potential uncertainties uncovered by our analysis of the available information. We begin the response analysis by describing the toxic mode and mechanism of action of carabaryl, carbofuran, and methomyl. Next we summarize the toxicity data presented in the three BEs and organize the information to applicable assessment endpoints (*e.g.*, survival, growth, etc.). The information provided by EPA addressed aspects of survival, growth and reproduction of aquatic species (freshwater and saltwater), as well as providing some discussion on other information found in the open literature, such as results from some field experiments and experiments that evaluated sublethal effects. NMFS is charged under the ESA to evaluate all direct and indirect effects of an action. We therefore evaluate all aspects of an action that may reduce fitness of individuals or reduce primary constituent elements of designated critical habitat. The evaluation includes information that EPA provided on survival, growth, or reproduction, but also encompasses a broader range of endpoints including behaviors, endocrine disruption, and other physiological alterations. The information we assessed is derived from published, scientific journals and information from government agency reports, theses, books, information and data provided by the registrants identified as applicants, and independent reports. Typically, the most relevant study results are those that directly measure effects to an identified assessment endpoint derived from experiments with salmonids, preferably listed Pacific salmonids or hatchery surrogates, exposed to the stressors of the action.

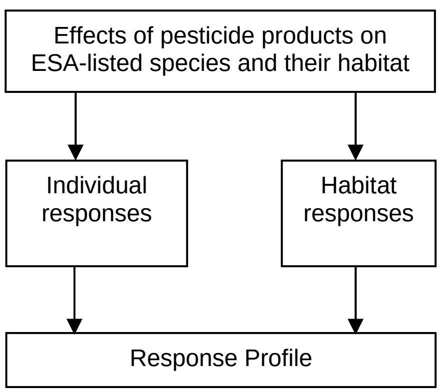


Figure 37. Response analysis

Mode and Mechanism of Action

Carbaryl, carbofuran, and methomyl share a similar mode and mechanism of toxic action and are a part of a group known as *N*-methyl carbamates. All three have similar chemical structures and act as neurotoxicants by impairing nerve cell transmission in vertebrates and invertebrates. They inhibit the enzyme AChE, which is present in cholinergic synapses. The normal function of AChE is to break down (hydrolyze) the neurotransmitter, acetylcholine, thereby serving as an "off-switch" for the electrochemical signal transmissions along nerve cells and neuromuscular junctions. AChE is prevalent in a variety of cell and organ types throughout the body of vertebrates and invertebrates (Walker and Thompson 1991). Interference of normal nerve transmission by *N*-methyl carbamates may affect a wide array of physiological systems in fish (Figure 4). Organophosphates (OPs) share this mode of action and physiological responses are similar.

The mechanism of action of *N*-methyl carbamates, inhibition of AChE, involves a series of enzyme-mediated reactions. Briefly, in a reversible reaction carbamates bind to AChE, thereby inhibiting AChE's normal activity to hydrolyze the neurotransmitter acetylcholine at nerve synapses. This reaction is similar to organophosphorus insecticides with the main exception being a carbamylation of AChE instead of a phosphorylation. Carbamate inhibition of AChE is "reversible" in cases of sublethal exposure and recovery of *N*-methyl carbamate-inhibited AChE is typically rapid compared to OP-inhibited AChE. The key result of AChE inhibition by carbamate and OP insecticides is accumulation of acetylcholine in a cholinergic synapse. The buildup of acetylcholine causes continuous nerve firing and eventual failure of nerve impulse propagation. A variety of adverse effects to organisms can result ranging from sublethal behavioral effects to death (Mineau 1991).

Incidences of acute poisoning from AChE inhibitors are prevalent for wildlife, particularly for birds and fish (Mineau 1991). The following passage describes the classic signs of AChE-inhibiting insecticide poisonings of fish:

> "Fish initially change normal swimming behavior to rapid darting about with loss of balance. This hyper excitability is accompanied by sharp tremors which shake the entire fish. The pectoral fins are extended stiffly at right angles from the body instead of showing the usual slow back and forth motion normally used to maintain balance. The gill covers open wide, and opercular movements become more rapid. With death the mouth is open and the gill covers are extended. Hemorrhaging appears around the pectoral girdle and base of the fins (Weiss and Botts 1957)."

Numerous reports, peer-reviewed journal articles (Williams and Sova 1966; Holland, Coppage et al. 1967; Coppage and Matthews 1974; Rabeni and Stanley 1975; Haines 1981; Antwi 1985) as well as multiple reviews, text books (Mineau 1991; Smith 1993; Geisy, Solomon et al. 1999), and wildlife poisoning cases document inhibition of AChE activity in exposed invertebrates

(Detra and Collins 1986; Detra and Collins 1991) and vertebrates including salmonids following exposures to carbamates and OPs (Beyers and Sikoski 1994, Eder et al. 2007, Grange 2002, Hoy et al. 1991, Laetz et al. 2009, Li H. X. et al. 2008, Li S. N. and Fan 1996, Liu et al. 2007, Sandahl J. F. et al. 2004, Sandahl J.F. et al. 2005, Scholz et al. 2006, St. Aubin 2004, Tierney et al. 2007, Yi et al. 2006, Zinkl et al. 1987).

One highly relevant study measured inhibition of brain AChE, duration of recovery, survival at 24 h, and tissue concentrations in juvenile rainbow trout (*O. mykiss*) following exposure to carbaryl at 0, 250, 500, 1,000, 2,000, and 4,000 μg/L (Zinkl, Shea et al. 1987). Rainbow trout showed dose-dependent AChE inhibition from 61 to 91 % when exposed to 250 – 4,000 μg/L for 24 h. Most trout that died had 85% or greater inhibition. Trout recovered AChE activity following 24 h in uncontaminated water, indicating that fish recover if given the opportunity following carbamate exposures (Zinkl, Shea et al. 1987). This study showed that carbaryl is acutely toxic to rainbow trout in a matter of hours (incidences of death at 1.5 – 4 h) at concentrations at or above 1,000 μg/L.

pH and toxicity

We located several data sets on toxicity to freshwater fish and aquatic invertebrates exposed to carbaryl and carbofuran at different pHs (Mayer and Ellersieck 1986). Acute toxicity of carbaryl and carbofuran increases as pH increased, based on the available freshwater fish assays (Mayer and Ellersieck 1986). This largely is considered a result of the influence pH has on the creation of hydrolysis products (Mayer and Ellersieck 1986). However, the persistence of carbaryl and carbofuran is reduced as pH increases. As noted in the exposure section, degradation half-lives can vary from hours (in alkaline waters) to weeks (in acidic waters) depending on pH. Within the Pacific Northwest and California pH varies seasonally and typically may range from 6 – 9. For methomyl, pH seems to have less of an influence on hydrolysis rates. Due to the influence of pH on persistence of carbaryl and carbofuran, we evaluated reported toxic effects within the context of pH if provided and discuss pH in relationship to salmonid habitat utilization.

Temperature and toxicity

We found no consistent correlation with temperature and toxicity of the three *N*-methyl carbamates.

Studies with mixtures of AChE inhibiting insecticides

Because the three carbamates share a common mechanism of action, are registered for use and applied in the same watersheds, and have demonstrated additive and synergistic effects in aquatic organisms, we evaluate the response of salmonids and their habitat not just from exposure to single carbamates, but also to common mixtures of *N*-methyl carbamates. We therefore include an analysis of combinations of carbaryl, carbofuran, and methomyl based on additive toxicity observed in recent publications with Chinook and coho salmon (discussed in the *Risk Characterization* section) (Scholz, Truelove et al. 2006; Laetz, Baldwin et al. 2009). Because NMFS identified mixture effects to listed salmonids as a critical data gap in understanding the occurrences of multiple insecticides in salmonid habitats, a number of the experiments described below were conducted by researchers at or associated with NOAA's Northwest Fisheries Science Center.

One of the earliest mixture studies available evaluated bluegill survival following a range of exposure durations (24, 48, 72, or 96 h) to binary combinations of 19 insecticide mixtures (Macek 1975). Carbaryl and several of the OP pesticides were tested. Macek (1975) used the equation AB/(A+B) = X to calculate mixture toxicity; where AB was the number of dead fish from a mixture of pesticides A and B, and A + B was the sum of dead fish from A and B alone. The resulting ratios, X, were designated by the author as less than additive for a ratio of less than 0.5, additive when the ratio fell between 0.5 and 1.5, and synergistic for a ratio of more than 1.5. Carbaryl containing mixtures resulted in additive toxicity for some compounds (DDT, methoxychlor, parathion, methyl parathion) and synergistic toxicity for other compounds (malathion, copper sulfate). Antagonism is when the cumulative toxicity of a mixture is less than additive. In this study, mixtures containing carbaryl were not antagonistic. Differences in classification between additive and synergistic combinations should be interpreted cautiously, as the threshold for synergism was arbitrarily set at 1.5 by the authors. Mixture results with DDT and toxaphene were 1.31 and 1.14, respectively. The binary combination of diazinon and parathion was classified as synergistic toxicants, (*i.e.*, more fish died than predicted based on an additive response). Validation of chemical concentrations with analytical chemistry was not conducted. Although the lack of raw data makes it difficult to determine exact concentrations tested and the lack of analytical confirmation precludes determination of precise concentrations, the study shows that binary combinations containing carbaryl exhibit greater toxicity than exposure to carbaryl alone. Additionally, the majority of pesticide-containing mixtures tested resulted in either additive or synergistic responses including mixtures containing carbamate and OP insecticides.

Additive toxicity of binary combinations of carbamates and OPs at a cellular level was demonstrated from *in vitro* experiments with Chinook salmon (Scholz, Truelove et al. 2006). Carbaryl and carbofuran, in addition to the oxons of diazinon, chlorpyrifos, and malathion caused additive toxicity as measured by AChE inhibition in salmonid brain tissue (Scholz, Truelove et al. 2006). Further, the joint toxicity of the mixtures could be accurately predicted from each insecticide's toxic potency, simply by adding the two potencies together at a given concentration. Since the experiments were conducted using *in vitro* exposures with the oxon degradates and not with the parent compounds, the authors conducted subsequent experiments to investigate whether additive toxicity as measured by AChE inhibition also occurred when live, juvenile coho salmon were exposed for 96 h to the parent compounds, *i.e.*, *in vivo* exposures.

The results of the second set of experiments were unexpected by the authors (Laetz, Baldwin et al. 2009). Measured AChE inhibition from some of the binary combinations was significantly greater than the expected additive toxicity, *i.e.*, synergistic toxic responses were found (Laetz, Baldwin et al. 2009). As with the *in vitro* study, brain AChE inhibition in juvenile coho salmon (*O. kisutch*) exposed to sublethal concentrations of the carbamates carbaryl and carbofuran, as well as the OPs chlorpyrifos, diazinon, and malathion, was measured (Laetz, Baldwin et al. 2009). Dose-response data for individual chemicals were normalized to their respective EC_{50} concentrations (AChE activity compared to control) and collectively fit to a non-linear regression. The regression line was used to determine whether toxicological responses to binary mixtures were antagonistic, additive, or synergistic. No binary mixtures resulted in antagonism. Additivity and synergism were both observed, with a greater degree of synergism at higher exposure concentrations. Moreover, certain combinations of OPs were lethal at concentrations that were sublethal in single chemical trials. Concentrations of each insecticide are listed in Table 63. Based on a default assumption of dose-addition, the five pesticides were combined in all possible pairings to yield target levels of AChE inhibitions in the brains of exposed coho salmon.

Table 63. Concentrations (μg/L) of insecticides used in mixture exposures. EC50s were calculated from dose-response data of AChE activity using non-linear regression. Coho salmon exposed to 1.0, 0.4, or 0.1 EC50 treatments had an equipotent amount of each carbamate or OP within the treatment *e.g.,* to attain the 1.0 EC50 treatment for diazinon and chlorpyrifos, 1.0 μg/L of chlorpyrifos (0.5 the EC50) was combined with 72.5 μg/L of diazinon (0.5 of the EC50).

Insecticide	Measured EC50	Concentration of each ingredient in binary combination to achieve treatment level		
		1.0 EC50 units	0.40 EC50 units	0.10 EC50 units
Carbaryl	145.8	72.9	29.2	7.3
Carbofuran	58.4	29.2	11.7	2.9
Chlorpyrifos	2.0	1.0	0.4	0.1
Diazinon	145.0	72.5	29.0	7.3
Malathion	74.5	37.3	14.9	3.7

As determined by the regression, these levels of enzyme inhibition would result from exposure to 0.1, 0.4, and 1.0 EC50 units, respectively. Two thirds (20/30) of pesticide pairs yielded AChE levels that were significantly lower, *i.e.*, indicative of synergism, than would be expected based on additivity *i.e.*, dose-addition (t-test with Bonferroni correction, $p < 0.005$). The number of combinations that were statistically synergistic increased with increasing exposure concentrations. All pairings analyzed at 1.0 EC50 showed synergism in AChE inhibition.

The combination of carbaryl at 79.9 μg/L and carbofuran at 29.2 μg/L to achieve 1.0 EC50 produced synergistic toxicity, however no incidences of coho mortality were observed. At 0.1 and 0.4 EC50 levels with the two carbamates, synergistic toxicity was observed, although the deviation from the predicted additive response was not statistically significant. In binary mixtures containing an OP and a carbamate synergistic toxicity occurred with some of the combinations at 0.1 and 0.4 EC50 treatments. The mechanism for synergistic toxicity in salmonids is unknown.

Additionally, pairings of two OPs produced a greater degree of synergism than mixtures containing one or two carbamates. This was particularly true for mixtures containing malathion coupled with either diazinon or chlorpyrifos. At the highest exposure treatment, 1.0 EC50 (malathion at 37.3, chlorpyrifos at 2, diazinon at 72.5 μg/L), binary combinations produced synergistic toxicity. Many fish species die following high rates of acute brain AChE inhibition, between 70-90% (Fulton and Key 2001).

Coho salmon exposed to combinations of diazinon and malathion (1.0 and 0.4 EC_{50}) as well as chlorpyrifos and malathion (1.0 EC_{50}) all died (Laetz, Baldwin et al. 2009). This result was in contrast to the predicted AChE inhibition from *in vitro* tests with Chinook salmon. Coho exposed to these OP mixtures showed toxic symptoms of inhibition of AChE, including loss of equilibrium, rapid gilling, altered startle response, and increased mucus production before dying. OP combinations were also synergistic at the lowest concentrations tested. Diazinon and chlorpyrifos were synergistic when combined at 7.3 μg/L and 0.1 μg/L, respectively. The pairing of diazinon (7.3 μg/L) with malathion (3.7 μg/L) produced severe (> 90%) AChE inhibition, including classical signs of poisoning as well as death. We expect that juvenile salmonids exposed to these effect concentrations in the environment will respond similarly.

Multiple studies indicate compounds that share a common mode of action frequently result in additive and at times synergistic responses in aquatic organisms (Anderson T. D. and Lydy 2002, Belden et al. 2007, Bocquene et al. 1995, Jin-Clark et al. 2002, Laetz et al. 2009, Lydy and Austin 2005, Macek 1975, Monserrat et al. 2001, Scholz et al. 2006). Unfortunately, we are unable to create a predictive model of synergistic toxicity as dose-response relationships with multiple ratios of pesticides are not available at the present time and the mechanism of synergism remains to be determined. That said, we conducted a mixture analysis with carbaryl, carbofuran, and methomyl based on additive toxicity with the caveat that synergism is likely where circumstances mirror the experimental conditions of this study, *i.e.,* similar exposure durations and pesticide concentrations. This is a reasonable approach based on the current state of the science. We used the mixture work of Laetz, Baldwin et al. (2009) and Scholz, Truelove et al. (2006) to construct mixture dose-response relationships predicated on additivity (see mixture analyses in the *Risk Characterization* section).

Summary of Toxicity Information Presented in the Biological Evaluations

Each BE primarily summarized acute and chronic toxicity data from "standardized toxicity tests" submitted by pesticide registrants during the registration process, tests from government laboratories available in EPA databases, or from published, peer-reviewed scientific publications (books and journals). The assessment endpoints from these tests for an individual organism generally included aspects of survival (death), reproduction, and growth measured in laboratory dose-response experiments (EPA 2004). Survival is measured in both acute and chronic tests. Reproduction and growth are generally measured and reported in the chronic tests. Population-level endpoints and analyses were generally absent in the BEs, other than a few measurements of fish and aquatic invertebrate reproduction. Adverse effects to organisms were not translated into consequences to populations. The BEs also presented some information on multispecies microcosm and mesocosm studies. For this Opinion, NMFS translates effects to individual salmonids into potential population-level consequences as explained in the *Risk Characterization* portion of the *Effects of the Proposed Action* section, and ultimately draws a conclusion on the likely risk to listed salmonids based on exposure and anticipated individual and population-level effects.

Survival of individual fish is typically measured by incidences of death following 96 h exposures (acute test) and incidences of death following 21 d, 30 d, 32 d, and "full life cycle" exposures (chronic tests) to a subset of freshwater and marine fish species reared in laboratories under controlled conditions (temperature, pH, light, salinity, dissolved oxygen, etc.,) (EPA 2004). Lethality of the pesticide is usually reported as the median lethal concentration (LC50), the statistically-derived concentration sufficient to kill 50% of the test population. For aquatic invertebrates it may be reported as an EC50, because death of these organisms may be difficult to detect and immobilization is considered a terminal endpoint. An LC50 is derived from the number of surviving individuals at each concentration tested following a 96 h exposure and is typically estimated by probit or logit analysis and recently by statistical curve fitting techniques. In FIFRA guideline tests, LC50s are typically calculated by probit analysis. If the data are not normally distributed for a probit analysis, than either a moving average or binomial is used, resulting in no slope being reported. Ideally, to maximize the utility of a given LC50 study, a slope, variability around the LC50, and a description of the experimental design- such as experimental concentrations tested, number of treatments and replicates used, solvent controls, etc.- are needed. The slope of the observed dose-response relationship is particularly useful in interpolating incidences of death at concentrations below or above an estimated LC50. The variability of an LC50 is usually depicted by a confidence interval (95% CI) or standard deviation/error and is illustrative of the degree of confidence associated with a given LC50 estimate *i.e.*, the smaller the range of uncertainty the higher the confidence in the estimate. Without an estimate of variability, it is difficult to infer the precision of the estimate. Furthermore, survival experiments are of most utility when conducted with the most sensitive life stage of the listed species or a representative surrogate. In the case of ESA-listed Pacific salmonids, there are several surrogates that are available for toxicity testing including hatchery reared coho salmon, Chinook salmon, steelhead, and chum salmon, as well as rainbow trout[2]. The available toxicity data include a varitety of salmonids. Unfortunately, slopes, estimates of variability for an LC50, and experimental concentrations frequently are not reported. In our review of the BEs, we did not locate any reported slopes of dose-response curves, although some of this information was presented in some of the corresponding Science Chapters. Consequently, we must err on the side of the species in the face of these uncertainties and select LC50s from the lower range of available salmonid studies. We selected LC50s and associated slopes as input in the population modeling exercises discussed later. We evaluate the likelihood of concentrations that are expected to kill fish and apply qualitative and quantitative methods to infer population-level responses of ESA-listed salmonids within the *Risk Characterization* section (Figure 2).

Growth of individual organisms is an assessment endpoint derived from standard chronic fish and invertebrate toxicity tests summarized in the BEs. It is difficult to translate the significance of impacted growth derived from a guideline study to fish growth in aquatic ecosystems. The health of the fish, availability and abundance of prey items, and the ability of the fish to adequately feed are not assessed in standard chronic fish tests. These are important factors affecting the survival of wild fish. What is generally assessed is size or weight of fish measured

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[2] Rainbow trout and steelhead are the same genus species (*Oncorhynchus mykiss*), with the key differentiation that steelhead migrate to the ocean while rainbow trout remain in freshwaters. Rainbow trout are therefore good toxicological surrogates for freshwater life stages of steelhead, but are less useful as surrogates for life stages that use estuarine and ocean environments.

at several times during an experiment. The test fish are usually fed twice daily, *ad libitum, i.e.,* an over abundance of food is available to the fish. Therefore, any reductions in size are a result of fish being affected to such an extent that they are not feeding even when presented with an abundance of food. Subtle changes in feeding behaviors or availability of food would not be detected from these types of experiments. If growth is affected in these experiments, it is highly probable that growth of fish in natural aquatic systems would be severely affected. If effects to growth are likely, we assess salmonid population-level consequences based on reductions in juvenile growth and subsequent reduction in size prior to ocean entry.

Reproduction, at the scale of an individual, can be measured by the number of offspring per female (fecundity), and at the scale of a population by measuring the number of offspring per females in a population over multiple generations. The BEs summarized reproductive endpoints at the individual scale from chronic freshwater fish experiments where hatchability and juvenile and larval survival are measured. NMFS considers many other assessment measures of reproduction, including egg size, spawning success, sperm and egg viability, gonadal development, reproductive behaviors, and hormone levels. These endpoints are not generally measured in standardized toxicity assays used in pesticide registration.

Sometimes qualitative observations of sublethal effects are summarized from 96 h lethality dose-response bioassays in EPA's risk assessments. These observations generally were limited in the BEs for carabaryl, carbofuran, and methomyl, and when noted, pertained to unusual swimming behaviors. None of these behaviors were rigorously measured and therefore are of limited value in assessing the effects of the three insecticides on Pacific salmonids. We do, however note a few of the observations when they pertained to a relevant assessment endpoint, such as impaired swimming. Some BEs presented toxicity information on degradates, metabolites, and formulations. However, toxicity information on other or "inert" ingredients found in pesticide formulations was usually not presented.

Results from multiple species tests, called microcosm and mesocosm studies, were also discussed in the BEs to a varying degree. These types of experiments are likely closer approximations of potential ecosystem-level responses such as interactions among species (predator-prey dynamics), recovery of species, and indirect effects to fish. However, the interpretation of results is complicated by how well the results represent natural aquatic ecosystems and how well the studies apply to salmonid-specific assessment endpoints and risk hypotheses. These studies typically measured individual responses of aquatic organisms to contaminants in the presence of other species. Some are applicable to questions of trophic effects and invertebrate recovery, as well as providing pesticide fate information. The most useful mesocosm study results for this Opinion are those that directly pertain to identified assessment endpoints and risk hypotheses. We discuss study results in the context of salmonid prey responses, emphasizing survival and recovery of prey taxa as well as shifts from preferred taxa to other taxa if measured. One of the notable limitations of these study types is they do not take represent real world aquatic ecosystems that are degraded from various stressors including contaminants and elevated water temperature..

Results from aquatic field studies were generally not discussed in great detail within the BEs. We discuss field studies that evaluated identified assessment endpoints, particularly those which address salmonid prey responses in systems with ESA-listed salmonids.

Ranges in toxicity values presented in the BEs for each a.i. are summarized in Table 64. Ranges in toxicity values (μg/L) are organized by assessment endpoint and associated assessment measures. The BEs provided toxicity information from EPA's EFED Pesticide Toxicity Database and from the ACQUIRE database.

Table 64. Assessment endpoint toxicity values (μg/L) presented in BEs and REDs for carbaryl, carbofuran, and methomyl

Assessment Endpoint		carbaryl (μg/L)			carbofuran (μg/L)			methomyl (μg/L)		
	Assessment measure	> 95% a.i.	< 95% a.i.	degradate (s)	> 95% a.i.	< 95% a.i.	degradate (s)	> 95% a.i.	< 95% a.i.	degradate (s)
							--			462,000 n=1
							--			--
	NOEC/LOEC		--	--	rainbow trout 24.8/ 56.7 n=1 (larval survival[1])	--	--	fathead minnow 57/117 n=1 (larval survival[1])	--	--
		--	--	--	rainbow trout[2] 24.8/ 56.7 n=1	--	--	fathead minnow 76,117/ 142,243 n=2	--	--
Habitat-salmonid prey	invertebrate survival	1.7 - 26 n=5	4.3 - 13.0 n=5	200-2,100 n=5	2.2 – 2,700 n=3	41 n=1	--	8.8 - 920 n=8	7.6 - 720 n=6	--
			--	--	--	--	--		--	--
	invertebrate reproduction	1.5-3.3 n=1	--	--	9.8/ 27 n=1	--	--	1.6- 3.1 n=1	--	--

1. Larval survival derived from 28 day fish assays
2. Juvenile growth measured at 60, 75, 90 d exposure
n is the number of studies

Assessment endpoint: Fish survival

Assessment measure: 96 h survival from laboratory bioassays reported as an LC50.

Carbofuran is the most toxic of three insecticides based on fish survival values (LC50s) followed by carbaryl and methomyl, respectively (Table 64). All three carbamates have a range of acute freshwater fish LC50s spanning 1 to 2 orders of magnitude. EPA reported the following ranges of LC50s: carbofuran LC50s ranged from 88-3,100 μg/L; carbaryl LC50s ranged from 250-290,000 μg/L; and methomyl LC50s ranged from 300-7,700 μg/L. Based on these LC50 ranges, EPA classified these insecticides as "highly toxic" to "moderately toxic". Salmonids were well represented in the data set, with 11 results for carbaryl, 11 for carbofuran, and 13 for methomyl. A cumulative frequency distribution of carbaryl LC50s for freshwater fish indicated that Atlantic salmon were the most sensitive of the species tested, and salmonids as a group were much more sensitive than fathead minnow and bluegill sunfish (EPA 2003).

EPA classified the three carbamates as very highly toxic to moderately toxic to estuarine and salt water species depending on the chemical and the fish species tested. Carbofuran LC50s for marine and estuarine fish ranged from 33 μg/L to more than 100 μg/L (n=5), indicating that saltwater/estuarine species were more sensitive to carbofuran than salmonids tested in freshwater. Two LC50s (2,200 and 2,600 μg/L, both sheepshead minnow) were reported for carbaryl and one LC50 (1,160 μg/L, sheepshead minnow) was reported for methomyl. Based on available data presented in EPA documents, it is uncertain whether marine and estuarine species are more sensitive, less sensitive, or equally sensitive to carbaryl and methomyl compared to freshwater fish. No data were presented on salmonids exposed in saline environments.

Assessment endpoint: Reproduction

Assessment measure: Number of offspring, hatchability, number of fish that attained sexual maturity by 136 d, and number of spawns per spawning pair

For carbaryl, one chronic study was listed in the BE (EPA 2003) and briefly discussed in the Science Chapter (EPA 2003), which evaluated a variety of assessment endpoints of fathead minnows including reproductive endpoints (referenced in (EPA 2003) as TOUCAR05 Carlson 1972, although this is an erroneous citation of (Carlson 1971)). Reproductive endpoints measured included number of mature males, number of mature females, number of immature fish, number of eggs per mature female, number of eggs spawned, and hatchability of eggs (Carlson 1971). Fathead minnows were exposed to five treatments (8, 17, 62, 210, and 680 μg/L [analytically verified]) of carbaryl in a flow through system for nine months; capturing the life cycle of the fathead minnow. Fathead minnows showed reduced number of eggs per female and reduced number of eggs spawned when exposed to 680 μg/L and of the eggs spawned, none hatched (Carlson 1971).

The carbofuran BE (EPA 2004) stated that no full life cycle fish tests were available, and as it did not report any early-life stage test results, presumably none of those were available either, as the only data reported were for a partial life cycle test for rainbow trout (*O. mykiss*). Reported NOAECs were based on growth effects, and no reproductive endpoints were discussed. A definitive reference to the study was not provided in the BE. The carbofuran Science Chapter (EPA 2005) reports on an early life stage test for rainbow trout, referring to it as Acc.# GEOCAR08. We were unable to locate a specific citation for this study in the references but the reported NOAEC (24.8 μg/L) and LOAEC (56.7 μg/L) were the same as the unreferenced study discussed in the BE. However, the Science Chapter stated that larval survival was the most sensitive endpoint of those evaluated, and that scoliosis was observed in the larval fish at carbofuran concentrations ≥56.7 μg/L. The Science Chapter also references a sheepshead minnow (*Cyprinodon varigatus*) early life stage study (MRID 432505-01) that resulted in a NOAEC of 2.6 μg/L and LOAEC of 6.0 μg/L based on reduced embryo hatching. Specific extent of reduced hatching was not reported. No sublethal effects, such as the scoliosis in the rainbow trout, were mentioned in this study or two other studies on sheepshead minnow (MRIDs 408184-01 and 426974-01) that were submitted but did not produce definitive NOAECs or LOAECs.

The Science Chapter for methomyl (EPA 1998) reports a NOAEC (57 μg/L) and LOAEC (117 μg/L) for fathead minnow (*Pimephales promelus*) based on larval survival. Specific reductions in larval survival were not reported. In the Science Chapter for methomyl (EPA 1998), the study is referred to as MRID 00131255 and Driscoll 1982 and appears to reference the same study as Acc. 251424. The same data are reported in the BE (EPA 2003), but without any citation. We located the data report and confirmed that embryo hatchability was not affected by the dose regime tested: 0, 57, 113, 254, 395, and 972 μg/L (measured concentrations).

Assessment endpoint: Fish growth

Assessment Measure: Growth rate, weight, length, or biomass of second generation as measured in chronic toxicity tests

The carbaryl BE (EPA 2003) did not report any effects on growth for either freshwater or estuarine/marine species, although reduced growth is listed as the affected endpoint in a fathead minnow (*Pimephales promelas*) study referenced in the Science Chapter (EPA 2003). For this study, the NOAEC is given as 210 μg/L and the LOAEC as 680 μg/L However, the fathead minnow study discussed within the reproductive endpoints summary measured growth of fathead minnow larvae (length) at 30 and 60 days (Carlson 1971). Although statistical tests were not used, no differences in growth were apparent between exposed and control fish at the concentrations tested: 8, 17, 62, 210, and 680 μg/L (Carlson 1971).

The carbofuran BE (EPA 2004) reports larval rainbow trout (*O. mykiss*) exposed to 56.7 and 88.7 μg/L carbofuran had significantly reduced lengths and reduced survival at 60, 75, and 90 d compared to unexposed trout in a partial life cycle test. Additionally, trout weight was significantly reduced after 75 d of exposure to 56.7 and 88.7 μg/L. Although, no definitive reference was provided for these data, NMFS obtained the study report to verify study results. Upon review, several additional adverse effects to fry were noted throughout the experiment. At day 20, trout exposed to 57.8 and 88.7 μg/L respired rapidly compared to control fish and were hyperactive at 88.7 μg/L. These effects continued for the remainder of the experiment (~ 70 days). Following 75 d of exposure to carbofuran at 56.7 and 88.7 μg/L, 23 and 35% of the fish, respectively, had curved spines. The Science Chapter (EPA 2005) reports growth was the most sensitive endpoint in two sheepshed minnow studies (*Cyprinodon varigatus*, MRIDs 40818401 and 42697401), but notes that neither of these tests produced a definitive NOAEC or LOAEC.

The methomyl BE (EPA 2003) lists a NOAEC of 76 μg/L and a LOAEC of 142 μg/L based on growth endpoints for fathead minnow (*Pimephales promelas*), a freshwater species. It also lists a NOAEC of 260 μg/L and a LOAEC of 490 μg/L for sheepshead minnow (*Cyprinodon* *varigatus*), an estuarine species, but the endpoint affected is described as reproduction and/or growth, so whether or not growth was affected is uncertain. The Science Chapter (EPA 1998) does not describe any data for estuarine/marine species.

Assessment endpoint: Habitat- salmonid prey

Assessment measure: Aquatic invertebrate survival, growth, reproduction from acute and chronic laboratory toxicity tests

The carbaryl BE (EPA 2003) lists a range of acute EC50 values for freshwater aquatic invertebrate survival. Six EC50 values are given for *Daphnia magna*, ranging from 4.3 -13.0 μg/L (mean 7.3 μg/L). Although not specified as such, given the test organism, the fact that they are noted as 48 h tests, and the range of percent a.i., we assume that these are guideline tests conducted by the registrant or an acceptable government lab. EC50s were also given for three species of stoneflies, ranging from 1.7-5.6 μg/L (mean 3.6 μg/L). An EC50 for an amphipod (*Gammarus fasciatus*) of 26 μg/L was also given. There were also EC50 values for a number of estuarine species. There were five EC50 values given for mysid shrimp (*Mysidopsis bahia*), a common guideline test organism, ranging from 5.7-20.2 μg/L (mean 10.3 μg/L. EC50s for various other shrimp species were given, ranging from 1.5-170 μg/L. All of these tests appear to have been conducted on the a.i. An LC50 for blue crab of 320 μg/L was given. The BE also reported a number of EC50/LC50 values for aquatic insects and other invertebrates derived from the USEPA ECOTOX database.

The carbaryl Science Chapter (EPA 2003) used an acute value of 5.1 μg/L (stonefly, *Chlorperla grammatica*) as the survival assessment endpoint for freshwater aquatic invertebrates, and 5.7 μg/L (mysid shrimp) as the survival assessment endpoint for estuarine/marine aquatic invertebrates. The chapter also notes:

> "Studies have indicated that acute exposure to carbaryl impacts predator avoidance mechanisms in invertebrates, reduces overall zooplankton abundance (Hanazato and Yasuno 1989; Havens 1995), and may actually promote phytoplankton growth through reduced predation by zooplankton."

Acute toxicity data for 1-napthol presented in the BE (EPA 2003) listed 48 h EC50s of 700-730 μg/L for *D. magna*, and EC50s of 200-210 μg/L for *M. bahia*. No additional information regarding acute toxicity of 1-napthol to aquatic invertebrates was provided in the Science Chapter (EPA 2003).

Overall, results presented show that carbaryl and formulations of carbaryl are acutely toxic to a wide array of aquatic invertebrates in the low μg/L range, frequently with EC50s/LC50s of less than 10 μg/L. The degradate, 1-napthol, appears less toxic with respect to comparable invertebrates; acute survival EC50s ranged from 200-730 μg/L. However, no data for the genera more sensitive to the parent carbaryl, such as the stoneflies, caddisflies, or mayflies, are available. Thus the lower end of the toxicity range is not well established.

The carbaryl BE (EPA 2003) lists a NOAEC of 1.5 μg/L and a LOAEC of 3.3 μg/L for *D. magna* based on reproduction and a NOAEC of 500 μg/L and a LOAEC of 1,000 μg/L for the midge fly (*Chironomous riparius*) based on emergence/developmental rate. Specific percentages of inhibition are not noted. The Science Chapter (EPA 2003) does not add any additional detail, but does note:

> "midge larvae are benthic macroinvertebrates and exposure may have been better characterized had it been based on sediment pore water concentrations as opposed to carbaryl concentrations in overlying water"

as a potential explanation for the difference in sensitivity between *D. magna* and *C. riparius*. No data regarding reproductive endpoints were presented for 1-napthol.

The carbofuran BE (EPA 2004) included acute EC50s for the *D. magna* (29-38.6 μg/L) and pink shrimp (*Penaeus duorarum*, 4.6-7.3 μg/L). All tests appear to have been conducted with the a.i. EC50s of 2.2-2.6 μg/L were reported in the Science Chapter (EPA 2005) for the freshwater water flea *Ceriodaphnia dubia*. The Science Chapter also provided toxicity values for the freshwater red crayfish (*Procambarus clarkii*, LC50 2,700 μg/L) and the eastern oyster (*Crassostrea virginica*, EC50s of >1,000->5,000 μg/L). Probit slopes were included in Appendix H of Appendix 1 (EPA 2005) for studies where they were reported or data were available to calculate them. Probit slopes were not available for *D. magna* or *C. dubia*, but were available for the red crayfish (slope 2.91) and the pink shrimp (slope 2.25). Data provided from chronic toxicity tests included a NOAEC of 9.8 μg/L and a LOAEC of 27 μg/L for the freshwater aquatic invertebrate *D. magna*, and NOAEC of 0.4 μg/L and a LOAEC of 0.98 μg/L for the estuarine invertebrate *M. bahia*. In both cases, the most sensitive endpoints appeared to be survival of the adults and/or growth rather than decreased production of offspring.

The BE (EPA 2004) and Science Chapter (EPA 2005) reference several field studies from the open literature that examined effects on aquatic invertebrates following application of carbofuran either to fields or directly to water. One study (Matthiessen, Shearan et al. 1995) noted complete mortality of caged amphipods (*Gammarus pulex)* in a stream draining a field treated with granular carbofuran at 2.7 lbs a.i./acre. EPA documents reported a pond enclosure study (Wayland 1991) noting decreases in amphipod *(Hyallela azteca)* abundance and biomass and Chironominae larvae biomass at concentrations of 25 μg/L in a pond enclosure study. EPA also reported several studies in which carbofuran applied directly to water caused mortality in aquatic invertebrates at concentrations ranging from 5-25 μg/L (Flickinger et al. 1986, Mullie et al. 1991, Wayland and Boag 1990).

The methomyl BE (EPA 2003) reported acute survival EC50s for the freshwater invertebrates water flea (*D. magna*), amphipod (*Gammarus pseudolimnaeus*), midge (*Chironomous plumosus*) and three genera of stoneflies (*Skwala* sp., *Pteronarcella badia*, and *Isogenus* sp.). Most values appeared to be derived from 48-or 96 h standard laboratory tests. Many of the tests were with a.i. (95-99% a.i.), but others were conducted with a 24% formulation (EPA 2003). The product name was not specified. Information in the BE is noted as having come from the EFED Pesticide Ecotoxicity Database, not the RED Science Chapter. There are some inconsistencies between these two documents regarding which EC50 is associated with the formulation versus the technical a.i. Based on information in the BE, which lists data for a formulation test on all species, the 24% formulation appears to be more toxic than the technical. EC50s for the technical range from 8.8-31.8 μg/L for *D. magna*, and are reported as 920 μg/L for *G. psuedolimnaeus*, 88 μg/L for *C .plumosus*, 34 μg/L for *Skwala* sp, 69 μg/L for *P. badia*, and 343 for *Isogenus* sp. EC50s for the 24% formulation are reported as 7.6 μg/L for *D. magna*, 720 μg/L for *G. psuedolimnaeus*, 32 μg/L for *C. plumosus* 29 μg/L for *Skwala* sp, 60 μg/L for *P. badia*, and 29 for *Isogenus* sp.

Acute EC50s are also reported for several species of estuarine/marine shrimp in both the BE (EPA 2003) and the RED Science Chapter (EPA 1998). Again, there are inconsistencies between the documents, with the same EC50s from formulation tests reported in the BE as 30% a.i. and in the Science Chapter as 24% a.i. Some EC50s were also reported for the a.i. Only one species, grass shrimp (*Palmonetes vulgaris*) appears to have been tested with both the a.i. and the formulation (EPA 2003). In this case, the formulation (EC50 130 μg/L, reported as a 30% a.i.), appears less toxic than the technical (EC50 49 μg/L). Other EC50s listed in the BE were for pink shrimp (*P. duorarum*, 19 μg/L) and mysid shrimp (*M. bahia*, 230 μg/L), both of which appear to be technical a.i. (90-98.4%).

The methomyl BE (EPA 2003) reports NOAECs and LOAECs for reproductive endpoints for two studies conducted on *D. magna*, both with the technical a.i. In one study, with a NOAEC of 0.4 μg/L and a LOAEC of 0.8 μg/L, the number of young per female was reduced. A second resulted in a NOAEC of 1.6 μg/L and a LOAEC of 3.1 μg/L, based on an unspecified reproductive endpoint. The BE also reports a NOAEC of 29 μg/L and a LOAEC of 59 μg/L for the estuarine mysid shrimp, based on "reproduction and/or growth."

The BE and Science Chapter both report on an outdoor microcosom study conducted with methomyl (MRID 437444-02). The Science Chapter, Appendix C (EPA 1998), describes the study design as: "Methomyl was applied to seven treatment groups, at two application rates, at three different application intervals, over a period of 22 days (pg 51)." Specific application rates and intervals were not provided, nor was use of a control specifically mentioned. Zooplankton (Cladocera, Copepodia and Rotifera) abundance and community composition were altered in at least some treatments, and Ephemeroptera abundance decreased in the two highest treatments.

Toxicity of Carbaryl, Carbofuran, and Methomyl Degradates

The BEs briefly addressed the issues of degradates. The carbaryl BE (EPA 2003c) provides acute fish and invertebrate toxicity data the degradate, 1-napthol. The LC50s for three species of fish tested (freshwater (*O. mykiss*, *Lepomis macrochirus*)) and marine/estuarine (*Cyprinodon varigatus*) range from 750 – 1,800 μg/L. LC50s presented for aquatic invertebrates include 700-

730 μg/L for *D. magna* (freshwater organism), 200-210 μg/L for *M. bahia* (estuarine organism) and 2,100 μg/L for *C. virginica* (estuarine organism).

During our open literature review, we located a study that compared acute lethalities between carbaryl and 1-naphthol in two species of fish (Shea and Berry 1983). In goldfish (*Carassius auratus*) and killifish (*Fundulus heteroclitus*), 1-naphtol was significantly more toxic than carbaryl based on 10 d acute lethality tests (Shea and Berry 1983). The degradate 1-naphthol was approximately five times more toxic than carbaryl in goldfish, and in killifish twice as toxic as carbaryl (Shea and Berry 1983). Additionally, fish exposed to 1-naphthol showed neurological trauma including pronounced erratic swimming behaviors and increased opercula beats following exposure to 5 and 10 mg/L after 4 and 24 h exposures, respectively. None of these symptoms were observed in the carbaryl treatments (Shea and Berry 1983).

Other degradates identified in fate studies submitted to EPA included 5-hydroxy-1-napthylmethylcarbamate (aerobic soil metabolism and anaerobic aquatic); 1-naphthyl (hydroxymethyl) carbamate (aerobic soil metabolism and anaerobic aquatic); 1,4-napththoquinones (degradates of 1-napthol); 4-hydroxy-1-napthyl methylcarbamate (anaerobic aquatic); 1,5-naphthalenediol (anaerobic aquatic); and 1,4-naphthalenediol (anaerobic aquatic). No toxicity information was presented for these degradates.

The carbofuran BE (EPA 2005) describes three degradates: 3-hydroxycarbofuran, 3-ketocarbofuran, and carbofuran 7-phenol in the toxicity section, and refers to two additional degradates: 3-hydroxy-7-phenol, and 3-keto-7-phenol in the section on environmental fate and transport. No toxicity data are included, and the document noted that “inclusion of environmental transformation products in the risk analysis of carbofuran would not be expected to result in substantive changes to conclusions drawn using the parent alone (pg 30)” (EPA 2005).

Based on the more detailed fate information in the Science Chapter for the Carbofuran RED (EPA 2005), carbofuran 7-phenol is a hydrolysis product (up to 75% of applied). Carbofuran 7-phenol is expected to be less toxic than parent carbofuran, as the toxic moiety (the carbamylating radical) has already dissociated (EPA 2005). No toxicity data for this compound are provided in either the BE or the Science Chapter. The degradates 3-hydroxycarbofuran and 3-ketocarbofuran are detected in soil photolysis and aerobic soil metabolism studies. These compounds were each approximately 3-5% of applied (EPA 2004a), and are structurally more similar to the parent carbofuran. The Science Chapter cites an open literature study using Microtox (Kross, et. al., as cited in EPA 2005) noting “3-ketocarbofuran appears as toxic or slightly more toxic than the parent, but 3-hydroxycarbofuran is much less toxic.” A second study evaluated (Gupta 1994, as cited in EPA 2005) indicates 3-hydroxycarbofuran is “equally as toxic as the parent.” No specific toxicity values were presented. Other degradates mentioned in the fate summary are 3-hydroxycarbofuran phenol (3-hydrocy-7-pheonl) and 3-ketocarbofuran phenol (3-keto-7-phenol), which are less than 5% of applied (EPA 2005).

The methomyl BE notes “a degradate (thiolacetohyroxamic acid 5-methyl ester)… was tested and found to be practically nontoxic to bluegill” (EPA 2003a). This study is also mentioned in the Environmental Risk Assessment for the RED (EPA 1998c). The bluegill LC50 for this degradate is 462,000 μg/L. No other degradate toxicity data were presented. The fate portion of the BE (EPA 2003a) and the Environmental Risk Assessment (EPA 1998c) both reference S-methyl-N-hydroxythioacetamidate as a product of hydrolysis in both water and soil. This may be the same compound expressed under different naming conventions, but no structures were provided to confirm this identification. No toxicity data were presented for acetonitrile or acetamide.

Formulations and other (inert) ingredients found in carbaryl’s, carbofuran’s, and methomyl’s formulations

Assessment endpoint: Fish survival, aquatic invertebrate survival, and primary production

Assessment measure: Aquatic invertebrate survival, growth, and reproduction from acute and chronic laboratory toxicity tests

The carabaryl BE (EPA 2003c) and the carbaryl Science Chapter (EPA 2003b) provide some toxicity data on formulations containing 5-81.5% carbaryl for both aquatic invertebrates and fish. We did not receive information on whether the formulations tested are currently registered. Data for formulations (referred to in the Science Chapter (EPA 2003b) as technical end-product or

TEP) are contained in Appendix D-1. The technical a.i. EC50 for water flea (*D. magna*), based on a single test, is 5.6 μg/L. No 95% confidence interval is given for the test. Data for several formulations with 43.7-81.5% carbaryl are presented. EC50s for the formulations range from 4.3-13.0 μg/L. Data are also available to compare toxicity of the technical grade to formulations (43.7-81.5%) for the estuarine mysid shrimp, *M. bahia*. No 95% confidence intervals are given. Two technical a.i. LC50s were provided in the data, 5.7 μg/L and 6.7 μg/L. Tests with formulations (n=3) resulted in LC50s of 9.3-20.2 μg/L. No chronic tests were described for any formulations.

For carbaryl, acute LC50 formulation data are also given for two species of fish, rainbow trout (*O. mykiss*, 44-81.5 % carbaryl) and bluegill sunfish (*Lepomis macrochirus*, 5-50% carbaryl). No 95% confidence intervals are given. Although other fish data for the technical carbofuran are provided, a same-species comparison of toxicity data is better than evaluating against a range of species, as the various species will exhibit a range of sensitivity. A single a.i. LC50 (1,200 μg/L) is given for rainbow trout. A total of four LC50s are given for formulations, ranging from 1,400-4,500μg/L. Three a.i. LC50s for bluegill are given, ranging from 5,000-14,000 μg/L. The bluegill LC50s (n=4) for formulations range from 9,800-290,000 μg/L. Based on these data, it appears the formulations tested are relatively similar in toxicity to the a.i. on an acute lethality basis (survival endpoint). No chronic tests, which evaluate reproduction and growth, were described for any formulations.

The carbofuran BE (EPA 2004a) lists some formulation toxicity data for aquatic invertebrates, referenced to the EFED Science Chapter and other data referenced to the EPA Acquire Database. In some cases, the formulation data referenced to the Science Chapter provides percent a.i. and in others it contains designations such as “5G” or “50 WP”. In cases where the percent a.i. is reported as a number, we presume the toxicity value has been corrected to percent a.i. It is unclear whether the others have been corrected, whether the data from the Acquire database has been corrected, or the specific products with which the toxicity data are associated.

Toxicity data presented in the carbofuran Science Chapter (EPA 2005) were found in Appendix H of Appendix 1 to the main document, titled Animal Toxicity Tests with DERs. Study data on formulations were presented for bluegill sunfish (*L. macrochirus*) and rainbow trout (*O. mykiss*). A single test on rainbow trout was conducted using the formulation Furadan 75WP with a reported LC50 of 458 μg/L. LC50s for rainbow trout tested with technical carbofuran ranged from 362-600 μg /L. A number of different formulations (Furadan 4F, Furadan 10G, Furadan 50WP and Furadan 75WP) were tested with bluegill sunfish. LC50s ranged from 240-488 μg carbofuran/L. LC50s for bluegill sunfish tested with technical carbofuran ranged from 88-126 μg a.i/L. Only one formulation test for freshwater invertebrates was reported. *D. magna* exposed to "5G" had a survival EC50 of 41 μg /L. Tests with *D. magna* on technical carbofuran produced survival EC50s of 29-38.6 μg /L. The estuarine fish Atlantic silverside (*Mendia menidia*) was tested in two formulations, Furadan 4F and Furadan 15G. LC50s from these tests were 36 μg a.i./L and 64 μg a.i./L, respectively. A test with technical grade on the same species produced an LC50 of 33 μg a.i./L. (95% CI 27-41 μg a.i./L). The estuarine/marine invertebrate pink shrimp was tested both in a formulation (Furadan 15G) EC50 13.3 μg a.i./L and with technical (EC50s 4.6-7.3 μg a.i./L, n=2). Based on these data, there is no indication that the carbofuran formulations tested are more toxic than the a.i. on survival as an endpoint. No long term studies, such as life cycle studies, with formulations were reported for either fish or invertebrates.

The methomyl BE (EPA 2003a) and Science Chapter for the RED (EPA 1998c) both reported toxicity data for a 24% formulation and a 29% formulation for both fish and aquatic invertebrates. In addition, the BE reported some toxicity data for a 30% formulation. While it is difficult to make a definitive comparison given the structure of the data, the 24% formulation appears more toxic to both fish and invertebrates. Neither the toxicity section of the BE nor the use characterization in the Science Chapter list specific products. Thus, we cannot cross-reference the data to a product or determine if it is a currently registered product.

Identified data gaps and uncertainties of carbofuran's, carbaryl's and methomyl's toxicity information presented in BEs and REDs:

Overall, data provided in the BEs and their Science Chapters were insufficient to allow a thorough evaluation of all identified assessment endpoints and measures considered by NMFS. The Data Evaluation Reviews (DERs) or the original studies may have helped reduce some of the uncertainties related to experimental design, dose-response slopes, and confidence estimates of the data, but we did not receive summaries of this information from EPA when preparing this Opinion. However, even if the DER data were submitted, other aspects of EPA's assessment endpoints (survival, reproduction, and growth) were not presented. When this missing information is combined with the absence of information presented on other non-assessed endpoints such as AChE inhibition in fish and invertebrates, swimming behaviors, and olfactory-mediated impairments, an incomplete picture emerges on the potential effects of EPA's action to listed salmonid and habitat endpoints.

Toxicity information presented in the BEs (EPA 2003c, EPA 2004a, and EPA 2003a) lacks details NMFS requires for analyses in support of this consultation. More complete information is presented in the Science Chapters for carbaryl (EPA 2003c) and carbofuran (EPA 2005), yet the information was not applied by EPA in its characterization of risk to listed salmonids or salmonid habitats. Specific data gaps identified include the following, although not all gaps apply to all three carbamates.

- Reported LC50s not accompanied by slopes, experimental design (number of treatments and replicates, life stage of organism, concentrations tested), measures of variability such as confidence intervals or standard deviations/errors;
- No analysis of the degree or magnitude of inhibition of acetylcholinesterase by the three carbamates and expected response by listed salmonids or aquatic, salmonid prey communities;
- Summary and discussion of fish sublethal data absent from BEs including effects to swimming and chemoreception;
- Limited or no toxicity data on current formulations,
- Limited or no toxicity data presented for identified surface water degradates of the three carbamates;
- Sensitivity of surrogate lab strains compared to wild, listed fish, particularly comparisons between warm and cold water fish species used in chronic guideline tests;
- No data summarized for mixture toxicity including tank mixtures and environmental mixtures to assessment endpoints;
- No toxicity data presented on "inert" or other ingredients present in formulations containing each of the a.i.s;
- No analysis of influence of environmental factors (pH, temperature) on exposure and toxicity.

Summary of Toxicity Information from Open Literature

To organize the available toxicity information on listed salmonids and habitat, we developed risk hypotheses with associated assessment endpoints as described in the *Approach to the Assessment* section. Recall that assessment endpoints are biological attributes of salmonids and their habitat potentially susceptible to the stressors of the action. In addition to toxicity data presented in the BEs, we also considered information from other sources to evaluate both individual and population-level endpoints. The results of those studies are summarized below under corresponding assessment endpoints. We qualitatively assigned the most significance to study results that were: 1) derived from experiments using salmonids (preferably listed Pacific salmonids or hatchery surrogates); 2) measured an assessment endpoint of concern *e.g.*, survival, growth, behavior, reproduction, abundance etc., identified in a risk hypothesis; 3) resulted from exposure to stressors of the action or relevant chemical surrogates (*i.e.*, other AChE inhibitors); and 4) had no substantial flaws in the experimental design. When a study did not meet these criteria, we highlighted the issue(s) and discussed how the information was used or why the information could not be used.

Assessment endpoint: Swimming

Assessment measures: Burst swimming speed, distance swam, rate of turning, baseline speed, tortuosity of path, acceleration, swimming stamina, and spontaneous swimming activity

Swimming is a critical function for anadromous salmonids that is necessary to complete their life cycle. Impairment of swimming may affect feeding, migrating, predator avoidance, and spawning (Little and Finger 1990). It is the most frequently assessed behavioral response of toxicity investigations with fish (Little and Finger 1990). Swimming activity and swimming capacity of salmonids have been measured following exposures to a variety of AChE-inhibiting insecticides including the carbamates carbaryl and carbofuran and a variety of OP insecticides. Swimming capacity is a measure of orientation to flow as well as the physical capacity to swim against it (Howard 1975; Dodson and Mayfield 1979). Swimming activity includes measurements of frequency and duration of movements, speed and distance traveled, frequency and angle of turns, position in the water column, and form and pattern of swimming. A review paper summarized many of the experimental swimming behavioral studies and concluded that effects to swimming activity generally occur at lower concentrations than effects to swimming capacity (Little and Finger 1990). Therefore, measurements of swimming activity are usually more sensitive than measurements of swimming capacity. A likely reason is that fishes having impaired swimming to the degree that they cannot orient to flow or maintain position in the water column are moribund (*i.e.*, death is imminent). The authors of the review also concluded that swimming-mediated behaviors are frequently adversely affected at 0.3 – 5.0% of reported fish LC50s[3], and that 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC50s (Little and Finger 1990). Both swimming activity and swimming capacity are adversely affected by AChE-inhibiting insecticides.

We located studies that measured impacts to salmonid swimming behaviors from exposure to carbaryl. Several studies also measured AChE inhibition from OPs and provided correlations between AChE activity and swimming behaviors. We did not locate any studies that tested mixtures of AChE-inhibiting insecticides on swimming behaviors of any aquatic species.

Carbaryl

Experiments with carbaryl have shown that cutthroat trout's (*Oncorhynchus clarki clarki*) swimming abilities are compromised by 6 h exposures to 750 and 1,000 μg/L, resulting in increased susceptibility to predation (Labenia, Baldwin et al. 2007). Cutthroat trout swimming capacity was not significantly affected by carbaryl at 500 μg/L, although muscle AChE activity was 29% relative to unexposed cutthroat. More sensitive swimming activity measurements were not evaluated. At 750 and 1,000 μg/L, AChE activity was 24 and 23% relative to unexposed fish, respectively. A known predator of juvenile cutthroat, lingcod (*Ophiodon elongates*), consumed on average more cutthroat that were exposed to carbaryl compared to those that were not exposed to carbaryl. The experimental design ensured that the predator was exposed to minimal concentrations when carbaryl-exposed fish were transferred in a large experimental chamber along with unexposed cuthroat. In the predation experiment with lingcod, cutthroat were exposed for 2 h to 200, 500, and 1,000 μg/L carbaryl. Results indicated a dose-dependent decrease in the ability of carbaryl-exposed trout to avoid being eaten by the lingcod predator. At

[3] The current hazard quotient-derived threshold for effects to threatened and endangered species used by EPA is 5 % (1/20th) of the lowest fish LC50 reported. If the exposure concentration is less than 5 % of the LC50 a no effect determination is made which likely underestimates risk to listed salmonids based on swimming behaviors.

200 μg/L, an increase in predation was evident, although not statistically significant. At 500 and 1,000 μg/L carbaryl, cutthroat trout were consumed at significantly higher rates than unexposed fish. Cutthroat trout's AChE activity was also reduced in a dose-dependent fashion showing greater reductions with increasing carbaryl concentrations. Six hour exposures significantly reduced AChE activity in brain and muscle, where 50% reductions (IC50) in brain AChE occurred at 213 μg/L carbaryl. The onset of inhibition occurred within 2 hours, at which point AChE activity was near its lowest value. Benchmark concentrations corresponding to 20% inhibition were 32 μg/L in brain and 23 μg/L in muscle tissues. Recovery of AChE activity (to pre-exposure levels) took 42 hours at 500 μg/L carbaryl. We ranked these sets of experiments as highly relevant to understanding the effects of carbaryl on inhibition of AChE and subsequent salmonid swimming behaviors. The test also provided information on lack of predator avoidance behaviors by salmonids.

Catfish (*Mystus vittatus*) showed increased swimming activity following 72 h of exposure to 12,500 μg/L and no mortalities were noted (Arunachalam, Jeyalakshmi et al. 1980). The 72 h LC50 for catfish was 17,000 μg/L, indicating this species of catfish is much less sensitive to acute concentrations of carbaryl than salmonids. Other sublethal endpoints were also affected by carbaryl, including food intake, growth, metabolism, and rate of opercular beats. Catfish increased their rates of opercular beats from 74 per minute exposed to freshwater alone to 124 per minute when exposed to 12,500 μg/L carbaryl, which the authors attributed to acute stress from carbaryl. We ranked this study as relevant because swimming was measured, but the high concentrations used, lack of chemical verification, and lack of compatibility to listed salmonids introduces uncertainty.

Multiple behavioral responses related to swimming were assessed in rainbow trout fry (0.5-1 g) following 96 h exposures to carbaryl at 10, 100, and 1,000 μg/L; nominal concentrations (Little, Archeski et al. 1990). These included swimming capacity (cm/sec), swimming activity (sec), prey strike frequency, daphnids consumed, percent consuming daphnids, and percent survival from predation. All endpoints were significantly affected at 1,000 μg/L carbaryl relative to unexposed fry. At 10, 100, and 1,000 μg/L carbaryl, significantly more rainbow trout were consumed relative to unexposed fish. By using very young fry, the studies also provide information on a sensitive early lifestage where swimming behaviors are critical to survival (*i.e.*, feeding and predator avoidance). We ranked these studies as highly relevant to a variety of essential swimming-associated behaviors.

One study tested whether schooling of fish in estuarine waters was affected by a single short-term exposure of carbaryl at 100 μg/L (Weis and Weis 1974). Carbaryl impaired schooling of Atlantic silversides (*M. menidia*) by increasing the school area by up to twice that of control groups. Recovery of schooling behavior took three days (Weis and Weis 1974). The authors suggested that increased schooling areas would increase energy expenditures of individual fish and also increase rates of predation. The study lacked analytical verification of exposure concentrations; only one concentration was tested; and the study was conducted with a non-salmonid fish. The study is relevant because it addressed an important swimming behavior (*i.e.*, schooling), which juvenile salmonids use at times, and presented data on time to recovery following an adverse behavioral affect. However, it remains uncertain at which concentrations juvenile salmonids that school would be affected.

Neurological effects on the startle response and ability to avoid predation by juvenile medaka (*Oryzias latipes*) were evaluated following exposures of 2.5, 5.1, 7.0, and 9.4 mg/L carbaryl (Carlson, Bradbury et al. 1998). At 5.1 mg/L and higher, carbaryl increased the time between motor neuron peak and initiation of muscle activity (*i.e.*, swimming response) and at 7.0 mg/L and higher response to stimuli ratios were also higher. Predation trials showed no differences between medaka exposed to carbaryl and unexposed fish. While these concentrations are extremely high compared to concentrations affecting salmonids, the results indicate that neurological-associated swimming behaviors are affected by carbaryl. Given that the 48 h LC50 of juvenile medaka is estimated at 9.4 mg/L and that detectable sublethal neurological effects occurred at 5.1 mg/L, roughly half the LC50, it is uncertain at what concentrations juvenile salmonids neurological endpoints would be affected. We ranked this experiment as relevant as it provided evidence that neurological effects manifest at lower concentrations than 48 h LC50s.

Carbofuran

Carbofuran adversely affected swimming behaviors in goldfish (*C. auratus*) following 24 and 48 h exposures to the lowest concentration tested, 5 μg/L (Bretaud, Saglio et al. 2002). Swimming activity (fish swimming from one zone to another), the least sensitive endpoint, was significantly affected at 500 μg/L carbofuran, while burst swimming, the most sensitive endpoint, was significantly affected at 5 μg/L following 24 h exposure (Bretaud, Saglio et al. 2002). Burst swimming behavior, sheltering, and nipping in goldfish were significantly increased by a 4 h exposure to 1 μg/L carbofuran (Saglio, Trijasse et al. 1996). At 12 h exposures, significant effects were observed at 100 μg/L for sheltering, nipping, and burst swimming. Grouping of goldfish showed non dose-dependent responses as significant effects ($p<0.05$) appeared at 10 μg/L, but were absent at 1 and 100 μg/L (Saglio, Trijasse et al. 1996). We ranked both of these studies as relevant, but due to difficulties inherent in translating the observed behavioral effects in goldfish to salmonids, combined with the lack of analytical verification of concentrations, the study did not receive a highly relevant ranking. The results support that ecologically relevant swimming-related behaviors are impacted at lower concentrations than more coarse measures of swimming such as swimming stamina or swimming capacity.

Methomyl

We were unable to locate any studies that evaluated effects to swimming related behaviors in fish. Although this is a data gap, we assume methomyl does inhibit swimming in the same fashion as the other carbamates because swimming behaviors are typically affected when AChE is sufficiently inhibited (*e.g.* by 30% or more). A review also concluded that swimming-mediated behaviors are frequently adversely affected at 0.3 – 5.0% of reported fish LC50s, and that 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC50s (Little and Finger 1990).

Other AChE-inhibiting insecticides effects on swimming and related behaviors

We also reviewed study results conducted with other carbamate and OP insecticides because both classes of compounds share the same mode of action. Recovery of swimming to pre-exposure levels is much more rapid in carbamate-affected fish compared to OP-affected fish due to the reversible binding of carbamates with AChE (Mineau 1991). In one study with carbaryl and rainbow trout, AChE activity returned to control levels after 24 h in clean water following

24 hr exposure (Zinkl, Shea et al. 1987). In another study with cutthroat trout within 42 h recovery occurred following 6 h exposure(Labenia, Baldwin et al. 2007). In the latter study, cutthroat trout recovered approximately half of their pre-exposure AChE activity within 6 h, following peak AChE inhibition within 2 h. Both of these studies indicate that recovery of AChE activity following exposure to carbamates occurs quickly and is highly dependent on environmental exposure conditions found in aquatic habitats.

We did not locate additional studies with other carbamates, but located multiple studies with OP insecticides. The OPs chlorpyrifos, diazinon, and malathion showed significant and persistent effects to a suite of swimming related behaviors in salmonids at concentrations expected in salmonid habitats; as reviewed in NMFS' November 18, 2008 Opinion on these three OP insecticides (NMFS 2008). Robust evidence of the three OPs showed reductions in swimming speed (Brewer, Little et al. 2001), distance swam (Brewer, Little et al. 2001), acceleration (Tierney, Casselman et al. 2007), food strikes (Sandahl, Baldwin et al. 2005) and significant correlations with AChE activity (Brewer, Little et al. 2001; Sandahl, Baldwin et al. 2005). Additionally, other OPs including fenitrothion, parathion, and methyl parathion, adversely affected a suite of swimming behaviors reviewed by (Little and Finger 1990). One noteworthy study investigated the effects of six pesticides including methyl parathion (OP) and tribufos (OP) on rainbow trout swimming behavior (Little, Archeski et al. 1990). All insecticides adversely affected spontaneous swimming activity, while DEF also reduced swimming capacity in juvenile rainbow trout (Little, Archeski et al. 1990). In bluegill, methyl parathion adversely affected burst swimming behavior at 300 μg/L (Henry and Atchison 1984). Respiratory disruptions, comfort movements, and aggression behaviors in bluegill were all adversely affected by 24 h exposures to methyl parathion at 3.5 μg/L. This suggests that these social behaviors are very sensitive to AChE inhibition (Henry and Atchison 1984).

Two month old juvenile rainbow trout, brook trout, and coho were exposed to malathion (Phillaps Malathion 55%) for 7-10 days depending on species (Post and Leasure 1974). Swimming performance, brain AChE activity, and recovery time were measured following exposure to malathion concentrations of 0, 40, 90, 120 μg/L in brook trout; 0, 55, 112, 175 μg/L in rainbow trout; and 0, 100, 200, 300 μg/L in coho. Additionally, once fish recovered AChE activity, they were subjected to a second exposure to determine if prior exposure altered susceptibility to malathion. Swimming performance and AChE activity did not differ from values of the initial exposure *i.e.*, a second exposure resulted in no evidence of increased susceptibility. Brook trout were the most sensitive based on AChE inhibition followed by rainbow trout and coho salmon, respectively. AChE inhibition of 25% relative to control fish occurred at 40 µg/L (brook trout), 55 µg/L (rainbow trout), and 100 µg/L (coho). Swimming performance was affected at the lowest concentrations tested in each salmonid species and showed a dose-dependent decrease in swimming performance as malathion concentration increased. The data indicated that AChE inhibition of approximately 20-30% resulted in a 5% or less reduction in swimming performance and as inhibition increased, swimming performance decreased. Note, however, that the swimming test conducted in the study is a coarse measure of swimming capacity. Thus, other non-measured swimming activity endpoints would likely be affected at lower concentrations (Little and Finger 1990; Little, Archeski et al. 1990).

In summary, the information presented on swimming behaviors from AChE-inhibiting insecticides provide a weight of evidence that carbamates (and OPs) adversely affect swimming behaviors which can reduce the fitness of affected salmonids.

Assessment endpoints: Olfaction and olfactory-mediated behaviors: Predator avoidance, prey detection and subsequent growth, imprinting of juvenile fish to natal waters, homing of adults returning from the ocean, and spawning/reproduction

Assessment measures: Olfactory recordings (electro-olfactogram), behavioral measurements such as detection of predator cues and alarm response, adult homing success, AChE activity in olfactory rosettes and bulbs, and avoidance/preference

The olfactory sensory system in salmonids is particularly sensitive to toxic effects of metals and organic contaminants. This is likely a result of the direct contact of olfactory neurons and dissolved contaminants in surface waters. Olfactory-mediated behaviors play an essential role in the successful completion of anadromous salmonid life cycles, and include detecting and avoiding predators, recognizing kin, imprinting and homing in natal waters, and reproducing. It is well established that Pacific salmon lose navigation skills when olfactory function is lost and consequently are unable to return to natal streams (Wisby and Hasler 1954).

We located studies that measured olfactory responses of fish to carbofuran and carbaryl. We found no studies with methomyl. Several studies with other carbamates were found, but most were with thiocarbamates which have a different mode of action than *N*-methyl carbamates, *i.e.*, they do not inhibit acetylcholineterase. We do not discuss or use studies with OPs as surrogates for effect to olfaction in salmonids because AChE inhibition does not appear to be the putative mode of action affecting olfaction, although more empirical data are needed to confirm this. Below we discuss the available literature of carbamate effects on fish olfaction.

The olfactory activity of juvenile cutthroat trout appeared unresponsive to carbaryl following 10 second pulses across the olfactory epithelia, and juveniles showed no preference/avoidance to carbaryl (Labenia, Baldwin et al. 2007). No departures relative to unexposed fish were observed at 5, 50, or 500 μg/L carbaryl from neurophysiological recordings *i.e.*, electro-olfactograms. Additionally, in a behavioral avoidance assay, cutthroat trout did not avoid seawater containing carbaryl at 500 μg/L. These results suggest that cutthroat trout do not actively avoid carbaryl and that short term exposures do not affect olfaction at the concentrations tested. We found no other studies that evaluated other salmonids in estuarine conditions or in freshwaters. We ranked these study results as highly relevant to the effects of carbaryl on salmonid olfaction and an olfactory-mediated behavior, avoidance.

In one set of experiments, coho salmon exposed for 30 minutes to three carbamates (carbofuran, antisapstain IPBC, mancozeb) separately, had reduced olfactory ability as well as disruption of normal AChE activity (Jarrard, Delaney et al. 2004). Carbofuran reduced olfaction by 50% (EC50) at 10.4 μg/L; IPBC reduced olfaction at 1.28 μg/L (EC50); and mancozeb reduced olfaction at 2.05 mg/L (EC50). In addition, carbofuran reduced AChE activity in the olfactory receptor at at 200 μg/L (68%), but none of the treatments (0.1, 10, or 200 μg/L) reduced brain or olfactory bulb AChE activity. This could be a result of the limited exposure duration.   The authors concluded that too little information exists to develop a causal relationship between AChE inhibition and olfaction (Jarrard, Delaney et al. 2004). The data do show that carbofuran inhibited both olfaction and AChE activity at 10 and 200 μg/L, respectively. We ranked these study results as highly relevant to carbofuran's effect on salmonid olfaction.

Goldfish showed no avoidance of carbofuran up to a concentration of 10 mg/L, but at the onset of the contaminated flow goldfish showed increased and immediate burst swimming responses relative to unexposed fish (Saglio, Trijasse et al. 1996). In the same study, olfactometric tests were performed to assess the influence of 1, 10, and 100 μg/L carbofuran on a suite of behavioral responses to a food extract (a chironomid–containing solution). Sheltering, grouping, nipping, burst swimming, and attraction behaviors were assessed for ten minutes at 4, 8, and 12 h time points. Significant effects to goldfish behavior in the presence of food stimuli were observed as low as 1 μg/L carbofuran. Sheltering activity increased, while attraction decreased in a dose-dependent manner at 4 h across treatment range (1-100 μg/L) relative to unexposed fish. At 10 and 100 μg/L, goldfish nipped more, relative to controls; which may be representative of increased aggression and stress. The authors concluded that olfactory-mediated behaviors in goldfish were adversely affected by sublethal concentrations of carbofuran. We ranked this study as relevant to the assessment endpoint of olfaction, as ecologically relevant olfactory-mediated behaviors were tested with sublethal concentrations of carbofuran. We note that concentrations were not analytically verified and direct correlation of olfactory-affected goldfish behaviors to salmonid behaviors remains uncertain.

The olfactory ability of male Atlantic salmon to detect a female priming hormone was measured following 5 d exposures to carbofuran (1.1, 2.7, 6.5, 13.9, 22.7 μg/L) (Waring and Moore 1997). The response to the female pheromone prostaglandin F2α, by male olfactory epithelium was reduced at carbofuran concentrations as low as 1.0 μg/L., while the threshold for detection was reduced 10-fold. Furthermore, at 2.7 μg/L carbofuran, male fish completely lost priming ability induced by the female pheromone resulting in no increase in milt or plasma steroids at 2.7 μg/L. These results suggest that Atlantic salmon exposed to these concentrations would have difficulty preparing for spawning due to their impaired priming. Reductions in productivity are possible if male fish miss spawning opportunities. We ranked this study as highly relevant to the effects of carbofuran on olfactory–mediated behaviors such as spawning synchronization.

Mixtures containing carbaryl, carbofuran, and methomyl

We located no experiments that tested mixtures of the three a.i.s, nor did we find mixture studies with other *N*-methyl carbamates. Therefore, the potential for mixture toxicity to olfactory endpoints of salmonids remains a recognized data gap. Given the differences in olfactory toxicity in salmonids for carbaryl (no observed toxicity) and carbofuran (inhibited olfaction at 10 μg/L) combined with no available information on methomyl's affect on olfaction, no apparent dose-response pattern of toxicity emerges for the three a.i.s.

Assessment endpoints: Toxic effects in salmonids from consuming contaminated prey

Assessment measures: Survival, swimming performance

A current uncertainty is the degree to which secondary poisoning of juvenile salmonids may occur from feeding on contaminated dead and dying drifting insects. Secondary poisoning is a frequent occurrence with OPs and carbamates in bird deaths (Mineau 1991), yet is much less studied in fish. Uptake, metabolism, and accumulation of carbaryl by a salmonid prey item, *Chironomus riparius* (midge), exposed for 24 h indicated significant uptake over the first 8 h, significant metabolism (more than 85-99%) of parent carbaryl to metabolites and low bioconcentration factors (5-10) (Lohner and Fisher 1990). These results suggest that contaminated prey items, such as aquatic invertebrates, do not accumulate significant carbaryl, and what they do accumulate is likely rapidly metabolized. That said, juvenile salmonids could still get a dose of carbaryl from feeding on drifting, contaminated insects that have not had time to metabolize carbaryl. Juvenile brook trout gorged on drifting insects following applications of carbaryl and AChE activity was reduced (15-34%) in the trout (Haines 1981). However, it is not possible to differentiate the contribution to AChE inhibition from the aqueous and dietary routes because concentrations were not measured in the water, prey, or fish. In another study, resident trout feeding on dying and dead drifting invertebrates (from the pyrethroid cypermethrin) caused a range of physiological symptoms in brook trout: loss of self-righting ability and startle response; lethargy; hardening and haemolysis of muscular tissue similar to muscle tetany; and anemic appearance of blood and gills (Davies and Cook 1993). The possibility that the adverse effects in the trout manifested from exposure to the water column instead of from feeding on contaminated prey was ruled out by the authors as measured field concentrations of pesticides did not produce known toxic responses. In a laboratory feeding study with the OP fenitrothion, brook trout (*S. fontinalis*) were fed contaminated pellets (1 or 10 mg/g fenitrothion for four wks) (Wildish and Lister 1973). Growth was reduced in both treatments. AChE inhibition was measured at 2, 12, and 27 d following termination of contaminated diet treatments. Trout had lower AChE activity than unexposed fish at both treatments, and by 27 d following termination, AChE levels recovered slightly. The treatment concentrations used in this study are very high and indicate that brook trout are not sensitive to diet-induced toxicity of fenitrothion. The experiment did show that AChE inhibition from the diet is possible, yet it is difficult to determine the relative toxicity of carbaryl, carbofuran, and methomyl found in contaminated insects consumed by Pacific salmonids.

Habitat assessment endpoints:

Prey survival, prey drift, nutritional quality of prey, abundance of prey, health of aquatic prey community, and recovery of aquatic communities following N-methyl carbamate exposures

Assessment measures: 24, 48, and 96 h survival of prey items from laboratory bioassays reported as EC/LC50s; sublethal effects to prey items; field studies on community abundance; indices of biological integrity (IBI); community richness; and community diversity.

Death of aquatic invertebrates in laboratory toxicity tests were reported in each of the BEs. Salmonid aquatic and terrestrial prey are highly sensitive to the three carbamate insecticides. Death of individuals and reductions in individual taxa and prey communities have been documented and are expected following applications of *N*-methyl carbamates that achieve effect concentrations-several of which are at the low μg/L levels (Schulz 2004). Complete or partial elimination of aquatic invertebrates from streams contaminated by insecticides has been documented for carbaryl (Muirhead-Thomson 1987) as well as many other insecticides. A review of field studies published from 1982-2003 on insecticide contamination concluded that "about 15 of the 42 studies revealed a clear relationship between quantified, non-experimental exposure and observed effects in situ, on abundance [aquatic invertebrate], drift, community structure, or dynamics" (Schulz 2004). Although the top three insecticides most frequently detected at levels expected to result in toxicity were chlorpyrifos (OP), azinphos-methyl (OP), and endosulfan, the *N*-methyl carbamates carbaryl, carbofuran, oxymyl, and fenobucarb all showed clear or assumed relationships between exposure and effect (Schulz 2004).

Drift, feeding behavior, swimming activity, and growth are sublethal endpoints of aquatic prey negatively affected by exposure to AChE inhibitors (Courtemanch and Gibbs 1980; Haines 1981; Hatakeyama, Shiraishi et al. 1990; Davies and Cook 1993; Beyers, Farmer et al. 1995; Schulz 2004). Drift of aquatic invertebrates is an evolutionary response to aquatic stressors. However, insecticides, particularly carbamates and OPs, can trigger catastrophic drift of salmonid prey items (Courtemanch and Gibbs 1980; Haines 1981; Hatakeyama, Shiraishi et al. 1990; Davies and Cook 1993; Schulz 2004). Some invertebrates may drift actively to avoid pesticides and settle further downstream, which can provide temporary spikes in available food items for feeding salmonids. Catastrophic drift can also deplete benthic populations resulting in long-term prey reduction that may affect salmonid growth at critical time periods. We located no studies that address this line of reasoning directly with Pacific salmonids. Davies and Cook (1993) did show aquatic invertebrate community changes, mortality of invertebrates, drift of dying and dead invertebrates, and affected trout following spraying of a pyrethroid pesticide, cypermethrin, an invertebrate and fish neurotoxicant (Davies and Cook 1993). Effect concentrations were estimated at 0.1-0.5 μg/L cypermethrin. It is difficult to compare these effect concentrations to carbamate insecticides. However, it is illustrative of how insecticides can damage multiple endpoints of an aquatic community including reducing abundance of prey (Davies and Cook 1993).

Several scientific peer-reviewed publications and EPA documents have reviewed aspects of the available information on multi-organism microcosm, mesocosm, and field test results for the AChE insecticides (Leeuwangh 1994; Barron and Woodburn 1995; Schulz 2004; Van Wijngaarden, Brock et al. 2005). Van Wijngaarden et al. (2005) conducted a literature review that listed ecological threshold values (*e.g.,* $NOEC_{eco}$ and $LOEC_{eco}$) for carbaryl, carbofuran, and bendiocarb (another *N*-methyl carbamate) from model ecosystems or “adequate” field studies. A $NOEC_{eco}$ represented “the highest tested concentration at which no, or hardly any, effects on the structure and functioning of the studied model ecosystem were observed. The $LOEC_{eco}$ is the lowest tested concentration at which significant treatment-related effects occurred” (Van Wijngaarden, Brock et al. 2005). Below we discuss some of this information in relation to effects on salmonid prey. The majority of studies were conducted in littoral systems, *i.e.,* ponds, and other static systems, but one study with carbaryl was conducted in a running water (lotic) system (Courtemanch and Gibbs 1980). Population densities of salmonid prey items (*i.e.*, Ephemeroptera, Diptera, Amphipoda, Cladocera, Copepoda, Isopoda, Ostracada, Trichoptera) decline following exposures to AChE-inhibiting insecticide concentrations, including carbaryl and carbofuran (Van Wijngaarden, Brock et al. 2005). Adverse effects to these groups occurred at or below "1 toxic unit"-where a toxic unit equals field concentrations normalized by dividing them by the 48 h EC50 of *Daphnia magna* for a given AChE inhibitor.

We did not locate any microcosm, mesocosm, or field experiments that measured responses of aquatic communities that contained salmonids and salmonid prey simultaneously; a recognized data gap. Several studies evaluated aquatic invertebrate responses to *N*-methyl carbamates (carbaryl, carbofuran, and bendiocarb) in static and running water systems. We found no field studies with methomyl. We summarize open literature studies with aquatic invertebrates organized by insecticide in Table 65. We found no studies that addressed effects of methomyl on aquatic invertebrates in the field or from multispecies microcosms or mesocosms.

The available literature from field experiments indicates that populations of aquatic insects and crustaceans are likely the first aquatic organisms damaged by exposures to carbaryl, carbofuran, and methomyl contamination. Benthic community shifts from sensitive mayfly, stonefly and caddisfly taxa, the preferred prey of salmonids, to worms and midges occur in areas with degraded water quality including from contaminants such as pesticides (Cuffney, Meador et al. 1997; Hall, Killen et al. 2006). Reduced salmonid prey availability correlated to OP use in salmonid bearing watersheds (Hall, Killen et al. 2006). We found no studies that evaluated the effects on carbamates and that made correlations to salmonid bearing watersheds. Subsequent effects to salmonid's growth from reduced prey availability and quality remain untested and are a current data gap.

We located a study addressing impacts on fish growth due to prey reduction. Although this study was conducted on chlorpyrifos, an insecticide not considered in this Opinion, we deemed it highly relevant due to the ecological context it provided. The study indicated that native fathead minnows exposed to chlorpyrifos had reduced growth due to reductions in prey item abundance in littoral enclosures (pond compartments) (Brazner and Kline 1990). The experiment tested the hypothesis that, "addition of chlorpyrifos would reduce the abundance of invertebrates and cause diet changes that would result in reduced growth rates." Nominal, chlorpyrifos treatment concentrations of 0.5, 5.0, and 20 μg/L (chemical analysis of water concentrations provided at 0, 12, 24, 96, 384, 768 h) all resulted in statistically significant reductions in growth at 31 days. A single pulse of chlorpyrifos was introduced into each enclosure at day 0. Invertebrate abundance was determined in each replicate on days -3, 4, 16, and 32. Fathead minnows were sampled from enclosures on days -2, 7, 15, and 31 where fish were weighed, measured, and dissected to determine gut content (dietary items identified). By day 7, significant differences in mean numbers of rotifers, cladocerans, protozoans, chironomids, mean total number of prey being eaten per fish, and mean species richness were greater in fish from the control enclosures than in some of the treatments. By day 15, control minnows were significantly larger than fish from treated levels. These experimental results support the conclusion that reductions in abundance of prey to juvenile fish can result in significant growth effects. It is reasonable to assume that reductions in prey from carbamate insecticides can also result in reduced juvenile salmonid growth and ultimately reduced survival and productivity. The precise levels of prey reduction necessary to cause subsequent reductions in salmonid growth remain a recognized data gap.

Recent declines in aquatic species in the Sacramento-San Joaquin River Delta in California have been attributed in part to toxic pollutants, including pesticides (Werner, Deanovic et al. 2000). Significant mortality or reproductive toxicity in *C. dubia* was detected in water samples collected at 24 sites in the Sacramento-San Joaquin River Delta in California. Ecologically important back sloughs had the largest percentage of toxic samples (14 - 19%). The TIEs identified carbofuran and carbaryl as well as the OPs chlorpyrifos, diazinon, and malathion as the primary toxicants in these samples responsible for the adverse effects (Werner, Deanovic et al. 2000).

Recovery of salmonid prey communities following acute and chronic exposures from carbaryl, carbofuran, and methomyl depends on the organism’s sensitivity, life stage, length of life cycle, among other characteristics. Univoltine species will take longer than multivoltine species to recover (Liess and Schulz 1999). Recovery of salmonid prey items such as caddisflies, stoneflies, and mayflies will be slow, considering their long life cycles and infrequent reproduction. Additionally, these species also require clean, cool waters to both recover and maintain self-sustaining populations. In several salmonid-supporting systems these habitats are continually exposed to anthropogenic disturbances, including pesticide contamination, which limits their recovery and can also limit recovery of multivoltine species. For example, urban environments are seasonally affected by stormwater runoff that introduces toxic levels of contaminants and scours stream bottoms with high flows. Consequently, urban environments do not typically support diverse communities of aquatic invertebrates (Paul and Meyer 2001; Morley and Karr 2002).

Similarly, yet due to a different set of circumstances, watersheds with intensive agriculture land uses show compromised invertebrate communities (Cuffney, Meador et al. 1997). Indices of biological integrity (IBI) and other invertebrate community metrics are useful measures of the health of an aquatic community because cumulative impacts of aquatic stressors are integrated over time. The IBI is also valuable because it converts relative abundance data of a species assemblage into a single index of biological integrity (Allan 1995). Salmonid-inhabited watersheds have been assessed using IBIs and other metrics of aquatic community health.

A study on the condition of Yakima River Basin's aquatic benthic community found that invertebrate taxa richness was directly related to the intensity of agriculture *i.e.*, at higher agriculture intensities taxa richness declined significantly both for invertebrates as well as for fish (Cuffney, Meador et al. 1997). Locations with high levels of impairment were associated with high levels of pesticides and other agricultural activities, which together with habitat degradation were likely responsible for poor aquatic conditions in the Yakima (Cuffney, Meador et al. 1997). Salmonid ESUs and DPSs occur in the Yakima River Basin as well as other watersheds where invertebrate community measurements indicate severely compromised aquatic invertebrate communities such as the Willamette River Basin, Puget Sound Basin, and the Sacramento-San Joaquin River Basin.

Table 65. Study designs and results with freshwater aquatic invertebrates

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
Carbaryl	multiple n= 25; including predators, herbivores, zooplankton	species richness, biomass, abundance, survival	510 μg/L; single application; simulating direct overspray	13 d	Reduced richness of community (by 15%), predators, zoo plankton; reduced biomass of predators; increased abundance of large herbivores; No effect on zooplankton abundance; elimination of diving beetle larvae, *Daphnia pulex*, *D. ambigua*	(Relyea 2005)
Carbaryl	*C. californica* (stonefly) *C. sp.* (mayfly) *A. sp* (mayfly) *B. americanus* (caddisfly) *P. sp.* (caddisfly) early instar *P. sp.* (caddisfly) late instar *L. unicolor* (caddisfly)		1-30 μ g/L 4-100,000 μ g/L 10-28 μg/L 5-55 μ g/L 5-80 μ g/L 5-80 μ g/L 5- 80 μ g/L			(Peterson, Jepson et al. 2001)
Carbaryl	Coleopteran (beetles), Diptera (flies), Trichoptera (caddisflies), terrestrial insects, Emphemeroptera (mayflies)	salmonid stomach contents	0.5 lbs/acre, 2 applications, 7 days interval	na	Increased numbers of Diptera in stomach after first spray; increased numbers of Emphemeroptera, Trichoptera, Diptera, and terrestrial insects indicating drift; condition factor of fish increased following applications due to increased feeding; fish AChE inhibited 15-34%fish AChE inhibited 15-34%	(Haines 1981)
Carbaryl	*Chironomus riparius* (midge)	Survival (EC50); Uptake rate; Bioconcentration factors (BCFs)	Multiple concentration ranges at pH 4,6,8; temps 12, 20, 30 ˚C	24 h	EC50s (61-133 μg/L); Lowest EC50 at 30 ˚C, pH 4 and 6; Highest EC50s at 10˚C, pH 4 and 6. Uptake rates increased with temp. BCFs = 5.36-10.12	(Lohner and Fisher 1990)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
Carbaryl	*C. californica* (stonefly) *Cinygma sp.* (mayfly)	Survival (EC50)	17.3, 173, 1730 μ g/L 10.2, 102, 204, 408, 1020 μ g/L	15 min 30 60 15 min 30 60 96 h recovery period	EC50= na EC50= na EC50 = 1139 95% CI (370-15400) μ g/L EC50 = 848 no CI EC50 = 220 no CI EC50 = 165 95% CI (124-232)	(Peterson, Jepson et al. 2001)
carbaryl	Stream invertebrate aquatic community	Aquatic invertebrate drift, survival; presence and abundance of riffle invertebrates; recovery of benthic invertebrates	0.75 lbs/ acre; 1 lb/acre	2, 30, 60, 365 day drift sampling post spray; 1,2, 3, 30, 60 day benthic samples	170-fold increase in drift at 2 d. All invertebrates in drift samples dead at 2 d (Plecoptera, Ephemeroptera Diptera were common), at 30 and 60 d drift below prey-spray levels. Within hrs larger stream inverts found dead (Plecoptera, Trichoptera, Emphemerptera), Tricoptera abandoned their cases; at 30 and 60 d significant population declines of salmonid-prey invertebrates compared to pre-spray levels. Total number of organisms not significantly reduced.	(Courtemanch and Gibbs 1980)
carbaryl	*Xathocnemis zealandica* (damselfly)	Emergence of damselflies, % emerged	1, 10, 100 μ g/L (concentrations replenished every 12 days at ½ the nominal treatment level)	67 days	90% reduction in emergence at 100 μg/L no significant differences at 1 and 10 μg/L.	(Hardersen and Wratten 1998)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbaryl	*Daphnia magna* (daphnids)		15 μg/L applied as a single pulse at different population phases (treatments): growing, density peak, stable	1 pulse at day 8 to growing treatment (3 replicates), day 11 to density peak treatment , day 25 stable treatment; 48 h		(Takahashi and Hanazato 2007)
carbaryl	Zooplankton pond community	Abundance of zooplankton species	500 μg/L; 1-3 applications to pond mesocosms; 0.1 mg/L	0 – 3 months	At 500 μg/L sustained reductions in daphnid and copepod populations. No observable reductions in rotifer populations. At 100 μg/L, reductions in daphnid population observed, but not with copepods or rotifers	(Hanazato and Yasuno 1990)
carbaryl formulation (43.3% a.i.)	Bluegill sunfish, multiple aquatic species including phytoplankton, macrophytes, zooplankton, and macroinvertebrates	Abundance and richness of species groups; Fish survival and growth	2, 6, 20, 60, 200 μg/L; applications at weekly intervals for 6 weeks	0 – 12 weeks, with 6 weekly pulses beginning at week 10	no long term reductions in major invertebrate groups were observed; no detectable differences in bluegill survival or growth between control and pesticide treatments, carbaryl half life was 30 min, mean pH of treatment groups in application events ranged from 9.2-10.3; mean temperatures ranged from 17 – 27 ◦C, with the majority between 20-24 ◦C.	Registrant-submitted study (1993)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
	Aquatic macroinvertebrate community; *Platygobio gracilis* (Flathead chub)	Invertebrate drift; AChE inhibition in resident fish	Aerial application in 1991 and 1993 applied at 0.5 lbs carabaryl/acre and 0.4 lbs/acre, respectively	River water sampled at 1, 2, 4, 8, 12, 24, 48, 96 h; AChE inhibition measured at 24 h; Drift sampled 4 times daily for 4 days	1 h average 85 μg/L and declined to 0.100 μg/L by 96 h in 1991; 2 h average 12 μg/L L and declined to 5.14 μg/L by 96 h in 1993; No detectable differences in AChE inhibition at 24 h, low statistical power (0.82 and 1); Note: AChE is expected to recover by 24 h; Increase in coefficient of variation of drift measure from day 1and 2 post application compared to reference site drift in 1991; Similar increase on day 2 in 1993; drift composed primarily of live mayflies (Emphemeroptera)	(Beyers, Farmer et al. 1995)
carbofuran	*Gammarus pulex* (fw amphipod)	*in situ* survival, feeding rates; laboratory bioassays measured 24 h survival (LC50) and feeding rates		Field study: 5 day; 2 rain events captured Lab study: Survival- 24-, 48- 96-hr Feeding rate- 7 day exposures		(Matthiessen, Shearan et al. 1995)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbofuran	1 h average 85 μg/L and declined to 0.100 μg/L by 96 h in 1991; 2 h average 12 μg/L L and declined to 5.14 μg/L by 96 h in 1993; No detectable differences in AChE inhibition at 24 h, low statistical power (0.82 and 1); Note: AChE is expected to recover by 24 h; Increase in coefficient of variation of drift measure from day 1and 2 post application compared to reference site drift in 1991; Similar increase on day 2 in 1993; drift composed primarily of live mayflies (Emphemeroptera)	Taxa abundance and biomass	5 μ g/L 25 μ g/L Note: pH of 9 in experimental pond reduced persistence of carbofuran by increasing the rate of hydrolysis (Chapman and Cole 1982)	Taxa assessed at 5, 11, 32, 55 days post application	No detectable reductions to abundance or biomass of taxa from 5 μg/L; *H azteca*: 10% reduction in abundance; 6% reduction in biomass Chironominae: 77% dead in 48-72 h dip net samples compared to 0% dead in control and 5 μg/L treatment; biomass reduced by 17% at 55 d; evidence that younger larvae were less sensitive than older larvae; No detectable reductions in remaining taxa	(Wayland 1991)
carbofuran	Aquatic invertebrate community in rice fields: ostracada, copepod, cladocera, mosquito larvae, chironomid larvae	Abundance of ostracada, copepod, cladocera, mosquito larvae, chironomid larvae	0.1 kg carbofuran/ha- applied once 0.3 kg carbofuran/ha applied in two regimes: 3, 52, 85, 97 days; 16, 57, 69 days	variable application intervals and frequencies	Ostracada: timing and magnitude of peak abundances not affected by carbofuran, multiple application regimes of 0.3 k/ha significantly reduced abundance compared to one application of 0.1 kg/ha Copepods: significant reductions at 45 and 57 d at 0.3 kg/ha. Cladoceran: rate of population growth significantly affected from 52 and 85 days at 0.3 kg/ha. Abundance lower at 0.1 kg/ha compared to 0.3 kg/ha. Chironomid larvae: isolated significant differences in abundances were found (*P*<0.05), but showed no dose-response relationship Mosquito larvae: No change in abundances	(Simpson, Roger et al. 1994)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbofuran	*Cerodaphnia dubia* (daphnids)	Survival (24 and 48 h LC50s); survival from mixture of carbofuran and methylpararthion (48 h LC50); Mean young per female and percent survival from 7 day chronic assays	Irrigation field-collected water samples (toxicity identification evaluations employed) Mixture toxicity (1:1, 3:1, 1:3 ratios tested) 0.16, 0.33, 0.65, 1.3, 2.6 μg/L	24, 48 hours 7 days	Carbofuran and methyl-parathion putative agents of toxicity in field-collected samples from CA rice drain. Carbofuran acute toxicity: 24 h LC50 3.4 μg/L (95% CI 2.8-4.3; 48 h LC50 2.6 μg/L (95% CI NA); chronic study- NOEC 1.3 μg/L; LOEC at 2.6 μ g/L Strict additive toxicity of combinations tested of carbofuran and methyl-parathion.	(Norberg-King, Durhan et al. 1991)
carbofuran	*Chironomus tentans* (midge)	Survival in spiked-sediments, mixed with atrazine, and field-collected runoff		10 days	Spiked sediment results: Nominal [carbofuran] LC50 47.9 μg/kg (95% CI 43.9-52.1); Sediment bound LC50 20.9 μg/kg (95% CI 18.4-33.2); Interstitial water LC50 11.8 μg/L (95% CI 10.9-17.1) Mixture results: Additive toxicity present with atrazine + carbofuran, no synergistic or antagonistic toxicity responses. Atrazine affected survival in the low mg/kg range. Field collected sediments in runoff: No chironomids survived exposure, note control survival was low (57%)	(Douglas, McIntosh et al. 1993)
carbofuran	*Chorophium volutator* (estuarine amphipod)	Survival and avoidance behavior in contaminated spiked sediments	2.5, 25, 250 ng/g (dry weight sediment)	48 and 96 h exposures	48 h LC 50 = 73 ng/g (95% CI 0-128); 48 h LC20 = 41 (95% CI 2-82) ng/g; no effect on avoidance behavior of sediment	(Hellou, Leonard et al. 2009)

Chemical	Taxa/species	Assessment measures	Concentrations tested	Exposure duration	Effects	Data source
carbofuran	*Daphnia magna* (daphnid) *Hydra attenuate* (fw jellyfish) *Macrobrachium rosenbergii* (fw shrimp) Aquatic macroinvertebrate community	Survival (acute LC50); macroinvertebrate community structure	Average concentrations detected 5 d post application 0.7 and 0.16 μg/L maximum carbofuran detection 2.1 μ g/L	24 , 48, 72, 96 h exposures to field collected water samples	No acute toxicity reported to tested species. Significant reduction of % EPT taxa and increase in community loss indices at treated sites. Mayflies were primary affected species from carbofuran applications	(Castillo, Martinez et al. 2006)
carbofuran	*Ceriodaphnia dubia (daphnid)* *Neomysis mercedi(* mysid*)* *Pimephales promelas (*fathead minnow*)*		Not reported; conducted toxicity identification evaluations (TIE)	96 h exposures to field collected water samples	3/47 irrigation samples had 100% toxicity to *C. dubia* attributed to carbofuran	(de Vlaming, DiGiorgio et al. 2004)
carbofuran	Wetland invertebrate aquatic community*:* *Daphnia magna,* *Chironomus riparius*		10, 100, 1,000 μg/L water only 48 h test; 10, 100, 1,000 μg/L mixed with soil and added to microcosms	48- hr; 30 day microcosm test	*D. magna* EC50 = 48 μ g/L *C. riparius* EC50 = 56 μ g/L Populations of *D. magna* were viable at day 1 in 10 and 100 μg/L, it took 4 d for a viable population (*i.e.,* 5 or more live daphnids) to establish at 1000 μg/L . No dead daphnids or chironomids observed at 14 d and 30 d. No effects to microbial community enzyme activity were found.	(Johnson 1986)

Studies with other AChE inhibitors on salmonid prey items:

Robust evidence shows that salmonid prey taxa and communities can be substantially reduced following exposures to the OP insecticides chlorpyrifos, diazinon, and malathion. NMFS reviewed these data and presented its findings in a biological opinion (NMFS 2007d). We use these findings to show the types of aquatic community responses following exposures to AChE inhibiting insecticides. The toxic potency of a pesticide is a function of concentration and duration of exposure, which in turn is a function of a pesticide's physical properties and interactions of the pesticide with environmental variables such as temperature, pH, sunlight, soil micro-organisms, etc. With this in mind, if carbaryl, carbofuran, and methomyl are at concentrations individually (or together in mixtures) expected to reduce salmonid prey communities, we infer a similar magnitude of response (reductions in abundance from death and catastrophic drift) and similar recovery period for those observed in affected aquatic communities treated with OP insecticides.

We note that individual aquatic invertebrates exhibiting adverse sublethal responses from carbamates will likely recover much more quickly than those exposed to OPs (Kallander, Fisher et al. 1997). The midge, *Chironomus tentans*, showed complete recovery of AChE activity in 24 h following two 1 h pulses of carbaryl separated by 24 h. Additionally, two, 1 h pulses of carbamates (carbaryl, carbofuran, aldicarb, propoxur) caused significantly fewer symptoms of intoxication than 2 h of continuous exposure when chironomids were given 2 to6 h of recovery in clean water between doses (Kallander, Fisher et al. 1997). In contrast, sublethal exposures to OPs were equally toxic when exposures are either pulsed or continuous.

Reviews of field, mesocosm, and microcosm studies with the three OPs document reductions in aquatic invertebrate populations and lengthy recovery times for populations of some taxa (Giesy, Solomon et al. 1998; Van Wijngaarden, Brock et al. 2005). A recent study found significant changes to macroinvertebrate assemblages of artificial stream systems following a 6 h exposure to chlorpyrifos at 1.2 μg/L (Colville, Jones et al. 2008). The addition of chlorpyrifos to the artificial streams resulted in a rapid (6 h) change in the macroinvertebrate assemblages of the streams, which persisted for at least 124 d after dosing (Colville, Jones et al. 2008). The chlorpyrifos dissipated from the system within 48 h (Pablo, Krassoi et al. 2008), however the macroinvertebrate community did not recover rapidly. Several species similar to salmonid prey items were significantly affected.

Zooplankton and insect taxa appeared the most sensitive in studies with diazinon. In particular, the salmonid prey taxa Trichoptera, Diptera, and Cladocera were highly sensitive (Giddings, Hall et al. 2000). Field studies in salmonid habitat also show reductions in salmonid prey abundances. For example, in listed steelhead habitat in the Salinas River, California, abundances of the salmonid prey items including mayfly taxa, daphnids, and an amphipod (*Hyalella azteca*) were significantly reduced downstream of an irrigation return drain compared to upstream (Anderson, Hunt et al. 2003; Anderson, Hunt et al. 2003; Anderson, Phillips et al. 2006). Diazinon and chlorpyrifos were detected above acute toxicity thresholds in surface waters and sediments. Combined toxicity of the two OPs using a toxic unit approach correlated strongly with mortality of daphnids. For *H. azteca*, acute toxicity was attributed to sediment pore-water concentrations of chlorpyrifos (Anderson, Hunt et al. 2003). Other pesticides, including carbaryl, were likely present and responsible for some of the toxicity in the Salinas River,. In a subsequent study on the Salinas River, TIE demonstrated that chlorpyrifos and diazinon were responsible for the observed death of the daphnid *C. dubia* (Hunt, Anderson et al. 2003). These data support the line of evidence that field concentrations of OPs can and do adversely affect aquatic invertebrates in salmonid habitats. It is reasonable to assume that the same situation occurs with carbamates, given the toxicity of the compounds and the similar mode of action.

Adjuvant toxicity

Assessment endpoints: Survival of fish and aquatic prey items, endocrine disruption in fish

Assessment measures: 24, 48, 96 h LC50s, and vitellogenin levels in fish plasma

Although no data were provided in the BEs related to adjuvant toxicity, an abundance of toxicity information is available on the effects of the alkylphenol polyethoxylates, a family of non-ionic surfactants used extensively in combination with pesticides as dispersing agents, detergents, emulsifiers, adjuvants, and solubilizers (Xie, Thrippleton et al. 2005). Two types of alkylphenol polyethoxylates, NP ethoxylates and octylphenol ethoxylates, degrade in aquatic environments to the more persistent, toxic, and bioaccumulative degradates, NP and octylphenol, respectively. We note that the technical registrant of methomyl stated that no nonylphenol ethoxylates are used within methomyl formulations. We did not receive information on the presence or absence of alkylphenol polyethoxylates in carbaryl- or carbofuran- containing formulations. Adjuvants are frequently mixed with formulations prior to applications, so although they may not be present in the formulations they could still be co-applied. Below we discuss NP's toxicity as an example of potential adjuvant toxicity, as we received no information on adjuvant use or toxicity within the BEs.

We queried EPA's ECOTOX online database and retrieved 707 records of nonylphenol's (NP) acute toxicity to freshwater and saltwater species. The lowest reported LC50 for a salmonid was 130 μg/L for Atlantic salmon. Aquatic invertebrates, particularly crustaceans, were killed at low concentrations of NP, with the lowest reported LC50 = 1 μg/L for *H. azteca*. These data indicate that a wide array of aquatic species are killed by NP at μg/L concentrations. We also queried EPA's ECOTOX database for sublethal toxicity and retrieved 689 records of freshwater and saltwater species tested in chronic experiments. The lowest fish LOEC reported was 0.15 μg/L for fathead minnow reproduction. Numerous fish studies reported LOECs at or below 10 μg/L. Additionally, salmonid prey species are sensitive to sublethal effects of NP at low μg/L concentrations. The amphipod, *Corophium volutator*, grew less and had disrupted sexual differentiation (Brown, Conradi et al. 1999). Multiple studies with fish indicated that NP disrupts fish endocrine systems by mimicking the female hormone 17β-estradiol (Brown and Fairchild 2003; Arsenault, Fairchild et al. 2004; Madsen, Skovbolling et al. 2004; Jardine, MacLatchy et al. 2005; Luo, Ban et al. 2005; McCormick, O'Dea et al. 2005; Segner 2005; Hutchinson, Ankley et al. 2006; Lerner, Bjornsson et al. 2007; Lerner, Bjornsson et al. 2007). NP induced the production of vitellogenin in fish at concentrations ranging from 5-100 μg/L (Hemmer, Bowman et al. 2002; Ishibashi, Hirano et al. 2006; Arukwe and Roe 2008; Schoenfuss, Bartell et al. 2008). Vitellogenin is an egg yolk protein produced by mature females in response to 17-β estradiol, however immature male fish have the capacity to produce vitellogenin if exposed to estrogenic compounds. As such, vitellogenin is a robust biomarker of exposure. A retrospective analysis of an Atlantic salmon population crash suggested the crash was due to NP applied as an adjuvant in a series of pesticide applications in Canada (Fairchild, Swansburg et al. 1999; Brown and Fairchild 2003). Additionally, processes involved in sea water adaptation of salmonid smolts are impaired by NP (Madsen, Skovbolling et al. 2004;

Jardine, MacLatchy et al. 2005; Luo, Ban et al. 2005; McCormick, O'Dea et al. 2005; Lerner, Bjornsson et al. 2007; Lerner, Bjornsson et al. 2007).

These results demonstrate NP is of concern to aquatic life, particularly salmonid endocrine systems involved in reproduction and smoltification. This summary is for one of the more than 4,000 inerts/other ingredients and adjuvants currently registered for use in pesticide formulations and there are likely others with equally deleterious effects. Unfortunately we received minimal information on the constituents found in carbaryl-, carbofuran-, and methomyl-containing formulations. Consequently, the effects that these other ingredients may have on listed salmonids and designated critical habitat remain an uncertainty and are a recognized data gap in EPA's action under this consultation.

Summary of Response Analysis:

We summarize the available toxicity information by assessment endpoint in Table 66. Data and information reviewed for each assessment endpoint was assigned a general qualitative ranking of either "low", "medium", or "high." To achieve a high confidence ranking, the information stemmed from direct measurements of an assessment endpoint, conducted with a listed species or appropriate surrogate, and was from a well-conducted experiment with stressors of the action or relevant chemical surrogates. A medium ranking was assigned if one of these three general criteria was absent and low ranking was assigned if two criteria were absent. Evidence of adverse effects to assessment endpoints for salmonids and their habitat from the three a.i.s was prevalent. However, much less information was available for other ingredients, in part, due to the lack of formulation information provided in the BEs as well as the statutory mandate under FIFRA for toxicity data on the a.i.s to support registration. We did locate a substantial amount of data on one group of adjuvants/surfactants, the NP ethoxylates. However, we received and located minimal information for the majority of tank mixes and other ingredients within formulations.

Table 66. Summary of assessment endpoints and effect concentrations

Assessment Endpoint	Evidence of adverse responses (yes/no)	Concentration range of observed effect or concentrations tested showing absence of effect (μg/L)	Degree of confidence in effects (low, medium, high)
Carbaryl			
Fish:			
-survival (LC50)	yes	250-4,500	high
-growth	no	8,17,62,210, 680	medium
-reproduction	yes	680	medium
-swimming	yes	10-12,500	high
-olfactory-mediated behaviors	no	5, 50, 500	high
Habitat: -prey survival (LC50)	yes	1.5-35[1]	high
Carbofuran			
Fish:			
-survival (LC50)	yes	88-3,100	high
-growth	yes	56.7	medium
-reproduction	yes	6	low
-swimming	yes	5, 100	medium
-olfactory-mediated behaviors	yes	1-200	high
Habitat: -prey survival (LC50)	yes	1.6[2]-2,700	high
Methomyl			
Fish:			
-survival (LC50)	yes	300-7,700	high
-growth	yes	142, 243	medium
-reproduction	no	31-500	low
-swimming	-	-	-
-olfactory-mediated behaviors	-	-	-
Habitat: -prey survival (LC50)	yes	8.8-920	high
Other ingredient:			
Nonylphenol (NP)			
Fish:			
-survival (LC50)	yes	130 - >1,000	high
-reproduction	yes	0.15 - 10	high
-smoltification	yes	5 - 100	medium
-endocrine disruption	yes	5.0 – 100	high
Habitat: -prey survival (LC50)	yes	1- >1,000	high
Additive toxicity of *N*-methyl carbamates	yes	multiple	high
Synergistic toxicity of *N*-methyl carbamates	yes	multiple	high

[1] reported 48 h EC50s of *D. magna* are low and high values of the range (Takahashi and Hanazato 2007)
[2] 48 h survival EC50 of *Chironomus tentans* (Karnak and Collins 1974)

Risk Characterization

In this section we integrate our exposure and response analyses to evaluate the likelihood of adverse effects to individuals, populations, species, and designated critical habitat (Figure 38). We combined the exposure analysis with the response analysis to: 1) determine the likelihood of salmonid and habitat effects occurring from the stressors of the action; 2) evaluate the evidence presented in the exposure and response analyses to support or refute risk hypotheses; 3) translate fitness level consequences of individual salmonids to population-level effects; and 4) translate habitat-associated effects to potential impacts on PCEs of critical habitat. The risk characterization section concludes with a general summary of species responses from population-level effects. We then evaluate the effects to specific ESUs and designated critical habitat in the *Integration and Synthesis* section.

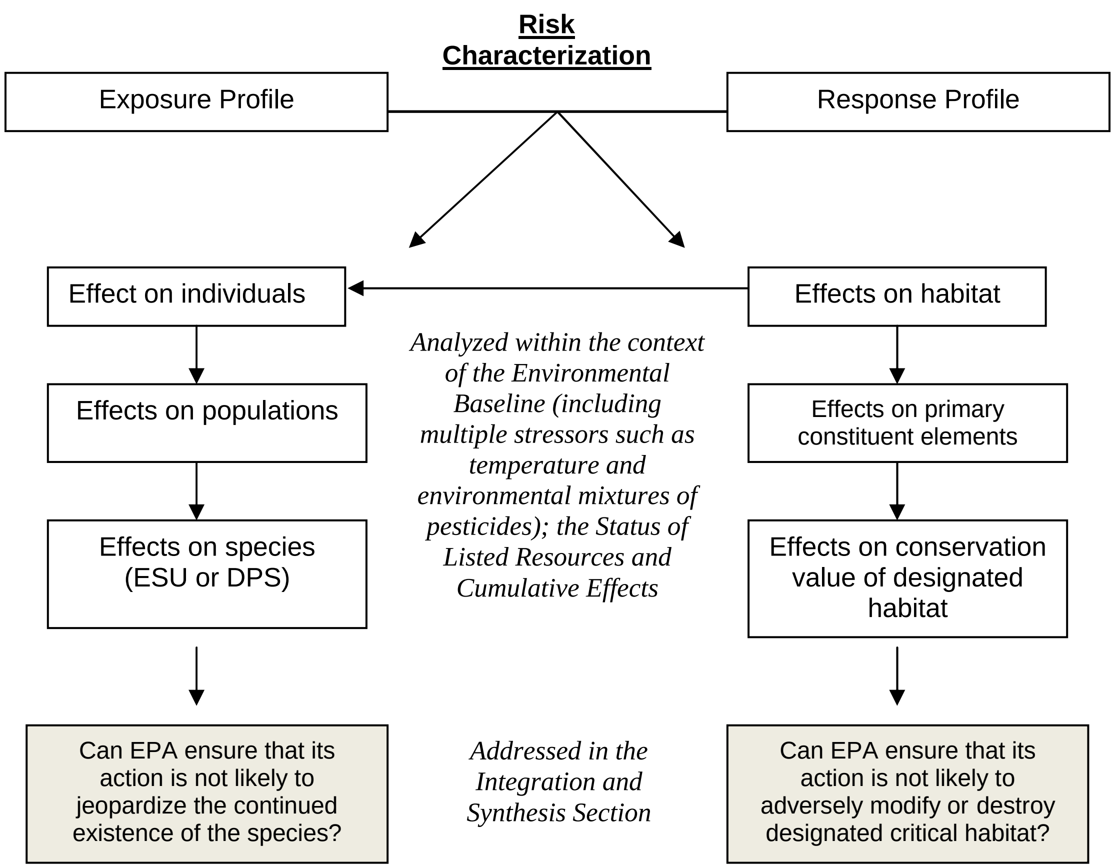

Figure 38. Schematic of the Risk Characterization Phase

Exposure and Response Integration

In Figures 39-41, we show the overlap between exposure estimates for the three carbamates and concentrations that affect assessment endpoints. The figures show the exposure concentration ranges (minimum – maximum values) gleaned from the three predominant sources of exposure data we analyzed: monitoring data; EPA's estimates presented in the BEs that represent crop uses; and NMFS' modeling estimates for off-channel habitats. None of the modeled exposure estimates were derived for non-crop use. This is a major data gap as carbaryl is used extensively in urban and residential areas. The effect concentrations are values taken from the toxicity data reviewed in the *Response Analysis Section*. With respect to the assessment endpoint survival, recall that the effect concentrations are LC50s, thus death of sensitive individuals is not represented by this metric and can occur at concentrations well below LC50s. Additionally, we cannot accurately predict at what concentrations death first occurs because dose-response slope information was generally not provided for most of the acute lethality studies. Although we are unable to determine at what concentration an individual organism might die, we do incorporate survival endpoints from acute 96 h studies using a default slope in a population modeling exercise discussed below. This slope is recommended by EPA when more relevant information is unavailable (EPA 2004). Where overlap occurs between exposure concentrations and effect concentrations NMFS explores the likelihood of adverse effects. If data suggest exposure exceeds adverse effects thresholds, we discuss the likelihood and expected frequency of effects based on species information and results of the exposure and response analyses.

This is a coarse analysis because it does not present temporal aspects of exposure nor does it show the distribution of toxicity values. However, it does allow us to systematically address which assessment endpoints are affected from carbaryl, carbofuran, and methomyl exposure. Where significant uncertainty arises, NMFS highlights the information and discusses its influence on our inferences and conclusions. We discuss the uncertainties related to these endpoints under associated risk hypotheses later in this section.

Carbaryl

Concentration ranges overlap with most of the assessment endpoint ranges indicating that adverse effects are expected in salmonids if exposed for sufficient durations (Figure 39). Prey survival appears to be the most sensitive endpoint as all three ranges encompass this endpoint and the maximum concentration values far exceed the prey survival range. Swimming is likely impaired at the higher end of the concentration range from monitoring data, at the middle and higher range from EPA estimates, and throughout the range predicted in the off-channel habitat estimates. Concentrations occurring in the off-channel habitats that NMFS modeled will kill juvenile salmonids. Furthermore, given the LC50 values for salmonids following 96 h exposures, we expect that fewer deaths of juveniles will occur in many of the freshwater habitats exposed to carbaryl. Fish reproduction (based on a single fathead minnow study) would be affected by concentrations in the off-channel habitat. We also note minimal data exist on effects to fish growth: only a single study with fathead minnows. We expect that carbaryl will impair swimming of salmonids, kill salmonid prey, and in certain circumstances kill salmonids when exposed for sufficient durations. The effect concentrations shown in the figure do not account for the potential enhanced toxicity of carbaryl to salmonids or their prey items in aquatic habitats where other AChE inhibitors are present. We also note that pH is a major factor in carbaryl's persistence in aquatic habitats. At pHs above 8, carbaryl breaks down fairly rapidly (half-life of 24 h) while at pHs less than 8 carbaryl is much more resistant to hydrolysis (half-life of 1- 30 d for pH of 7.9 – 5.7). The pH of natural surface waters commonly ranges from 7 to 9, thus pH is an important consideration when evaluating toxicity of carbaryl.

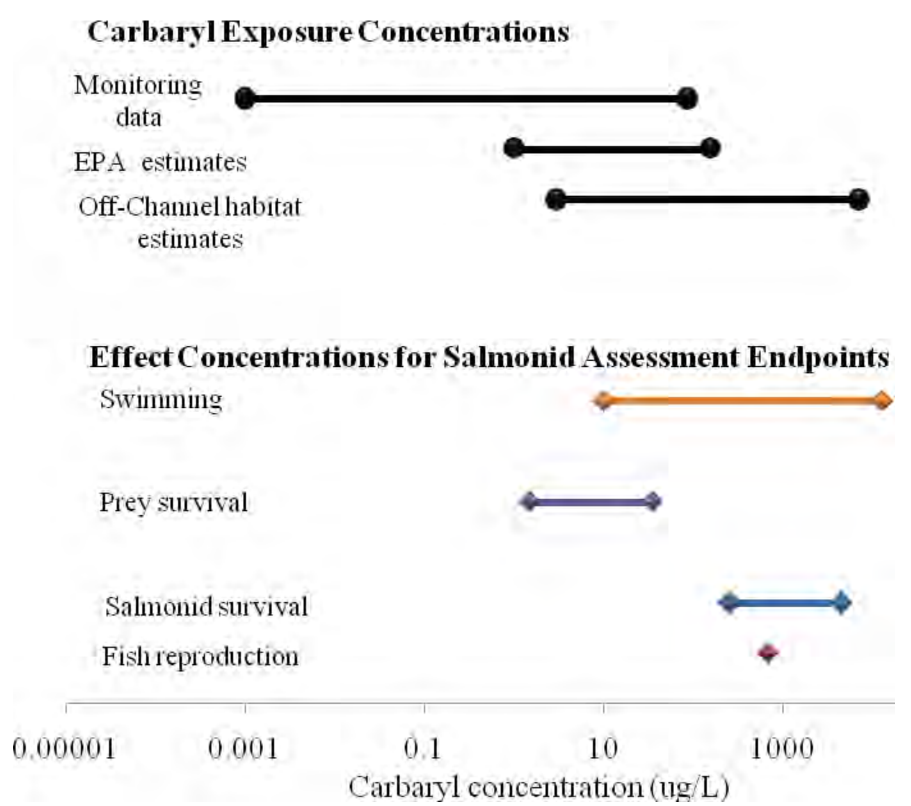

Figure 39. Carbaryl exposure concentrations and salmonid assessment endpoints' effect concentrations in µg/L

Carbofuran

Concentration ranges overlap with most of the assessment endpoint ranges, indicating that adverse effects are expected in salmonids if exposed for a sufficient duration (Figure 40). Salmonid prey items appear just as sensitive to carbofuran as to carbarylalthough significant variation among prey species was observed in reported acute survival EC/LC 50s. Fish reproduction, swimming and salmonid olfactory-mediated behaviors are encompassed or exceeded by all three exposure ranges. The fish growth value (56 µg/L LOEC) was exceeded by the off-channel habitat concentrations. Salmonid survival was the least sensitive response reviewed. As with carbaryl, concentrations of carbofuran occurring in the off-channel habitats that NMFS modeled will kill juvenile salmonids. Furthermore, given the LC50 values for salmonids following 96 h exposures, we expect that fewer deaths of juveniles will occur in many of the freshwater habitats exposed to carbofuran. As with carbaryl, we note that pH is a major factor in the degree of toxicity of carbofuran. At pHs above 9, carbofuran breaks down fairly rapidly (half-life of 0.8-15 h) while at pH 7 reported half-lives range from 2- 28 d. Therefore, pH is an important consideration when evaluating toxicity of carbofuran in the aquatic environment. We discuss these effects and other considerations in more detail under the risk hypotheses.

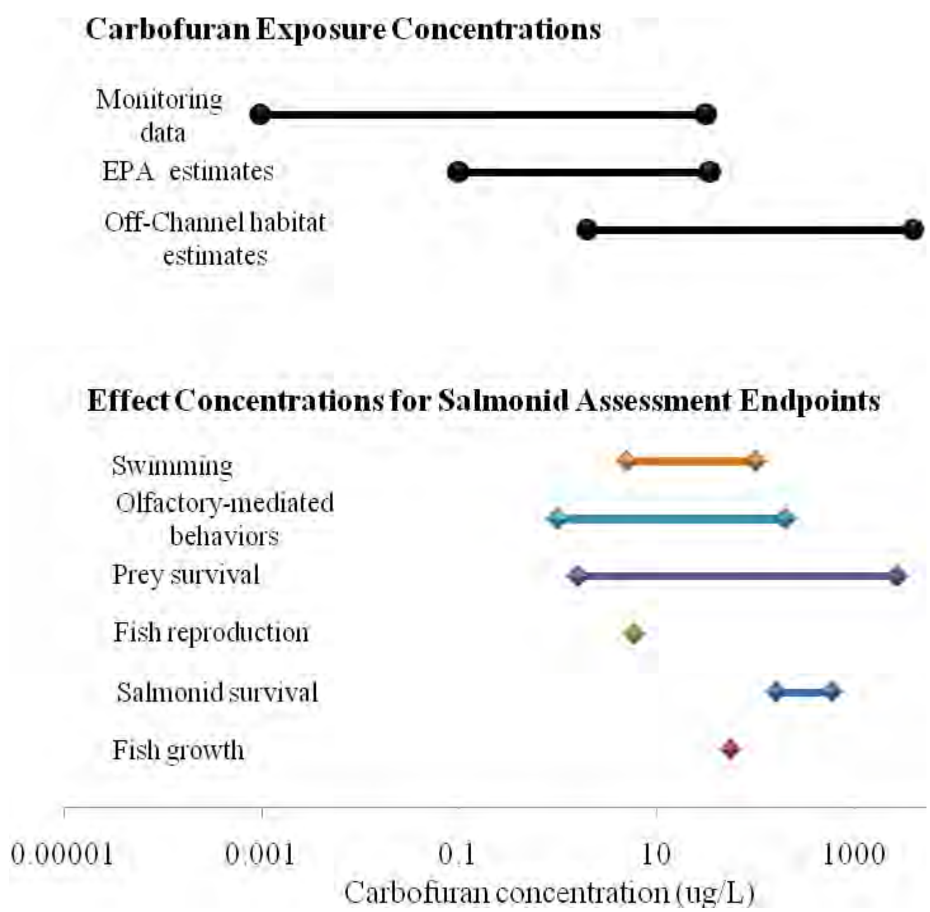

Figure 40. Carbofuran exposure concentrations and salmonid assessment endpoints' effect concentrations in µg/L

Methomyl

We expect a portion of fish to be exposed to upper end concentrations which would overlap with most of the effect thresholds. (Figure 41). As with carbaryl and carbofuran, prey survival was the most sensitive endpoint assessed. More sensitive prey, with EC50s at the lower end of the distribution, are expected to die based on all three exposure estimate ranges, with the highest rates of mortality based on EPA estimates and NMFS' off-channel habitat estimates. The magnitude of reduction in prey abundance will depend on which taxa are present in the exposed water body and the actual concentrations and exposure durations. If reductions in prey abundance, especially of the smaller organisms, coincide with fry feeding for the first time following yolk sac absorption, starvation is likely. We discuss this in greater detail in the risk hypotheses below. The lower value of the fish reproduction endpoint, 31 µg/L, overlapped with both EPA's and NMFS' off-channel habitat estimates, and the highest value in the monitoring data was greater. A notable data gap is the absence of information on methomyl's toxicity to olfactory-mediated behaviors. Although we found no studies that evaluated swimming, we expect swimming impairment at concentrations less than LC50s because swimming is impaired with other AChE inhibitors and AChE inhibition correlates to impaired swimming (Little and Finger 1990; Little, Archeski et al. 1990).

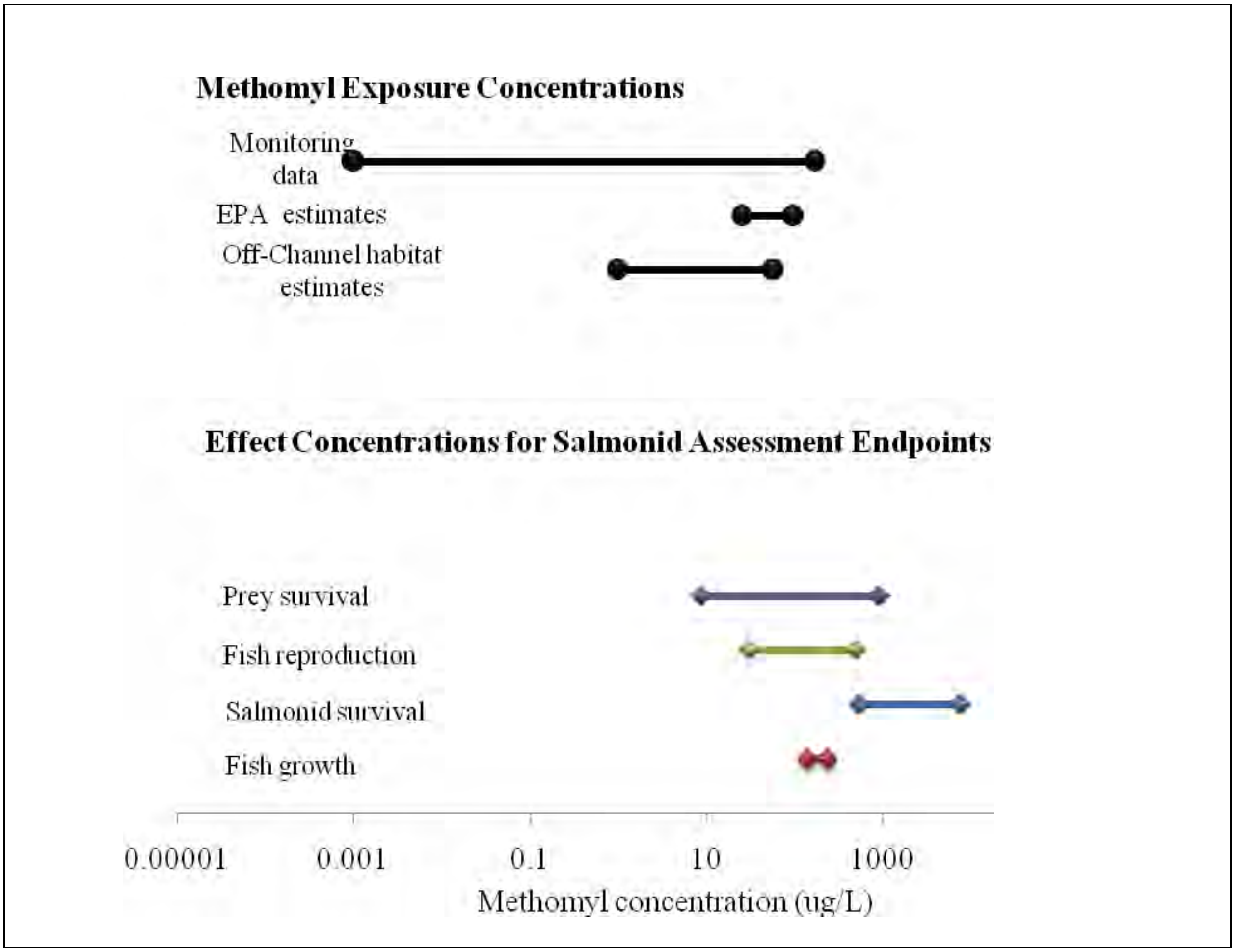

Figure 41. Methomyl exposure concentrations and salmonid assessment endpoints' effect concentrations in µg/L

Relationship of pesticide use to effects in the field

Schulz (2004) reviewed 45 field and *in situ* studies published in peer-reviewed journals (from 1982-2003) that evaluated relationships between insecticide contamination and biological effects in freshwater aquatic ecosystems and included invertebrates and fishes. For each study, the authors classified the relationship of exposure to effect in one of four categories: no relation, assumed relation, likely relation, and clear relation based on the cited authors' judgment of their own results. A relationship was classified as clear only if the exposure was quantified and the effects were linked to exposure temporally and spatially. The review concluded that "about 15 of the 42 studies revealed a clear relationship between quantified, non-experimental exposure and observed effects *in situ*, on abundance [aquatic invertebrate], drift, community structure, or dynamics" (Schulz 2004). Although the top three insecticides most frequently detected at levels expected to result in toxicity were chlorpyrifos (OP), azinphos-methyl (OP), and endosulfan, the *N*-methyl carbamates carbaryl, carbofuran, oxymyl, and fenobucarb all showed clear, likely or assumed relationships between exposure and effect (Schulz 2004). No studies involving methomyl were evaluated. It should be noted that these studies were not designed to establish effect thresholds and in our review, are not sufficient to define thresholds. However, the data do provide information on concentrations of insecticides known to cause biological and ecological effects under field conditions.

Schulz (2004) noted that for all of the studies "that seem to establish a clear link between exposure and effect, the pesticide concentrations measured in the field were not high enough to support an explanation of the observed effects simply based on [laboratory bioassays] acute toxicity." Some authors have suggested differences in field measured exposure and actual organismal exposure (aquatic, sediment, dietary; environmental variables that affect exposure) as a reason for higher mortalities *in situ* than predicted by laboratory toxicity data. Schulz concluded that on the basis of present knowledge, it cannot be determined whether the measured concentration in the field regularly underestimates the actual exposure or if a general difference between the field and laboratory reactions of aquatic invertebrates is responsible; both are reasonable assertions. The review by Schulz shows a body of evidence that natural aquatic ecosystems can be adversely affected by AChE inhibitors including carbaryl and carbofuran. We expect a similar relationship for methomyl because it shares the same mechanism of action as carbaryl and carbofuran (AChE inhibition) and concentrations that lead to death of aquatic invertbrates are similar to carbaryl and carbofuran.

We reviewed multiple studies that showed direct, adverse effects to salmonid prey species following exposures to *N*-methyl carbamates (Table 67). These included reduction in individual aquatic species, changes in community abundance, richness, and diversity with salmonid prey items.

Table 67. . Examples of published field studies designed to establish a relationship between *N*-methyl carbamate contamination of aquatic habitats and agricultural practices (adapted from Table 2 in Schulz 2004).

Source	Concentration µg/L	Duration	Endpoint	Species	Relationship of exposure and effect
Carbaryl					
Aerial application	0.1-85.1	24	aquatic invertebrate Drift	Various invertebrates	Likely
Carbofuran					
	0.05-26.8	Few hours	mortality	Amphipod (*Gammarus pulex*)	Clear

Field studies in ESA-listed salmonid habitats with other AChE inhibitors

A group of field studies evaluated macroinvertebrate community responses in the orchard-dominated Hood River Basin, Oregon and correlated results with chlorpyrifos and azinphos-methyl use and detections (Grange 2002, St. Aubin 2004, Van der Linde 2005). Hood River Basin contains several listed anadromous salmonids, including lower Columbia River steelhead. The goals of the studies were to determine whether in-stream OPs affected steelhead AChE activity and changed the aquatic macroinvertebrate community. An additional objective addressed how changes in macroinvertebrate community might affect salmonid growth. A suite of reference and orchard-dominated sampling sites within the Hood River Basin were sampled pre and post the two primary application seasons, spring (chlorpyrifos) and summer (azinphos-methyl). Significant differences in macroinvertebrate community assemblages were found between upstream reference sites and downstream agricultural sites (St. Aubin 2004), similar to the results described in a California stream (Hall, Killen et al. 2006). However, no significant differences were found at each individual site, before and after summer spraying (St. Aubin 2004). Therefore, the second Hood River study investigated the spring spray events as well as the summer spray events to determine seasonal effects (Van der Linde 2005). Sharp declines in species abundance between reference sites and downstream sites during the spring-spray period correlated to chlorpyrifos applications and subsequent aquatic detections (one site over an 8 d period showed chlorpyrifos ranging from 0.032 -0.183 µg/L). There were more pollutant tolerant taxa and less intolerant taxa at the agricultural sites (Van der Linde 2005). Collector-gatherer species, many of which are salmonid prey items, declined rapidly at agricultural sites compared to abundances at the reference sites following applications. Interestingly, reductions in biodiversity in 2001 agricultural sites compared to reference sites was not seen in 2002 (Van der Linde 2005). The authors commented that diversity metrics do not always behave consistently or predictably in response to environmental stress. More than two years of data are likely needed to more sufficiently address community variability at this site.

Two sets of field experiments directly investigated juvenile steelhead (hatchery- reared) AChE activity from caged-fish studies in an agricultural basin in Hood River Basin, OR (Grange 2002; St. Aubin 2004). The studies analyzed water samples for chlorpyrifos, azinphos-methyl, and malathion before, during, and after orchard spray periods. One of the studies also monitored the aquatic invertebrate community's response (discussed later under prey effects) in conjunction with the AChE inhibition (St. Aubin 2004). Steelhead from reference sites had statistically significantly greater AChE activity than steelhead from orchard-dominated areas. The reductions in AChE activity corresponded to the application seasons and detections of chlorpyrifos and azinphos-methyl insecticides.

The data from one study indicated that OP-insecticides inhibited AChE activity in steelhead held in cages in the Hood River Basin and this inhibition correlated to chlorpyrifos and azinphos-methyl detections and to a lesser degree with malathion detections (Grange 2002). None of the pesticides were detected at reference sites and both chlorpyrifos (range in maxima of (0.077-0.196 μg/L) and azinphos-methyl were frequently detected at orchard stream and river sites. AChE activity was inhibited up to 21% in smolts, and 33% in juveniles relative to reference locations. Temperature was a confounding factor as lower temperatures showed lower AChE activity while higher temperatures showed higher AChE activity at reference sites. The authors normalized data to temperature and found a greater number of statistically significant reductions in AChE in steelhead. Study results show that steelhead in these systems exposed to OP insecticides lose AChE activity (up to 33%) which, depending on the percentage of inhibition, can manifest into fitness level consequences (Grange 2002; St. Aubin 2004).

The field studies conducted in Hood River Basin, Oregon show that salmonids' AChE activity was reduced in streams near orchards during chlorpyrifos and azinphos-methyl applications.

Additionally, the macroinvertebrate communities in these systems were compromised to such an extent that there was a reduction in salmonid prey abundance. These findings highlight the importance of characterizing the presence of AChE insecticides, *N*-methyl carbamates and OPs, within aquatic habitats because the compounds share a mode of action and adverse biological responses to mixtures containing multiple AChE insecticides are likely cumulative. These data can serve as a surrogate for expected effects of the aquatic community when concentrations of carbaryl, carbofuran, and methomyl attain effect thresholds.

Field studies in ESA-listed salmonid habitats: Willapa Bay and Grays Harbor Washington

History of Carbaryl use in Washington estuaries

Willapa Bay and Grays Harbor support large commercial oyster-producing areas (approximately 600 acres in Willapa and 200 acres in Grays Harbor) and are located north of the Columbia River along the southwestern coast of Washington. The Pacific oyster (*Crassostrea gigas*), introduced from Japan in the 1920s, is the principal species cultivated in those estuaries (Feldman, Armstrong et al. 2000). Two species of endemic shrimp, ghost shrimp (*Neotrypaea californiensis*) and mud shrimp (*Upogebia pugettensis*), are abundant in the oyster growing areas of these estuaries and create burrows in the sediment that are unfavorable for optimal oyster cultivation. The Washington State oyster industry has used aerial applications of carbaryl to kill the burrowing shrimp in oyster beds since the early 1960s. EPA authorizes the use of carbaryl on oyster beds in Washington through a SLN label, or Section 24(c) (EPA Reg. No. 264-316). Carbaryl is applied directly to intertidal areas when they are exposed at low tide, primarily by helicopter (Dumbauld, Brooks et al. 2001). This use has been controversial and there have been numerous studies dating back several decades that evaluated the ecological impact of carbaryl use in these two estuaries (Feldman, Armstrong et al. 2000). Although the EPA label does not limit applications to Willapa Bay and Grays Harbor, historically these have been the only areas treated with carbaryl. Additionally, Washington State requires applicators to obtain NPDES permits for applications to aquatic habitats.

ESA-Listed Pacific salmonids that occur in Willapa Bay and Grays Harbor

Juvenile and adult Chinook salmon from ESA-listed populations are expected to be present in Willapa Bay and Grays Harbor at the time of carbaryl applications. It is expected that juvenile

Chinook salmon from the Lower Columbia River use these estuaries during the period when carbaryl is applied based on their behaviors (Personal communication Eduardo Casillas NMFS 2/18/2009, Anne Shaffer WDFW 3/04/2009, and Thom Hooper NMFS, 2/12/2009). Specifically, their use of dendritic channels to access inundated mud flats suggests juvenile Chinook salmon will be exposed to peak carbaryl concentrations in the water column and also from feeding on live and dead contaminated prey. Also, recent information (six years of coastline surveys) shows that juvenile Columbia River Chinook salmon "bounce" north along the Washington coastline and south along the Oregon coastline in the nearshore surf zones (Personal communication Eduardo Casillas NMFS 2/18/2009) and have been found occupying nearshore habitats as far North as the Strait of Juan de Fuca (Shaffer, Crain et al. submitted). In one study, juvenile Chinook salmon captured along the coast Washington State were genetically analyzed to determine natal river origins. Of the fish collected for genetic analysis, 45% of juvenile Chinook salmon collected in Crescent Bay, 75% of those in Freshwater Bay, and 60% of those in Pysht Bay were from ESA-listed stocks of the Columbia River (Shaffer, Crain et al. submitted).

DNA analysis revealed that adult salmonids harvested in Willapa Bay in July and August, when carbaryl is typically applied, include several ESA-listed species: Lower Columbia River Chinook salmon, California Coastal Chinook salmon, Puget Sound Chinook salmon, and Snake River Fall-run Chinook salmon (Kassler and Marshall 2004). Given this information we also expect these same species of Chinook salmon will be present in Grays Harbor as Grays Harbor is located just north of Willapa Bay. Further, we expect adults of these stocks may be exposed to carbaryl given that they occupy the estuary during the application periods and oyster plots occur throughout the Bay. It is possible that chum salmon from the Columbia River also occur in the Willipa Bay and Grays Harbor estuaries. However, we are unaware of documentation of their occurrence at those locations, and unlike juvenile Chinook salmon from the Columbia River, juvenile chum have not been found in recent sampling efforts of nearshore surf zones. We therefore conclude that Columbia River ESA-listed chum are unlikely to be present in Willapa Bay or Grays Harbor.

Measured Concentrations in Willapa Bay and Grays Harbor following Carbaryl Applications

Numerous monitoring studies have been conducted in coordination with applications of carbaryl to control burrowing shrimp in commercial oyster beds in Washington State. The results of the studies are quite variable, as are the study designs and objectives. Several investigations have documented water column concentrations exceeding several mg/L (which is substantially higher than the µg/L concentrations observed in freshwater aquatic habitats) following the first flood tide post-application (Hurlburt 1986; Creekman and Hurlburt 1987; Tufts 1989; Tufts 1990). Water column concentrations measured at Willapa Bay in 1984 detected a mean concentration of 10.6 mg/L upon initial flooding of the treated area (Hurlburt 1986). Concentrations were monitored at varying depths for several hours following flooding. The concentrations decreased rapidly but were detectable in the low µg/L range throughout sampling. In 1985 samples were collected along transects from treated areas following the direction of the tide to monitor the dispersal of carbaryl (Creekman and Hurlburt 1987). The data indicate that carbaryl is transported off the treated area by the tide. Average concentrations measured above the treated area ranged from 2.4 – 5.5 mg/L. The highest average concentration (7.9 mg/L) was collected 510 ft from the treated area. Average concentrations at the most distant sampling point (650 ft from initial treated area) were 2.5 mg/L. In 1996, further study evaluated carbaryl movement off the treated spray area (Tufts 1989). Samples were collected along a transect corresponding to the direction of the incoming tide. The peak concentration measured was 27.8 mg/L. Over one treated area, the concentration of carbaryl decreased from 13.2 mg/L to 9.3 mg/L as the water depth increased from 1.5 to 10 inches. The concentration further decreased to 600 µg/L when the water rose to 16 inches. The monitoring indicated concentrations of carbaryl decreased with increasing distances along the transect, but concentrations greater than 1 mg/L were detected in several instances at distances several hundred feet from treated plots. Carbaryl was found in the water column at the detection limit (0.1 mg/L) as far as 1,725 ft from the treated area.

In 1987, surface water monitoring was conducted in Willapa Bay to determine “the dilution pattern of carbaryl washed from sprayed oyster beds, and to determine carbaryl concentrations in shallow pools and streams where marine fish might be found” (Tufts 1990). Carbaryl concentrations in the mg/L range were frequently detected with the incoming tide. Peak concentration for one sample site was 18.8 mg/L upon initial flooding. The concentrations decreased to 0.2 mg/L when covered by 18 inches of water. Concentrations of carbaryl were variable among sites. For example, at one of the sampling sites carbaryl was not detected until the water was 7 inches deep. A peak concentration of 8 mg/L was detected at this site when the water depth was 11 inches. This sample station was located 300 ft from a treated area. Maximum carbaryl concentrations detected at other sites were 17.4, 7.8, and 4 mg/L with samples collected at depths of 6, 4, and 13 inches from the bottom. Carbaryl's primary toxic degradate, 1-naphthol, was detected at concentrations as high as 1.4 mg/L. Average concentrations of carbaryl detected in tide pools and small streams ranged from 3.6 to 11.2 mg/L. The 11.2 mg/L detection was associated with application of 5 lbs carbaryl/acre. An average concentration of 7.8 mg/L in tide pools and streams was associated with an application rate of 4 lbs carbaryl/acre, approximately half the maximum rate allowed. Similarly, a more recent study found a peak of 820 µg/L in the water column 50 ft from the application site following a typical treatment application at the maximum rate of 8 lbs/acre (n=3) (Weisskopf and Felsot 1998).

Several studies demonstrate that carbaryl dissipates fairly rapidly from over the treatment site due to degradation, metabolism, dilution, and off-site transport (Hurlburt 1986; Creekman and Hurlburt 1987; Tufts 1989; Tufts 1990; Weisskopf and Felsot 1998). In 2006 and 2007, samples collected at the mouth of channels adjacent to treated oyster beds in Willapa Bay had maximum concentrations of 29.1 µg/L after 6 h (high tide), 38 µg/L after 12 h (low tide), and 21.1 µg/L after 24 h (low tide) (Major, Grue et al. 2005).

The NPDES permit for Willapa Bay and Grays Harbor requires annual monitoring of water column concentrations in treated areas. It specifies an acute effluent limit of 3 µg/L and a chronic limit of 0.06 µg/L. However, those data are of highly questionable value because the monitoring plan specifies that monitoring is suspended for the first 48 h following application to assess the acute effluent, and further suspends monitoring for 30 days from the last application to assess the chronic limit. We expect that most of the carbaryl will be degraded and transported to other locations by the time carbaryl monitoring is initiated.

Carbaryl sprayed on mud flats can be transported substantial distances at concentrations that may have ecological impacts. Researchers found that close to 100% of Dungeness crabs were killed up to 100 m off the carbaryl application area (Doty, Armstrong et al. 1990). Levels decrease to below 1 mg/L when transported more than 200m or more. Washington DOE reports that carbaryl concentrations in the "potential effects threshold range" of 0.1-0.7 μg/L have been detected at locations several miles from oyster beds soon after large areas were treated (Johnson 2001).

Measured Concentrations in Invertebrates

Samples of crustaceans from carbaryl treated areas in 1984 showed high tissue levels (Table 68). A single ghost shrimp and Dungeness crab were analyzed and contained concentrations of 24.9 and 41.9 mg/kg (ppm), respectively (Hurlburt 1986). Analysis of dead shrimp in 1985 following treatment at 7.5 lbs carbaryl/acre revealed average concentrations of approximately 4.5 mg/kg. When left on the treated oyster beds the concentrations declined to approximately 10% of the initial concentration 24 h post-treatment, then remained relatively stable at the 48, 72, and 96 h sampling events. Concentrations in shrimp and annelid worms were investigated following the 1985 applications and associated with different application rates (Tufts 1989). Average concentrations in shrimp ranged from 5.3 to 13.8 mg/kg. The concentration in worms ranged from 57-58.6 mg/kg. The 10 lb application rate is less relevant as the current label restricts applications to 8 lbs a.i./acre. Concentrations in other, more relevant salmonid prey items were apparently not assessed although we assume they may be comparable for other organisms that occupy similar habitat.

Table 68. Carbaryl concentrations in aquatic organisms on treated sites (mg/kg).

Year	Dungeness Crab	Burrowing Shrimp	Annelid Worms
1984	41.9	24.9 (rate not reported)	-
1985	-	4.5 (7.5 lb a.i./acre)	-
1986	-	13.8 (10 lb a.i./acre) 8.7 (7.5 lb a.i./acre) 5.3 (5 lb a.i./acre)	75.5 (5 lb a.i./acre) 57 (7.5 lb a.i./acre) 58.6 (5 lb a.i./acre)

Ecological Effects

There have been concerns that carbaryl applications in Willapa Bay and Grays Harbor may have adverse effects to the commercial Dungeness crab fishery (Feldman, Armstrong et al. 2000). Consequently, Washington Department of Fisheries conducted several surveys to estimate the number of crabs killed (Hurlburt 1986; Creekman and Hurlburt 1987; Tufts 1989; Tufts 1990).

Transect surveys conducted on treatment beds the day following applications indicate large numbers of crabs are killed by the applications (Table 69). Other acute mortalities to non-target organisms were generally not reported. However, transect surveys were conducted during 1986 and 1987 due to concerns over potential impacts to fish given staff observations from the Washington Department of Fisheries reported that fish mortality was "routinely" noted, but not quantitatively assessed following applications of carbaryl (Tufts 1989). Mortalities were characterized as small fish apparently trapped in shallow pools during low tide and directly exposed during carbaryl treatments. The surveys indicate several thousand fish were killed each year from carbaryl applications to approximately 400 acres at rates of ≤ 7.5 lbs a.i./acre (Table 70). Currently, the NPDES permit allow for treatment of up to 600 acres. The current 24 (c) label does not specify acreage restrictions and allows for applications of carbaryl up to 8 lbs a.i./acre.

Table 69. Estimated dungeness crab mortalities resulting from carbaryl applications in Willapa Bay and Grays Harbor, Washington

Year	Maximum Application Rates*	Total Acres Treated**	Dungeness Crab Killed
1984	≤ 10 lbs a.i. / acre	490	38,410
1985	≤ 7.5 lbs a.i./acre	391	59,933
1986	≤ 7.5 lbs a.i./acre	398	16,286
1987	≤ 7.5 lbs a.i./acre	434	44,053

* Current 24(c) label allows for application of up to 8 lbs a.i./acre
** Current NPDES permit allows for a total of 600 hundred acres to be treated in Willapa Bay and Grays Harbor, WA. The Label places no restrictions on acres or geographic restrictions for use of carbaryl on oyster beds in Washington.

Given that the fish that reside in standing water on mud flats are likely to be the most vulnerable to carbaryl exposure, staff from the Washington Department of Fisheries also estimated the available marine fish habitat that existed on exposed mudflats of treated areas. They characterized water that was at least 2 inches in depth, a depth that is adequate for salmon fry, as marine habitat. This habitat comprised substantial portions of the treated areas. In 1986, surveys indicate 135 acres of the 398 of the treated area (approximately 34%) were marine fish habitat (Tufts 1989). In 1987, 67 of the 434 (15%) acres were characterized by Washington Department of Fisheries as marine fish habitat. The current NPDES permit specifies that there be a 200 ft buffer zone for sloughs and channels when carbaryl is applied by helicopter. A 50 ft buffer is required for those aquatic habitats when carbaryl is applied by hand sprayer. AgDrift estimates for aerial application at an application rate of 8 lbs a.i./acre with a 200 ft buffer indicate the average initial concentration from drift to a body of water that is 2 inches deep (~ 5 cm) and 10 m wide would be 771 µg/L. It seems highly unlikely that helicopter applications would be able to successfully avoid direct overspray of some of the marine fish habitats on the exposed mudflats. Additionally, the 24(C) labels do not contain buffer zone restrictions. A direct overspray of 2 inches of water would result in an average initial concentration of approximately 18 mg/L, a concentration comparable to measured concentrations associated with initial tidal inundation of treated mud flats (see surface water detections above). Acute exposure to these concentrations are expected to kill a portion of exposed salmonids in a matter of hours *i.e.*, 24 h LC50s for carbaryl range from 948-8,000 µg/L, n= 60 (Mayer and Ellersieck 1986), and significantly reduce AChE activity that will lead to myriad sublethal effects such as impaired swimming.

Table 70. Estimated fish mortalities resulting from carbaryl applications in Willapa Bay and Grays Harbor, Washington

Year	Maximum Application Rates*	Total Acres Treated**	Fish Killed
1986	≤ 7.5 lbs a.i./acre	398	14, 954
1987	≤ 7.5 lbs a.i./acre	434	8,041

* Current 24(c) label allows for application of up to 8 lbs a.i./acre
** Current NPDES permit allows for a total of 600 hundred acres to be treated in Willapa Bay and Grays Harbor, WA. The Label places no restrictions on acres or geographic restrictions for use of carbaryl on Oyster beds in Washington.

Salmonids were not reported in the sampling data, which was taken over two seasons. The authors report that the fish kills were extremely variable and unpredictable. The fish mortality reported for transect surveys included only four species in 1986: saddleback gunnel, threespine stickleback, staghorn sculpin, and arrow goby. In 1987, mortalities included English sole, sand sole, kelp greenlings, shiner perch, saddleback gunnels, staghorn sculpin, and arrow goby.

Response of benthic community

Samples were collected in 1985 to assess impacts of carbaryl on the intertidal invertebrate community (Tufts 1990). Treated area showed decreases in numbers of infaunal benthic crustaceans at 15 and 30 d post-treatment while a concurrent increase in crustacean abundance was observed on control plots (Table 71). Abundance of benthic crustaceans was low on both control and treated plots 5 months after application in December. The authors suggest the data may represent a seasonal decline in benthic crustaceans.

Table 71. Total number of benthic crustaceans (tanaids, cumaceans, amphipods, copepods, and ostracods; adapted from Tufts 1990).

Date	Sampling	Control Plot (change from pretreatment)	Treated Plot (change from pretreatment)
June 30	Pretreatment	75	33
July 16	15 days post-treatment	96 (+28%)	2 (-94%)
August 01	30 days post-treatment	136(+181%)	4 (-82%)
December 10	161 days post-treatment	2	6

Field incidents reported in EPA incident database

NMFS reviewed reported incidents of fish deaths from field observations throughout the U.S. because the information reflects real world scenarios of pesticide applications and corresponding death of freshwater fish. We recognize that much of the information is not described in sufficient detail to attribute an incident to a label-permitted use leading to the death of fish, or to make conclusions regarding the frequency of fish kills that may be associated with the use of pesticides. NMFS uses the information as a component to evaluate a line of evidence- whether or not fish kills have been observed from labeled uses of the three pesticide products. EPA categorizes incidents in the database into one of five levels of certainty: highly probable, probable, possible, unlikely, or unrelated. The certainty level indicates the likelihood that a particular pesticide caused the observed effects. EPA uses the following definitions to classify fish kill incidents:

- Highly probable (4): Pesticide was confirmed as the cause through residue analysis or other reliable evidence, or the circumstances of the incident along with knowledge of the pesticides toxicity or history of previous incidents give strong support that this pesticide was the cause.

- Probable (3): Circumstances of the incident and properties of the pesticide indicate that this pesticide was the cause, but confirming evidence is lacking.

- Possible (2): The pesticide possibly could have caused the incident, but there are possible explanations that are at least as plausible. Often used when organisms were exposed to more than one pesticide.
- Unlikely (1): Evidence exists that a stressor other than exposure to this pesticide caused the incident, but that evidence is not conclusive.

- Unrelated (0): Conclusive evidence exists that a stressor other than exposure to the given pesticide caused the incident.

NMFS reviewed several incident reports provided by EPA from OPP's incident database. This database is populated with reports received by EPA from registrants that are defined as reportable under FIFRA 6(a)(2) and includes other information received from registrants and other sources.

Relatively few incidents involving lethality to fish were documented in the database, considering the length of time these products have been registered and their toxicity profiles. EPA provided the following table summarizing known mortality incidents (Table 72). NMFS is uncertain as to how representative this database is of known fish kills and the level of coordination that has occurred with various state and federal agencies that investigate these incidents. The mortality events discussed above for carbaryl applications in Willapa Bay and Grays Harbor were not recorded in the database. Several of the incidents provided by EPA are discussed in more detail below.

Table 72. EPA summary of field incident data with carbaryl, carbofuran, and methomyl; highlighted incidents (*) are discussed.

Carbaryl		Carbofuran		Methomyl	
Incident ID	**Certainty**	**Incident ID**	**Certainty**	**Incident ID**	**Certainty**
B0000-246-01*	Probable	I000165-052	Possible	B0000-216-19	Unlikely
B0000-501-92*	Probable	I000599-008*	Probable	B0000-501-36	Unlikely
I000799-003	Possible	I001605-003	Probable	I000108-001*	Probable
I000910-001	Unlikely	I005416-001	Highly Probable[1]		
I007720-020	Probable	I012265-001	Possible		
I015419-644	Possible	I012265-003	Probable		
I018436-001	Possible	I012265-004	Possible		
I013436-001*		I013436-001*	Possible	I013436-001*	Possible

[1]Waterfowl mortality incident in flooded rice

Incident I013436-001: A report of a large fish kill on October 16, 2001, was investigated by the California Department of Fish and Game. The kill involved several thousand fish. Although salmonids were not identified, it occurred in the San Joaquin River, which is used by listed salmonids. EPA characterized this event as "possible" regarding its association with carbaryl, carbofuran, and methomyl. Methomyl was detected in fish gills and several other pesticides were detected in the water including azinphos-methyl, another cholinesterase inhibitor, and several chlorinated hydrocarbons (chlordane, DDT, dieldrin, methoxychlor, mirex, and others). It is likely that the kill was in part due to mixture toxicity, particularly with the combinations of carbamates and OP insecticides.

Incident I000599-008: A report by the California Department of Fish and Game indicates approximately 3,000 fish, 4,000 crayfish and frogs, 200 birds, and at least 5,000 invertebrates were killed on North Temple Creek in San Joaquin County, California. The adjacent vineyard had been treated with Furadan 4F (carbofuran). Chemical analysis confirmed carbofuran in the gizzards of starling up to 9.8 mg/kg, crayfish collected on site had 0.6 mg/kg, the stomach contents of great egrets, which were primarily crayfish, contained 0.3 mg/kg, and surface water had detections up to 7,800 µg/L. The California Department of Fish and Game concluded that the preponderance of evidence indicated carbofuran was the cause of death. EPA characterized the cause of death as probable.

Incident B0000-246-01: According to a 1975 report by EPA, 22,000 catfish were killed in 7 miles of stream at an agricultural site in Oklahoma. Minimal information on the incident was reported. Chemical analysis was not conducted but EPA classified this association with carbaryl as probable. Other a.i.s were identified as possibly contributing to the kill were toxaphene, methyl parathion, and endrin.

Incident B0000-501-92: EPA concluded that a fish kill in a New Jersey pond was caused by drift of carbaryl following the application of Sevin in 2000. This incident was characterized as probable. No chemical analysis was conducted.

FMC reported several incidents called in to their emergency telephone number in the second quarter of 1992. One incident reported a small number of dead bluegill observed in a drain pond adjacent to corn and tobacco fields. The caller indicated Furadan 15G (carbofuran), Counter (chlorpyrifos), and Temik (aldicarb) had been applied to the crops. Chlorpyrifos is an AChE inhibiting organophosphate insecticide. Aldicarb and carbaryl are both cholinesterase inhibiting *N*-methyl carbamates. Dead fish were observed following heavy rains. No analysis was done on the fish to determine the cause of death. EPA characterized carbofuran as a “possible” cause of the incident. It seems likely that the three cholinesterase inhibiting insecticides may have resulted in a cumulative exposure that was sufficient to cause the fish kill due to mixture toxicity.

Incident I000108-001: DuPont reported a fish kill in 1992 in southern Georgia that occurred following an application of Lannate LV (methomyl). One hundred and twenty-five fish were found dead in a pond located 50 to 75 yards from a large field of sweet corn. The 200 acre field was treated with Lannate LV five times between June 1 and June 16. Additionally, Lorsban insecticide (chlorpyrifos) was applied four times during the same interval including co-applications of four fertilizer treatments. On the day of the fish kill (June 16), applications of both Lannate and Lorsban were made. Lannate LV was applied at rates of 1.5 pints/acre and Lorsban was applied at a rate of 1 pint/acre. During the first 16 days of June a total of 10.52 inches of rain were received. The State of Georgia took water samples on June 17. Analysis indicated 136 µg/L of methomyl in the pond. Chlorpyrifos concentrations were not reported and it is not known if it was an analyte. Water temperatures were relatively high (reported to be about 90ºF) but dissolved oxygen was reported to be “normal (7.2-10)”. DuPont suggested the high temperatures, fertilizers, and suspended solids could be stressful to fish and low oxygen content at night might have played a part in the deaths. The species involved in the incident (carp and bluegill) were warm water fishes. EPA characterized methomyl as a “probable” cause of the fish kill. Given the reported rapid dissipation of methomyl it is likely that the concentrations measured do not represent peak concentrations that occurred in the pond. It seems likely that chlorpyrifos may have also been a contributing factor in this incident.

Mixture Analysis of Carbaryl, Carbofuran, and Methomyl

As noted earlier, pesticides most often occur in the aquatic environment as mixtures. In our review and synthesis of the available exposure and response information, we find the three *N*-methyl carbamates carbaryl, carbofuran, and methomyl share the same mechanism of toxic action (AChE inhibition) and are expected to co-occur in salmonid habitats. Therefore, we employ a simple mixture analysis derived from empirical data with Pacific salmonids to predict potential effects to individual salmonid’s AChE activity and their survival from short-term exposures. The analysis is predicated on the toxic potencies of the three insecticides added together to predict the resulting cumulative effect to AChE activity and mortality.

Mixture toxicity is typically described by three general responses: antagonistic, additive, or synergistic. Antagonism and synergism are where the toxic response is not predicted by the individual potencies of the pesticides found in the mixture. Antagonistic effects of a mixture lead to less than expected toxicity on the organismal endpoint. Mechanistically, the pesticides are likely interacting with one another to reduce the toxic potency of individual pesticides. Synergistic effects of a mixture lead to a greater than expected effect on the organismal endpoint and the pesticides within the mixture enhance the toxicity of one another. The third general type of mixture toxicity and the one most frequently reported is additivity (known also as dose-addition or concentration-addition). This type of response is defined by adding the individual potencies of pesticides together to predict the effect on the biological endpoint. Additivity has been demonstrated for many pesticide classes as well as other organic compounds such as PAHs, PCBs, and dioxins.

Additive toxicity of chemicals that share a mode or mechanism of toxic action is well established in the scientific literature, and as a result has been informing regulatory decisions for more than a decade. In 1996, the National Academy of Sciences recommended a dose-additive approach to assessing risks to human infants and children from pesticide exposure. EPA currently assesses human risk of pesticide mixtures for pesticides that share a common mechanism of toxic action *e.g.*, *N*-methyl carbamates [such as carbaryl, carbofuran, methomyl], organophosphorus insecticides, chloroacetanilide and triazine herbicides, as mandated by FQPA. The analysis EPA conducts is predicated on additive toxicity and applies dose-addition to set tolerance limits of pesticide residues on food. For example, the toxic potencies of the *N*-methyl carbamates are added together to determine pesticide tolerance limits for edible crops. Although additive toxicity is evaluated when determining risk to humans, EPA OPP has yet to apply a similar approach to address cumulative toxicity of pesticides that share a common mode or mechanism of action in the evaluation of terrestrial and aquatic species. That said, the use of dose-addition for mixtures containing acetylcholinesterase-inhibiting pesticides is well established and has been extended to protection of aquatic life (Belden, Gilliom et al. 2007).

Dose-addition assumes the cumulative toxicity of the mixture can be predicted from the sum of the individual toxic potencies of each component of the mixture. Within the past decade, government regulatory bodies started to use dose-addition models to predict toxicity for those chemicals that share a common mode of action. In California, the CVRWQB used dose-addition models (based on the toxic-unit approach) to develop TMDLs for the OP insecticides diazinon and malathion. NMFS Biological Opinions have also recognized the environmental reality of co-occurring pesticides in species' aquatic habitats and applied additive toxicity models to predict potential responses of salmonids (NMFS 2004; NMFS 2005; NMFS 2005; NMFS 2005; NMFS 2008).

In salmon, dose-additive inhibition of brain AChE activity by mixtures of OPs and carbamates was demonstrated *in vitro* (Scholz, Truelove et al. 2006). More recently, it has been found that salmonid responses to OP and carbamate mixtures vary *in vivo*; some interactions were synergistic, rather than just additive (Laetz, Baldwin et al. 2009). We used the dose-addition method to predict responses by applying the modeling exposure estimates and measured concentrations of carbaryl, carbofuran, and methomyl presented in the *Exposure Analysis*. Effects of the three carbamates individually and in combination on AChE inhibition Figure 42 (A) and survival (B), are shown below. Based on additivity, the mixture is expected to be more toxic than the individual carbamates for both endpoints. Due to the steep slopes of the two dose-response curves, and especially the mortality slope, small changes in concentrations elicit large changes in observed toxicity. The exposure values represent concentrations from EPA PRZM-EXAMs 60 d average modeling estimates for surface waters (carbaryl: 4 aerial applications at 5 lbs a.i./acre, citrus in FL; carbofuran: 1 ground application at 2 lbs a.i./acre, artichokes in CA; methomyl: 3 aerial applications at 1.8 lbs a.i./acre, peaches). We recognize that this approach is likely to under-predict toxicity for mixtures that produce synergistic rather than additive responses (Laetz, Baldwin et al. 2009).

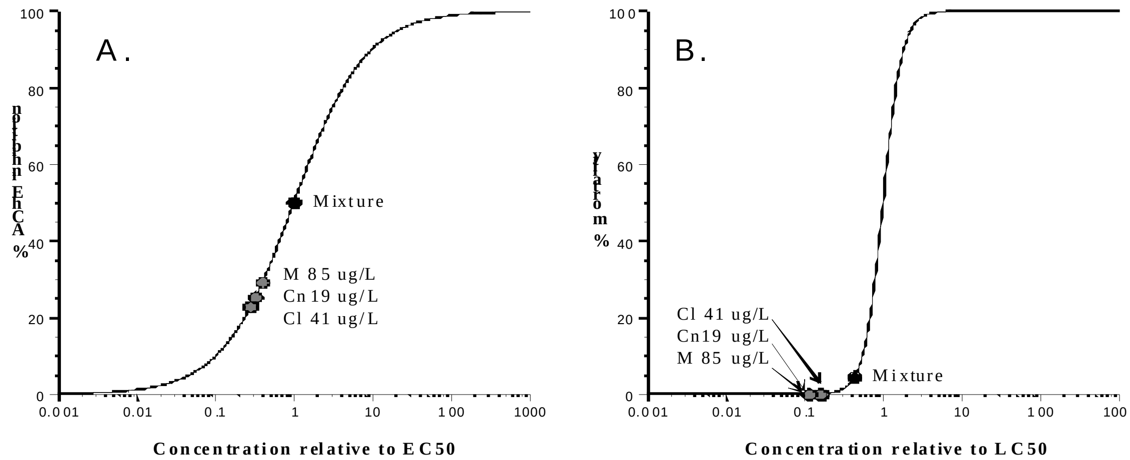

Figure 42. Percent AChE inhibition (A.) and percent mortality (B.) for salmonids expected from exposure to carbaryl (Cl), carbofuran (Cn), and methomyl (M) as separate constituents and as mixtures (Cl 41 µg/L, Cn 19 µg/L, and M 85 µg/L)[4].

We used a variety of exposure estimates and monitoring data to evaluate responses to different mixtures of carbaryl, carbofuran, and methomyl (Table 73). The predicted additive responses from these mixtures ranged from 16-74% inhibition of AChE and 0.01-74% mortality. The predicted additive response to AChE inhibition is likely to result in increased behavioral consequences to salmonids. What is not captured in these responses is the likelihood of exposure to the various mixture concentrations. The PRZM-EXAMS values were estimates selected from EPA simulations of western crops. The scenarios were representative of use rates and numbers of applications on current product labels. Additionally, we used 60 d, time-weighted averages of exposure rather than predicted peak concentrations as exposure to multiple pesticides would be expected to occur more frequently over chronic durations. This may underestimate effects as responses assumed 96 h exposure. Site specific considerations will also have an influence on the frequency of exposure.

[4] EPA's default pesticide slope was used for acute mortality (3.63 or probit slope of 4.5) [EPA 2004]. The slope used for AChE inhibition was based on pooling data from five cholinesterase-inhibiting insecticides; including carbofuran, carbaryl, chlorpyrifos, diazinon, and malathion Laetz, C. A., D. H. Baldwin, et al. (2009). "The synergistic toxicity of pesticide mixtures: Implications for risk assessment and the conservation of endangered Pacific salmon." Environmental Health Perspectives **117**: 348-353..

Table 73. Predicted AChE inhibition and mortality from estimated and measured exposure to carbaryl, carbofuran, and methomyl.

	Concentration (µg/L)	% AChE Inhibition	% Mortality
Modeling: PRZM-EXAMS 60-day averages[1] (from Table 49)			
Carbaryl	12	8.34	0.00
Carbofuran	19	25.39	0.04
Methomyl	81	28.33	0.09
Additive response		**44.30**	**1.38**
Modeling: GENEEC 90-day averages (from Table 50)			
Sweet Corn			
Carbaryl	229	60.67	42.10
Carbofuran	53	47.67	1.63
Methomyl	49	19.61	0.01
Additive response		**72.24**	**73.62**
Potatoes			
Carbaryl	101	41.28	3.59
Carbofuran	106	63.93	17.02
Methomyl	34	14.66	0.00
Additive response		**71.95**	**59.44**
Citrus			
Carbaryl	280	65.25	60.14
Methomyl	21	9.76	0.00
Additive response		**66.25**	**62.97**
Monitoring: NAWQA maxima in 4 states (from Table 54)			
Carbaryl	33.5	19.59	0.07
Carbofuran	32.2	36.02	0.27
Methomyl	0.82	0.48	0.00
Additive response		**44.22**	**1.79**
Monitoring: California CDPR database maxima (from Table 55)			
Carbaryl	8.4	6.07	0.00
Carbofuran	5.2	8.93	0.00
Methomyl	5.4	2.85	0.00
Additive response		**15.58**	**0.01**
Monitoring: Washington EIM database maxima (from Table 56)			
Carbaryl	10	7.09	0.00
Carbofuran	2.3	4.29	0.00
Methomyl	0.17	0.11	0.00
Additive response		**10.60**	**0.00**

[1]PRZM-EXAMS estimates for carbaryl in California peaches (2 applications at 7 lb a.i./acre), carbofuran in California artichokes (1 application at 2 lb a.i./acre), and methomyl in lettuce (10 applications at 0.9 lb a.i./acre). Although current labeling may not be consistent for these uses (peaches, artichokes, and lettuce), the use rate and number of applications are consistent with other labeled uses within the action area.

The GENEEC estimates are 90 d, time-weighted averages that were based on labeled uses of the three compounds in sweet corn, potatoes, and citrus. We found no restrictions that would prevent co-application or sequential applications of carbaryl, carbofuran, and methomyl. The application rates assumed were consistent with current labels and generally representative of use rates authorized for many crop and non-crop uses. An exception was the assumed carbaryl use rate in citrus (12 lb a.i./acre), which is substantially higher than the maximum use rate approved for most crops.

The NAWQA, CDPR, and EIM monitoring values represent the maximum concentrations found in the respective databases. The values cited were all measured within the four states and the vast majority is from waters that contain or drain to listed salmonid habitats. Most of the detections in these, and other monitoring studies that did not target specific applications of the three chemicals occurred at or below the µg/L level. We expect that exposure at these levels will be common in drainages where the three products are used extensively.

Evaluation of Risk Hypotheses: Individual Salmonids

In this phase of our analysis we examine the weight of evidence from the scientific and commercial data to determine whether it supports or refutes a given risk hypothesis. We also highlight general uncertainties and data gaps associated with the data. In some instances there may be no information related to a given hypothesis. If the evidence supports the hypothesis we determine whether it warrants an assessment either at the population-level, or affects PCEs to such a degree, to warrant an analysis on the potential to reduce the conservation value of designated critical habitat.

1. Exposure to carbaryl, carbofuran, and methomyl is sufficient to:

A. Kill salmonids from direct, acute exposure.

A large body of laboratory toxicity data indicates that anadromous salmonids die following short-term (< 96 h) exposure to the three insecticides at levels above 200 µg/L. We expect concentrations of carbaryl, carbofuran, and methomyl in salmonid off-channel habitats will reach lethal levels based on exposure concentrations derived from monitoring data, EPA's modeling estimates, and NMFS modeling estimates (See *Exposure Analysis*). The youngest swimming salmonids appear to be the most likely to die from short-term, acutely toxic exposures in these habitats. It is less likely that adults would be killed by acute concentrations in most freshwater aquatic habitats compared to juveniles. However, if adults are present in smaller off-channel habitats during aerial applications or severe runoff events death is possible, particularly from carbaryl and carbofuran. The available monitoring data, if representative of salmonid habitats, indicated that concentrations rarely achieve LC50 values for the three compounds in freshwaters. However, it is unlikely that peak concentrations are reflected in the monitoring data, and given the acutely toxic nature of carbamates, a brief exposure would be sufficient to cause effects. As described in the *Exposure Analysis*, monitoring data are limited when compared to the range of habitats used by salmonids. Few data were found that targeted applications and subsequent concentrations in edge of field habitats which typically show much higher concentrations than weekly, monthly, or seasonal monitoring efforts. Although we found no information on egg survival following acute exposures, we do not expect death of eggs from these insecticides as entry into the eggs via the water column is unlikely. Further support for acute lethality to fish is found in field incidences of death attributed to carbaryl, carbofuran, and methomyl that EPA ranked as "probable". We located several incidents showing death of freshwater fish following exposures to *N*-methyl carbamates. We expect juveniles of listed salmonids to be at the highest risk of death when in small freshwater off-channel and edge habitats, and secondarily in estuarine habitats where carbaryl is applied directly to mudflats. In conclusion, the available information on measured and expected concentrations of the three insecticides supports this hypothesis. We translate the fitness level consequences of reduced survival from mortality of juvenile salmonids to potential population-level consequences using population models (see population modeling section below).

B. Reduce salmonid survival through impacts to growth.

Fish growth is reduced following long-term exposures to carbofuran and methomyl in fathead minnows and rainbow trout, respectively. EPA reported on a single test that measured growth effects to fathead minnows following 30 and 60 d exposure to carbaryl at 0, 8, 17, 62, 210, and 680 µg/L (Carlson 1971). No statistically significant effects on growth were reported in this study. It is difficult to extrapolate from this one study with a warm water species to potential growth effects to ESA-listed salmonids especially given that salmonids appear substantially more sensitive to carbaryl's acute toxicity than fathead minnows (96 h LC50s for salmonids range from 250-4,500µg/L carbaryl and for fathead minnows 7,700-14,600 µg/L). Additionally, 20% and 50% inhibition of AChE in salmonids occurs at concentrations as low as 23 and 185 µg/L, respectively (Labenia, Baldwin et al. 2007) and this inhibition was found to affect swimming behavior (Labenia, Baldwin et al. 2007). Reduced growth occurred at 56 µg/L for carbofuran and 142 µg/L for methomyl. Only one test result was reported for each pesticide. We did not identify any studies that provided a quantitative relationship between growth and fish survival in the field or lab. However, there is abundant literature that shows salmonids that are smaller in size have reduced first year survival (Appendix 1). Additionally, exposure to sublethal concentrations of other AChE inhibitors (chlorpyrifos and diazinon) for acute durations does cause reduced feeding success, which likely results in impacts to growth (Scholz, Truelove et al. 2000; Sandahl, Baldwin et al. 2005). We expect that juvenile fish exposed to carbaryl, carbofuran, and methomyl during their freshwater residency will feed less successfully, resulting in lower growth rates and reduced sizes. Exposure concentrations will likely vary temporally and spatially for salmonids depending on life history, pesticide use, and environmental conditions. The available information support that growth is likely reduced where salmonids are exposed to more than 56 µg/L carbofuran and to concentrations below LC50 for carbaryl and methomyl, although the exact concentrations that may cause growth reduction are currently unknown. The weight of evidence supports the conclusion that fitness level consequences from reduced size are likely to occur in individual salmonids exposed to the three *N*-methyl carbamates. Therefore, we address the potential for population-level repercussions due to reduced growth using species-specific population models.

C. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey

We address several lines of evidence to determine the likelihood of reduced salmonid growth from impacts to aquatic invertebrate prey. The first line of evidence we evaluated is whether salmonid prey items are sensitive to acute and chronic exposures from expected concentrations of the three carbamates. These primarily involved evaluating laboratory experimental results that reported on incidences of death or sublethal effects. We located a total of 10, 4, and 14 survival estimates (24, 48, 72, and 96 h EC/LC50s) for carbaryl, carbofuran, and methomyl, respectively. Based on an evaluation of the assessment endpoints, we found a robust body of exposure and toxicity data that indicated salmonid aquatic prey are highly sensitive and affected by expected exposures to each of the insecticides as well as from mixtures containing the three insecticides. We expect death and a variety of sublethal effects to salmonid prey items.

The second line of evidence is whether field level reductions in aquatic invertebrates correlate to *N*-methyl carbamate insecticide use and/or concentrations in salmonid habitats. We found several examples supporting this line of evidence. The available laboratory and field data show reductions in aquatic invertebrate taxa and reductions in invertebrate abundances following applications of carbofuran and carbaryl (Table 67). Furthermore, field studies explicitly investigated whether real world applications and subsequent pesticide drift and/or runoff into aquatic environments affected receiving water aquatic communities. One compelling study measured pre- and post-application aquatic community responses from field applications of carbaryl at 0.75 and 1 lb/acre (these rates are at the lower end of approved maximum application rates, up to 12 lbs/acre) (Courtemanch and Gibbs 1980). Applications correlated to substantial drift of salmonid prey type aquatic invertebrates, of which most were dead or dying. More striking was that population reductions of plecoptera (stoneflies) persisted for sixty days (Courtemanch and Gibbs 1980). Runoff from fields treated with carbofuran at 2.68 lbs a.i./ acre contained concentrations as high as 264 μg/L in the field ditches that drained to a stream where carbofuran was measured at 27 μg/L (Matthiessen, Shearan et al. 1995). Caged amphipods present in the stream all died during peak runoff following a rain event one month after the carbofuran application. This indicated that carbofuran can persist in soil and mobilize by rain into subsequent runoff and ultimately elicit toxic responses from stream invertebrates (Matthiessen, Shearan et al. 1995). Although we found no field studies with methomyl, we expect that it would similarly reduce aquatic invertebrate populations at higher concentrations than carbaryl and carbofuran based on its toxicity profile.

The third line of evidence we evaluated was whether salmonids showed reduced growth in areas of low prey availability, particularly those areas coinciding with use of carbaryl, carbofuran, and methomyl. An evaluation of this line is complicated by multiple factors affecting habitat quality *i.e.*, water quantity, quality, riparian zone condition, etc., which in turn affects prey items and salmonids. We were unable to locate information attributing reduced growth in salmonids to specific insecticide exposures that reduced prey, as most studies focused on measuring direct effects on salmonids or direct effects on invertebrates (see review by Schulz 2004). However, there are multiple field experiments and studies that demonstrate reduced fish growth resulting from reduced prey availability (Brazner and Kline 1990; Metcalfe, Fraser et al. 1999; Baxter, Fresh et al. 2007) or document fish growth rates below maximal potential growth rates when prey are limited (Dineen, Harrison et al. 2007).

One study, in particular, tested the hypothesis that single applications of the OP insecticide chlorpyrifos (0.5, 5, 20 μg/L) to outdoor ponds (littoral enclosures) would reduce the abundance of invertebrates and cause diet changes that would result in reduced growth rates of juvenile fish (Brazner and Kline 1990). The results are direct, empirical evidence that support this hypothesis. Growth rates of fathead minnow larvae were reduced significantly in all chlorpyrifos-containing treatments due to reduction in prey abundance. At 15 d post-treatment, the reductions in growth rate compared to control fish were the most pronounced and coincided with the greatest reductions in invertebrates. Stomach contents of minnows were identified throughout the experiment. By day 7 mean numbers of protozoans, chironomids, rotifers, cladocerans, mean total number of prey being eaten per fish, and mean species richness were greater in unexposed treatments compared to some of the other treatments. On day 15, most of the differences were more pronounced. The results strongly support the conclusion that foraging opportunities were better in untreated enclosures and unexposed larvae grew significantly more compared to chlorpyrifos-treated enclosures. Furthermore, the reductions in prey items in diets mirrored the reduction in prey items in the enclosures. We did not find any study results with *N*-methyl carbamates, but we make the inference that concentrations of the three *N*-methyl carbamates that are sufficient to reduce aquatic prey would also lead to reduced fish growth. This further supports the hypothesis that reduction in prey abundances translates to reductions in subsequent ration as well as individual growth. The authors concluded that "low levels of contaminants that induce slower growth in young-of-the-year fish through food chain effects or other means may eventually reduce the survival and recruitment of these fish."

Collectively, the lines of evidence strongly support the overall hypothesis. Thus, we carry reduced prey impacts to the next level of analysis (*i.e.,* the population-level). We conducted population modeling exercises based on reduced abundances of salmonid prey, presented in the next section, (*Effects to Salmonid Populations from the Proposed Action*).

D. Impair swimming which leads to reduced growth (via reductions in feeding), delayed and interrupted migration patterns, survival (via reduced predator avoidance), and reproduction (reduced spawning success).

Swimming is a critical function for anadromous salmonids. The primary line of evidence for this hypothesis is impaired swimming behaviors following exposure to carbaryl, carbofuran, and methomyl. A secondary line of evidence for this hypothesis is studies showing swimming behavior modification following exposure to other AChE-inhibiting chemicals such as the OPs, as we anticipate the results are similar in nature. Several studies regarding the effects of carbaryl on the swimming related behaviors were reviewed, with effects on predator avoidance, schooling, and feeding behaviors occurring at carbaryl concentrations of ~100-1,000 µg/L in laboratory studies (Weis and Weis 1974; Arunachalam, Jeyalakshmi et al. 1980; Little, Archeski et al. 1990; Carlson, Bradbury et al. 1998; Labenia, Baldwin et al. 2007). Only one study was available for carbofuran showing effects to social behaviors at 5 µg/L (Saglio, Trijasse et al. 1996), and no studies were located for methomyl. Concentrations at which effects were noted for carbaryl and carbofuran are within the ranges of concentrations estimated by the various modeling methods, especially for the off-channel habitat locations. We anticipate similar effects will occur with methomyl because it has the same mechanism of action as carbaryl and carbofuran, inhibition of AChE (which is correlated to swimming impacts). However, we expect it will take greater concentrations of methomyl compared to carbaryl and carbofuran to affect swimming. We expect that these levels are likely from concentrations resulting from aerial drift into off-channel habitats. This is because concentrations of methomyl necessary for AChE inhibition and acute lethality (LC50) are greater than carbaryl and carbofuran concentrations necessary to inhibit AChE and kill fish. We also discussed compelling evidence that OPs impair salmonid swimming behaviors and also show associated reductions in AChE activity.

The most sensitive swimming endpoints are those associated with swimming activity compared to those that measure swimming capacity (Little and Finger 1990; Little, Archeski et al. 1990). The ecological consequences to salmonids from aberrant swimming behaviors are implied primarily through the impairment of feeding, translating to reduced growth. Impaired swimming behavior correlated with both AChE inhibition and increased mortality from predation (Labenia, Baldwin et al. 2007). Although NMFS was unable to locate results from field or laboratory experiments for the other remaining endpoints of this hypothesis, we conclude that swimming behaviors are affected by the three insecticides. Adverse effects to swimming-associated behaviors are directly attributed to AChE inhibition, leading to potential reductions in an individual's fitness (*i.e.*, growth, migration, survival, and reproduction). We therefore translate impaired swimming to potential impacts on salmonid populations. Based on concentrations generated in modeling, NMFS believes concentrations of carbaryl and carbofuran, applied in accordance with current labels, are sufficient to impair swimming behavior of salmonids in some environments. As we located no studies that evaluated the effects of methomyl on swimming behaviors in fish, we do not know the exact levels sufficient to impair swimming except to say that levels well below reported LC50s can impair swimming. Swimming-mediated behaviors are frequently impaired at 0.3 – 5.0% of reported fish LC50s, and that 75% of reported adverse effects to swimming occurred at concentrations lower than reported LC50s (Little and Finger 1990). Taken together this information supports methomyl affecting swimming behaviors below reported LC50s. We expect methomyl at hundreds of μg/L (expected from currently approved applications) to impair swimming.

E. Reduce olfactory-mediated behaviors resulting in consequences to survival, migration, and reproduction.

In the Opinion regarding effects of chlorpyrifos, diazinon, and malathion on listed salmonids (NMFS 2008), sufficient data were available to conclude that olfactory-mediated behaviors were affected by those pesticides. Fewer data were available to assess the effects of carbaryl, carbofuran, and methomyl on these endpoints. Evidence is unclear as to mode of action for insecticide-mediated olfactory impairment. Thus, conclusions drawn based on data for OPs may not necessarily be applicable to *N*-methyl carbamates. Data currently available are conflicting, with the one study available for carbaryl showing no apparent effect on olfaction on juvenile cutthroat trout at concentrations of up to 500 μg/L. Three studies regarding olfactory effects of carbofuran were located and reviewed, and these indicated olfactory effects in several species of fish (including two salmonids) at concentrations ranging from 1-10 μg/L. These concentrations are within both the ranges estimated by modeling, and the ranges measured in all monitoring data sets. No data were available for methomyl. While data are not conclusive, based on what is known, and giving the benefit of the doubt to the species, NMFS believes it is reasonable to assume that these types of effects will occur from carbofuran exposures. Therefore, we discuss the potential implications at the population-level.

F. Exposure to mixtures of carbaryl, carbofuran, and methomyl can act in combination to increase adverse effects to salmonids and salmonid habitat.

The exposure and toxicity information we compiled, reviewed, and analyzed support the risk hypothesis, although less mixture data were available for the carbamates addressed in this Opinion than the OPs addressed in the previous Opinion (NMFS 2008). Evidence of additive and synergistic effects on survival and AChE inhibition in salmonids were identified. Multiple, independent study results supported additive toxicity from measured AChE inhibition. We therefore conducted an analysis of potential mixtures on the levels of AChE inhibition and the potential for a greater reduction in survival predicated on simple additivity. The analysis showed that both survival and AChE inhibition of individuals is likely affected to a greater degree than from exposure to a single chemical alone. We also expect that assessment endpoints influenced by AChE inhibition are likely affected to a greater degree when in the presence of more than one of the three insecticides. Considerable uncertainty arises as to the level of impairment caused by mixtures for some endpoints, as dose responses have not been characterized for some pesticide combinations. We conclude that this hypothesis is supported by the available information and we assess the potential for population-level consequences below.

G. Exposure to other stressors of the action including degradates, adjuvants, tank mixtures, and other active and other ingredients in pesticide products containing carbaryl, carbofuran, and methomyl cause adverse effects to salmonids and their habitat.

Fifteen of the carbamate formulations contain other a.i.s, including malathion (OP insecticide), bifenthrin (pyrethroid insecticide), rotenone (insecticide/piscicide), metaldehyde (mulloscicide), captan (phthalidimide fungicide), and cupric sulfate (toxic to algae, aquatic invertebrates, and fish). No data regarding the joint toxicity of these multi-a.i. products were presented in the BEs or located in open literature. However, given that these pesticides are also toxic to aquatic life it is reasonable to assume the salmonids and/or their prey items will exhibit a greater toxic response when exposed to multiple a.i.s than to a single one. We are unable to estimate the magnitude of such a response, and other than for malathion, are not currently able to evaluate whether the response would fall into an additive, synergistic, or antagonistic category. Based on mixture data discussed elsewhere in this Opinion, we assume additivity for carbaryl, carbofuran, and methomyl.

Some toxicity data were available in the BEs for formulations not containing additional a.i.s. Based on that data, no specific conclusions can be drawn as to whether formulations are, in general, more or less toxic than the technical grade a.i. However, it should be noted that adjuvants which increase either uptake (such as penetrants or surfactants that make physiological membranes more permeable) or length of exposure (attractants, or emulsions that increase time in the water column) are likely to increase toxicity of the a.i.

Only a limited amount of data were available to evaluate toxicity of the degradates of carbaryl, carbofuran, and methomyl. Monitoring data indicate that 1-napthol, a degradate of carbaryl, and 3-hydoxycarbofuran, a degradate of carbofuran, occur in measurable quantities in the environment. Based on the monitoring data, 1-napthol co-occurs with carbaryl in both water column and sediment. Toxicity information presented in the BE indicates that 1-napthol is approximately an order of magnitude less toxic than parent carbaryl for aquatic invertebrates. No data were given in the BE regarding the toxicity of 1-napthol to fish. We did locate one open literature study evaluating the effects of 1-napthol on fish, which indicated it was 2-5 times more toxic than the parent carbaryl (Shea and Berry 1983). No toxicity data were located for 3-hydroxycarbofuran or 3-ketocarbofuran, but based on structural similarities, the toxicity is likely similar to the parent carbofuran. Based on fate data presented in the BEs, 3-hydroxycarbofuran and 3-ketocarbofuran appear to occur in soil rather than in the water column. Both are shown as being non-detectable to representing 2-5% of applied parent material. Together, they could represent 6-7% of applied, assuming they were measured in the same test. Neither the BE nor the Science Chapter specify if that is the case.

Overall, while NMFS cannot quantify the increased toxicity of exposure to other stressors of the action such as additional a.i.s, degradates, and adjuvants, the existing body of information indicates these compounds are likely to cause and/or exacerbate adverse effects on salmonids and their habitat caused by carbaryl, carbofuran, and methomyl.

We did not receive a complete list of the currently registered formulations containing carbaryl, carbofuran, and methomyl from EPA. Thus, we cannot make any definitive conclusions for every stressor of the action. However, we did evaluate the exposure and response to a commonly used surfactant/adjuvant mixed with, or found in pesticide formulations. We reasoned if the data support adverse effects from this one of more than 4,000 substances, then other unidentified inert ingredients could also be toxic and may pose a significant risk to salmonids and their habitat. We selected NP polyethoxylates and NP because of their widespread use in pesticide formulations and abundance of information regarding environmental concentrations and adverse effects to salmonids and their prey. The data indicated that these surfactants can kill outright, disrupt endocrine systems, particularly reproductive physiology, and bioaccumulate in benthic invertebrates from expected concentrations in the environment (Arsenault, Fairchild et al. 2004; Madsen, Skovbolling et al. 2004; Jardine, MacLatchy et al. 2005; Luo, Ban et al. 2005; McCormick, O'Dea et al. 2005; Segner 2005; Hutchinson, Ankley et al. 2006; Lerner, Bjornsson et al. 2007; Lerner, Bjornsson et al. 2007). Importantly, we found studies that linked Atlantic salmon population crashes in Canada to use of NP in insecticide formulations. However, the BEs did not provide any information as to the prevalence of this material in formulations of the three OP insecticides that pertain to this consultation. We did receive confirmation from the technical registrant of methomyl that no methomyl containing formulation they make contains NP. Significant uncertainty surrounds the number and type of compounds, as well as the toxicity of these other materials used in pesticide formulations. As a result, we must caveat our conclusions regarding population-level responses with the uncertainty that the actual risk posed to listed salmonids and their habitat is likely greater when all ingredients are taken into account.

H. Exposure to other pesticides present in the action area can act in combination with carbaryl, carbofuran, and methomyl to increase effects to salmonids and their habitat.

The available toxicity and exposure data support the hypothesis. Other carbamates and OPs found in the action area likely result in additive or synergistic effects to exposed salmonids and aquatic invertebrates. The magnitude of effects will depend on the duration and concentrations of exposure.

I. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

We found no consistent correlation with elevated temperature and toxicity of the three *N*-methyl carbamates. However, salmonids are coldwater species, and exposure to elevated temperatures increases physiological stress, thus making them more susceptible to other stressors. Additionally, other AChE inhibitors such as OPs are more toxic to salmonids and aquatic invertebrates exposed to elevated temperatures, so where elevated temperatures co-occur with OPs and carbamates the effect to aquatic life is likely greater than at lower temperatures. Many salmonid populations reside in watersheds which have been listed by the four western states as impaired due to temperature exceedances. We therefore discuss qualitatively temperature impacts on salmonids population responses to the stressors of the action.

J. Exposure to specific pH ranges can affect the toxicity of the stressors of the action.

Some data indicate acute toxicity of carbaryl and carbofuran increases as pH increased based on the available freshwater fish assays (Mayer and Ellersieck 1986). Other data indicated that toxicity was reduced in pH greater than 9 due to rapid hydrolysis *i.e.*, a half-life of 30 minutes in pond mesocosms. For methomyl, pH seems to have less of an influence on hydrolysis rates. Within the Pacific Northwest and California pH varies seasonally and typically may range from 6 – >9. We expect that salmonids exposed to both pH ranges at or near physiological tolerance limits and the three insecticides concurrently may die at relatively lower concentrations compared to salmonids exposed in laboratory assays. We therefore discuss qualitatively pH impacts on salmonids population responses to the stressors of the action.

Effects to Salmonid Populations from the Proposed Action

Here we translate individual fitness consequences to potential population-level effects using both quantitative and qualitative methods. We quantitatively translate reduced survival of juveniles based on four-day exposures to four populations of salmonids including ocean-type Chinook, stream-type Chinook, coho, and sockeye salmon. We employ a life history population model that incorporates changes in first-year juvenile survival rates and then translates them into predicted changes in the modeled population’s intrinsic rate of growth, *i.e.*, lambda (Appendix 1). We discuss the percent change in lambda in the context of expected concentrations of the three insecticides in salmonid habitats. We focus on the concentrations at which a significant departure occurs from the unexposed population and compare them to expected environmental concentrations described in the *Exposure Analysis*. We also discuss in general terms the likelihood of exposure to the range of pesticide concentrations that occur in salmonid habitats.

In a second modeling exercise, we translate reductions in growth of juvenile salmon from AChE inhibition and from reduced prey abundances to potential population impacts using individual-based growth and life history population models (Appendix 1). These two endpoints (AChE inhibition and reduced prey abundance) are combined in the model to evaluate population-level effects due to reductions in first year survival of juveniles (Appendix 1). Similar to the survival models, percent change in lambda is the output. We discuss the significance of population changes in the context of departures from normal variability of the unexposed population and expected environmental concentrations. We conclude this section on population-level effects with a discussion of population-level responses to other affected salmonid endpoints that were not modeled. These include effects from other stressors of the proposed action, mixture effects, and effects to behaviors from impaired olfaction and AChE inhibition such as swimming behaviors.

Salmonid Population Models

We selected four generalized life history strategies to model (Appendix 1). We ran general life history matrix models for coho salmon (*Oncorhynchus kisutch*), sockeye salmon (*O. nerka*) and ocean-type and stream-type Chinook salmon (*O. tshawytscha*). We did not construct a steelhead (*O. mykiss*) life history model due to the lack of demographic information. Chum salmon (*O. keta*) were omitted from the growth model exercise because they migrate to marine systems soon after emerging from the gravel and the model assesses early life stage growth effects over a minimum of 140 d in freshwater systems. The basic salmonid life history we modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. For specific information on how we constructed the models see Appendix 1.

Effects to salmonid populations from death of sub-yearling juveniles

An acute toxicity model was constructed that estimated the population-level impacts of subyearling juvenile (referred to as juveniles within this section) mortality resulting from exposure to concentrations of carbaryl, carbofuran, and methomyl. These models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from an annual 4 d exposure of all juveniles in the population to carbaryl, carbofuran, and methomyl from single exposures. Death of juveniles was implemented as a change in first-year survival rate for each of the salmon life-history strategies modeled. We also addressed mixture toxicity of the three insecticides in the model using exposure concentrations derived from EPA modeling estimates and from NMFS' modeling estimates of off-channel habitats. Model output is displayed in Tables 74-77.

The percent changes in lambdas increased as concentrations of the three carbamates increased. Increases in direct mortality during the first year of life produced large impacts on the population growth rates for all the life history strategies. Model results for stream-type Chinook salmon showed significant impacts at lower concentrations than the other modeled populations. This result is primarily due to the size of the standard deviation of the unexposed population. Percent changes in lambda were deemed significant if they were outside of one standard deviation from the unexposed population. The relative sensitivity of the life history models producing the greatest to the least changes in population growth rate for equivalent impact on survival rates was coho salmon, ocean-type Chinook salmon, stream-type Chinook salmon, and sockeye salmon. We note that the choice of LC50 is an important driver for these results. Therefore, an LC50 above or below the ones used here will result in a different dose-response. We selected the lowest reported salmonid LC50 from the available information to ensure that risk is not underestimated. However, if the actual environmental 96 h LC50 is lower, then the model will under-predict mortality. If the actual environmental acute LC50 is higher, then the model will over-predict mortality.

These results indicate that exposure of salmonid populations to carbaryl, carbofuran, and methomyl for four days at the reported LC50s would have severe consequences to the population's growth rate. If exposure occurred every year for each new cohort, population growth rate would be reduced and recovery efforts would be slowed. For those natural populations with current lambdas of less than one, risk of extinction would increase substantially, especially if several successive generations were exposed. For each of the combinations of species and insecticide, we denoted the relative concentration at which the percent change in lambda is deemed significantly different from the unexposed populations *e.g.,* a 9.1% change in lambda is estimated at 190 μg/L carbaryl for ocean-type Chinook salmon. These population effect thresholds assume exposure to all the juveniles in the population.

Carbaryl's thresholds for the four species ranged from 129-190 μg/L, carbofuran's from 89-126 μg/L, and methomyl's from 287-431 μg/L. These results can be compared to expected concentrations shown in Figure 39, Figure 40, and Figure 41. All of the thresholds overlap with expected concentrations for the three insecticides. The overlap primarily occurs with NMFS modeled estimates for off-channel habitats.

When we compare the population threshold concentrations to expected levels in salmonid habitats described in the exposure section, it is likely that some individuals within a population will be exposed during their freshwater juvenile life stage, particularly those juveniles exposed while using off-channel habitats. Additionally, four day averages from GENEEC runs, but not from PRZM-EXAMS, exceed these threshold concentrations. It is uncertain how appropriate EPA's model estimates are for salmonid habitats, but taken at face value juveniles exposed to GENEEC-derived concentrations would die. Population-level effects are expected from these exposures if the majority of the individuals that comprise the populations are exposed during their freshwater residency. The likelihood of population effects from death of juveniles increases for those populations that spend longer periods in freshwaters such as steelhead, stream-type Chinook, and coho salmon. We also expect additional acute mortalities from juveniles that are exposed to 24(c) carbaryl applications in Washington estuaries, although we do not know how many individuals are exposed each year. For those populations with lambdas greater than one, reductions in lambda from death of juveniles can also lead to consequences to abundance and productivity. Consequently, attainment of recovery goals would take longer to achieve for populations with reduced lambdas. Many of the populations that are categorized as core populations or are important to individual strata, have lambdas just above one and are essential to survival and recovery goals. Slight changes in lambda, even as small as 3-4%, would result in reduced abundances and/or increased time to meet population recovery goals.

Table 74. Modeled output for Ocean-type Chinook salmon exposed to 4 d exposures of carbaryl, carbofuran, and methomyl reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population.

Carbaryl	0 µg/L	50 µg/L	100 µg/L	200 µg/L	250 µg/L	350 µg/L	500 µg/L	750 µg/L
% dead	0	0	3	31	50	77	93	98
Lambda (STD)	1.09 (0.1)	1.08 (0.1)	1.08 (0.1)	0.98 (0.09)	0.89 (0.08)	0.71 (0.06)	.53 (0.05)	0.36 (0.03)
% change in lambda	NA	NS	NS (-1)	-10	-18	-34	-52	-67
Threshold for significant change in lambda	-9.1 % ~ 190 µg/L							
Carbofuran	0 µg/L	50 µg/L	100 µg/L	150 µg/L	164 µg/L	200 µg/L	250 µg/L	350 µg/L
% dead	0	1	14	42	50	67	82	94
Lambda (STD)	1.09 (0.10)	1.09 (0.10)	1.04 (0.10)	0.93 (0.08)	0.89 (0.08)	0.78 (0.07)	0.67 (0.06)	0.5 (0.04)
% change in lambda	NA	NS	NS (-4)	-15	-18	-27	-39	-54
Threshold for significant change in lambda	-9.1 % ~ 126 µg/L							
Methomyl	0 µg/L	250 µg/L	400 µg/L	500 µg/L	560 µg/L	700 µg/L	900 µg/L	1,100 µg/L
% dead	0	5	23	40	50	69	85	92
Lambda (STD)	1.09 (0.10)	1.07 (0.10)	1.08 (0.9)	0.94 (0.08)	0.89 (0.08)	0.78 (0.07)	0.64 (0.06)	0.54 (0.05)
% change in lambda	NA	NS(-1)	NS(-7)	-14	-18	-29	-41	-51
Threshold for significant change in lambda	-9.1 % ~ 431 µg/L							

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

Table 75. Modeled output for Stream-type Chinook salmon exposed to 4 d exposures of carbaryl, carbofuran, and methomyl reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population

Carbaryl	0 μg/L	50 μg/L	100 μg/L	200 μg/L	250 μg/L	350 μg/L	500 μg/L	750 μg/L
% dead	0	0	3	31	50	77	93	98
Lambda (STD)	1.00 (0.03)	1.00 (0.03)	0.99 (0.03)	0.91 (0.03)	0.84 (0.03)	0.69 (0.02)	0.53 (0.02)	0.37 (0.01)
% change in lambda	NA	NS	NS (-1)	-9	-16	-31	-47	-63
Threshold for significant change in lambda	-3.1 % ~ 129 μg/L							
Carbofuran	0 μg/L	50 μg/L	100 μg/L	150 μg/L	164 μg/L	200 μg/L	250 μg/L	350 μg/L
% dead	0	1	14	42	50	67	82	94
Lambda (STD)	1.0 (0.03)	1.0 (0.03)	0.96 (0.03)	0.87 (0.03)	0.84 (0.03)	0.76 (0.02)	0.65 (0.02)	0.5 (0.01)
	NA	NS	-4	-13	-16	-24	-35	-50
Threshold for significant change in lambda	-3.1 % ~ 89 μg/L							
Methomyl	0 μg/L	250 μg/L	400 μg/L	500 μg/L	560 μg/L	700 μg/L	900 μg/L	1,100 μg/L
% dead	0	5	23	40	50	69	85	92
Lambda (STD)	1.0 (0.03)	0.99 (0.03)	0.94 (0.03)	0.88 (0.03)	0.84 (0.03)	0.75 (0.02)	0.63 (0.02)	0.54 (0.02)
% change in lambda	NA	NS(-1)	-6	-12	-16	-25	-37	-46
Threshold for significant change in lambda	-3.1 % ~ 287 μg/L							

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

Table 76. Modeled output for Coho salmon exposed to 4 d exposures of carbaryl, carbofuran, and methomyl reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population.

Carbaryl	0 μg/L	50 μg/L	100 μg/L	200 μg/L	250 μg/L	350 μg/L	500 μg/L	750 μg/L
% dead	0	0	3	31	50	77	93	98
Lambda (STD)	1.03 (0.05)	1.03 (0.05)	1.02 (0.05)	0.91 (0.05)	0.82 (0.04)	0.69 (0.03)	0.42 (0.02)	0.27 (0.01)
% change in lambda	NA	NS	NS(-1)	-12	-21	-39	-58	-74
Threshold for significant change in lambda	-5.3 % ~ 144 μg/L							
Carbofuran	0 μg/L	50 μg/L	100 μg/L	150 μg/L	164 μg/L	200 μg/L	250 μg/L	350 μg/L
% dead	0	1	14	42	50	67	82	94
Lambda (STD)	1.03 (0.05)	1.02 (0.05)	0.98 (0.05)	0.86 (0.05)	0.82 (0.04)	0.71 (0.04)	0.58 (0.03)	0.40 (0.02)
% change in lambda	NA	NS(-1)	NS(-5)	-17	-21	-31	-44	-61
Threshold for significant change in lambda	-5.3 % ~ 98 μg/L							
Methomyl	0 μg/L	250 μg/L	400 μg/L	500 μg/L	560 μg/L	700 μg/L	900 μg/L	1,100 μg/L
% dead	0	5	23	40	50	69	85	92
Lambda (STD)	1.03 (0.05)	1.01 (0.05)	0.94 (0.05)	0.87 (0.05)	0.82 (0.04)	0.69 (0.04)	0.55 (0.03)	0.44 (0.02)
% change in lambda	NA	NS(-2)	-8	-16	-21	-32	-47	-57
Threshold for significant change in lambda	-5.6 % ~ 338 μg/L							

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

Table 77. Modeled output for Sockeye salmon exposed to 4 d exposures of carbaryl, carbofuran, and methomyl reporting the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population

Carbaryl	0 μg/L	50 μg/L	100 μg/L	200 μg/L	250 μg/L	350 μg/L	500 μg/L	750 μg/L
% dead	0	0	3	31	50	77	93	98
Lambda (STD)	1.01 (0.06)	1.01 (0.06)	1.00 (0.06)	0.92 (0.05)	0.86 (0.05)	0.71 (0.04)	0.55 (0.03)	0.40 (0.02)
% change in lambda	NA	NS	NS(-1)	-8	-15	-30	-46	-61
Threshold for significant change in lambda	-5.6 % ~ 169 μg/L							
Carbofuran	0 μg/L	50 μg/L	100 μg/L	150 μg/L	164 μg/L	200 μg/L	250 μg/L	350 μg/L
% dead	0	1	14	42	50	67	82	94
Lambda (STD)	1.01 (0.06)	1.01 (0.06)	0.97 (0.05)	0.89 (0.05)	0.86 (0.05)	0.78 (0.04)	0.67 (0.04)	0.52 (0.03)
% change in lambda	NA	NS	NS(-4)	-12	-15	-23	-34	-48
Threshold for significant change in lambda	-5.6% ~ 114 μg/L							
Methomyl	0 μg/L	250 μg/L	400 μg/L	500 μg/L	560 μg/L	700 μg/L	900 μg/L	1,100 μg/L
% dead	0	5	23	40	50	69	85	92
Lambda (STD)	1.01 (0.06)	1.00 (0.06)	0.95 (0.05)	0.89 (0.05)	0.86 (0.05)	0.76 (0.04)	0.65 (0.04)	0.56 (0.03)
% change in lambda	NA	NS(-1)	-6	-11	-15	-24	-36	-45
Threshold for significant change in lambda	-5.3 % ~ 391 μg/L							

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

The population exercises discussed thus far focused on the effects from exposures to a single annual application of an insecticide; however, we know that these insecticides likely occur together in environmental mixtures and are frequently applied multiple times. To address the potential population-level effects to environmental mixtures of the three insecticides, we ran the same models with a calculated percent mortality predicted using dose-addition. We applied the same additivity model described in the previous *Mixtures* section to predict the cumulative percentage of death in an exposed population. Table 78 shows the model's prediction from three scenarios. We did not use the data from ambient monitoring programs to devise a scenario because the programs were not designed to capture peak concentrations from drift or runoff into juvenile salmonid rearing areas. We did run a few modeling runs with median values of the three insecticides taken from ambient monitoring data. The results showed no statistically significant reductions in lambda for the four populations. In scenario 1, 5% of exposed individuals are expected to die following 4 d exposures to the estimated concentrations from PRZM-EXAMS 24 h averages from EPA's BEs. For carbaryl, we selected a 2 lbs/acre applied aerially with four applications to apples in Oregon which resulted in a concentration of 19 μg/L. For carbofuran, we selected a 2 lb/acre ground application to artichokes in California which resulted in a concentration of 35 μg/L. For methomyl, we selected 0.9 lbs/acre applied ten times to lettuce in California which resulted in a concentration of 88 μg/L. If these events were to occur in a watershed, 5% of the individuals are expected to die, which would not lead to a population-level effect (*i.e.*, reduction in lambda), based on juvenile mortality.

In contrast, the other two scenarios showed substantial and severe percent reductions in lambda for each of the modeled populations; reductions in lambda ranged from 27-52% (Table 78) In scenario 2, we selected 90 d GENEEC-modeled scenarios for corn from Table 50. Concentrations of the three insecticides were as follows: carbaryl = 229 μg/L, carbofuran = 53 μg/L, and methomyl = 49 μg/L. The toxicity is largely a result of the carbaryl concentration, as it was close to the 96 h LC50 of 250 μg/L. In scenario 3, concentrations from drift into off-channel habitats (0.5 m deep) were calculated using the AgDrift model and included a 100 ft no-spray buffer for methomyl. For carbaryl, 5 lb/acre applied once aerially with a fine-medium spray droplet size resulted in a concentration of 335 μg/L. For carbofuran, 1 lb/acre applied once also with a fine-medium spray droplet size resulted in a concentration of 67 μg/L. For methomyl, 0.9 lbs/acre applied once resulted in a concentration of 17.1 μg/L. Cumulatively, these three concentrations would result in 89% mortality of exposed juveniles which reduce lambdas of the four salmonid populations by 41-52%, a substantial effect.

The likelihood of scenarios 2 and 3 occurring is difficult to predict due to the lack of detailed information on watershed characteristics, salmonid presence, the numbers of salmonids exposed, the duration of exposure, and the climatic variables leading to drift. Scenarios 2 and 3 may represent infrequent events, but if a substantial part of a population of listed salmonids is exposed to these mixtures, a severe reduction in a population's abundance is expected.

Table 78. Modeled output for Ocean-type Chinook, Stream-type Chinook, Sockeye, and Coho salmon exposed to 4 d exposures of carbaryl-, carbofuran-, and methomyl-containing mixtures. The table denotes the impacted factors of survival as percent dead, lambda and standard deviation, and percent change in lambda compared to an unexposed population

Scenario 1: PRZM-EXAMS 24-h averages	Ocean-type Chinook	Stream-type Chinook	Sockeye	Coho
% dead	5	5	5	5
Lambda (SD)	1.07 (0.10)	0.99 (0.03)	1.00 (0.06)	1.01 (0.05)
% change in lambda	NS(-1)	NS(-1)	NS(-1)	NS(-2)
Scenario 2: GENEEC 90-d averages	Ocean-type Chinook	Stream-type Chinook	Sockeye	Coho
% dead	74	74	74	74
Lambda (SD)	0.74 (0.07)	0.72 (0.02)	0.74 (0.04)	0.66 (0.04)
% change in lambda	-32	-28	-27	-36
Scenario 3: Offchannel habitats 0.5 m deep	Ocean-type Chinook	Stream-type Chinook	Sockeye	Coho
% dead	89	89	89	89
Lambda (SD)	0.59 (0.05)	0.58 (0.02)	0.60 (0.03)	0.49 (0.03)
% change in lambda	-46	-42	-41	-52

NA denotes non applicable; NS denotes values less than one standard deviation of lambda expressed as the percent of lambda. (Calculated value, omitted when less than or equal to one)

Effects to salmonid populations from reduced size of juveniles due to impaired feeding and reduced abundance of aquatic prey

To evaluate the potential for adverse effects to juvenile growth resulting from carbaryl, carbofuran, and methomyl on Pacific salmonid populations, we developed a model (Appendix 1). The model links AChE inhibition, feeding behavior, prey availability, and somatic growth of individual salmon to the productivity of salmon populations expressed as a percent change in lambda (a population's intrinsic rate of growth). The model scenarios (single insecticide or multiple pulses of single insecticides) assume annual exposure of all the subyearling juveniles in the population and their prey to the insecticide. Similar to the survival model, we developed the growth model for four populations of salmonids: ocean-type Chinook, stream-type Chinook, sockeye, and coho salmon.

We integrated two avenues of effect to juvenile salmonids' growth from exposure to the three *N*-methyl carbamates (Appendix 1). The first avenue is a result of AChE inhibition on the feeding success and subsequent effects to growth of juvenile salmonids. Study results with juvenile salmonids show that feeding success is reduced following exposures to AChE inhibitors (Sandahl, Baldwin et al. 2005). The second avenue the model addresses is the potential for reductions in juvenile growth due to reduction in available prey. Salmon are often found to be food limited in freshwater aquatic habitats, suggesting that a reduction in prey due to insecticide exposure may further stress salmon and lead to reduced growth rates. Field mesocosm data support this assertion, showing reduced growth of juvenile fish following exposure to the AChE inhibitor, chlorpyrifos (Brazner and Kline 1990). Furthermore, based on our review of the sensitivities of aquatic invertebrates to the three insecticides, we expect reductions in densities and altered composition of the salmonid prey communities.

Reductions in aquatic prey are included in the model because of the high relative toxicity of pesticides to salmonid prey and the extended duration of effects on prey communities. Juvenile salmonids are largely opportunistic, feeding on a diverse community of aquatic and terrestrial invertebrate taxa that are entrained in the water column or on the surface (Higgs, Macdonald et al. 1995). As a group, these invertebrates are among the more sensitive taxa for which there is toxicity data, but within this group, there is a wide range of sensitivities (Table 64, Table 65).

The three insecticides are highly toxic to aquatic macroinvertebrates; and concentrations that are not expected to kill salmonids are often lethal for their invertebrate prey (*e.g.*, for carbaryl, range of mean LC50s for salmonids = 250-3,000 μg/L, vs. range of geometric mean EC50s for water fleas = 3-120 μg/L). In particular, prey items that are preferred by small juvenile salmonids (including midge larvae, water fleas, mayflies, caddisflies, and stoneflies) are among the most sensitive aquatic macroinvertebrates. In addition, effects on the prey community can persist for extended periods of time (weeks, months, years), resulting in effects on fish feeding and growth long after an exposure has ended (Ward, Arthington et al. 1995; Van den Brink, van Wijngaarden et al. 1996; Liess and Schulz 1999; Colville, Jones et al. 2008).

Selection of aquatic invertebrate toxicity values to represent salmonid prey items

The model requires for each insecticide an EC50 (defined as a 50% reduction in the biomass of salmonid prey items) and a corresponding slope (Appendix 1). The term "EC50" will be used in this section to describe short-term survival data for aquatic invertebrates (death and immobility). To determine what levels of the three pesticides reduce aquatic invertebrate numbers, we reviewed the available field and laboratory studies. We found robust data for carbaryl, carbofuran, and methomyl with respect to laboratory acute toxicity tests that measured survival at 24-, 48-, 72-, and 96 h with an array of aquatic invertebrates. We did not locate a field study that measured aquatic community response to a range of concentrations of the three insecticides. Therefore, we did not select concentration data from field experiments as we did in NMFS' 2008 Opinion on the registration of chlorpyrifos, diazinon, and malathion (NMFS 2008).

To determine a single effect concentration to use in the model analyses, a search was completed using the EPA's ECOTOX database for each pesticide (http://cfpub.epa.gov/ecotox/). Several criteria were used to determine which reported effect concentrations were included in the final analysis. The data included were from studies on taxa that are known to be salmonid prey (or are functionally similar to salmonid prey); these include a diverse group of aquatic insects and worms and fresh and saltwater crustaceans. Studies with exposures of at least 24 h and not more than 96 h were included. Studies examining shorter and longer exposure times are known to affect invertebrates (Peterson, Jepson et al. 2001), but these were excluded so that estimated EC50s would be comparable. Studies reporting survival EC50s in which mortality or immobilization was the recorded endpoint were included. Data derived for sublethal endpoints (*e.g.,* growth or reproduction) were not included. If specific data were represented more than once in the ECOTOX output, duplicates were eliminated. Data from recent peer-reviewed studies that report survival EC50s were also included (Norberg-King, Durhan et al. 1991; Takahashi and Hanazato 2007). Next, we calculated the geometric mean when more than one survival EC50 was reported for a species.

Probability distributions of aquatic prey survival toxicity values (EC50s)

We plotted the survival EC50 data for each of the three insecticides using cumulative probability distributions. We also plotted the data based on all test results for the species without taking the geometric means. From the distributions of the data, a single effect concentration and slope were derived to best represent the diverse community of prey available in juvenile salmonid freshwater and estuarine habitats. The distributions of individual EC50s and the geometric means of EC50s by taxa were analyzed to estimate the 50th, 10th, and 5th percentiles. Figure 43 shows the distributions of geometric means of EC50s by taxa. Specifically, for each pesticide, a probability plot was used to graph the EC50 concentrations normalized to a normal probability distribution. For each plot, the X axis is scaled in probability (between zero and 100%) and shows the percentage of entire data whose value is less than the data point. The Y axis displays the range of the data on a log scale. The results of a linear regression of the log-transformed concentrations are shown and highlight the lognormal distribution of the data (Figure 43). In the regression equation, the normsinv function returns the inverse of the standard normal cumulative distribution. The standard normal distribution has a mean of zero and a standard deviation of one. For example, given a percentile value of 50 (*i.e.,* a probability of 0.5), normsinv(50) returns a value of zero. The plots and regressions were performed using KaleidaGraph 4.03 (Synergy Software).

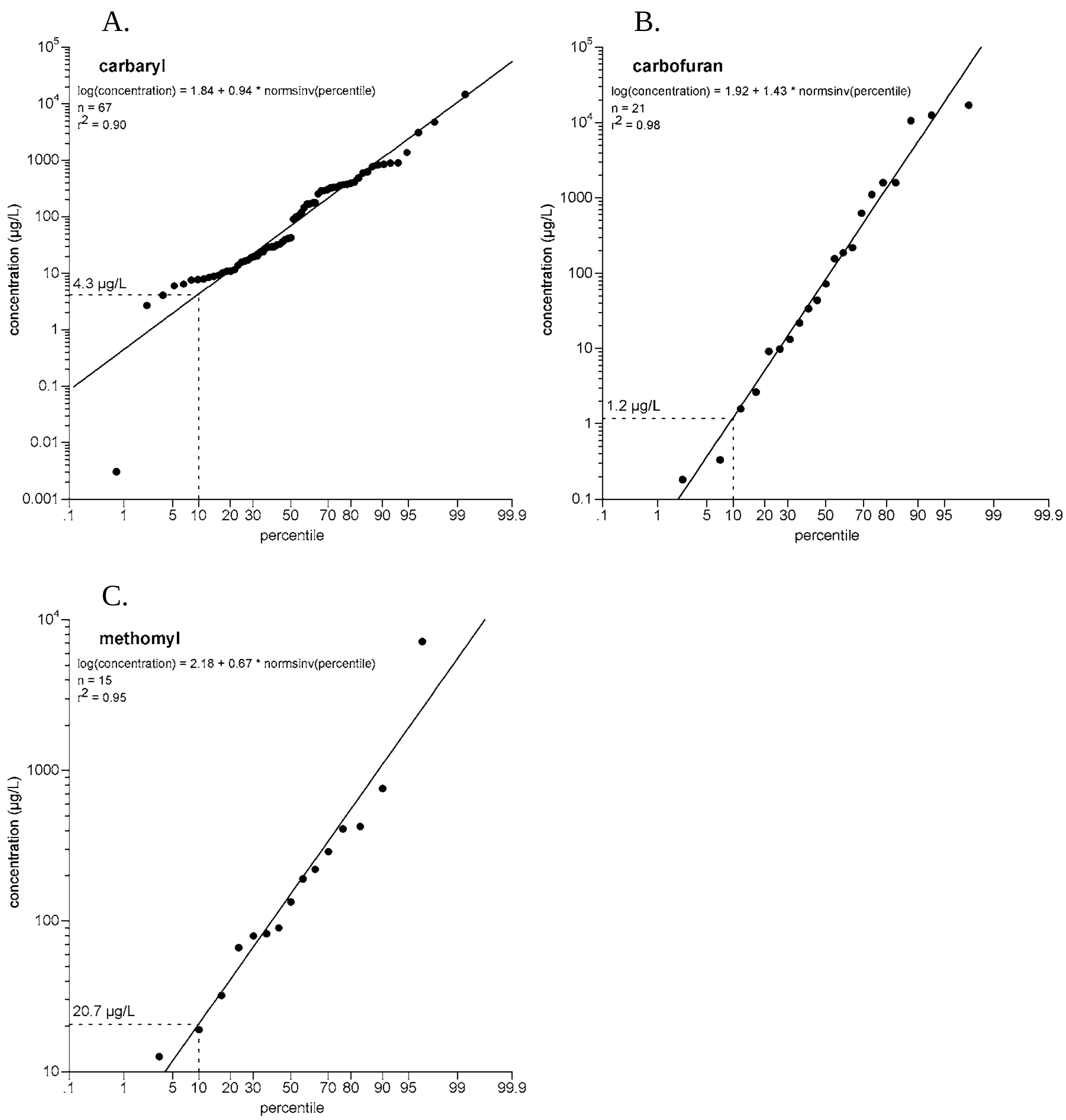

Figure 43. Probability plot for each pesticide showing the distribution of the geometric means of the EC50s for each aquatic invertebrate species. The straight line shows the result of a linear regression. In the regression equation, the normsinv() function returns the inverse of the standard normal cumulative distribution. See text for more details. A) Plot of carbaryl EC50s. B) Plot of carbofuran EC50s. C) Plot of methomyl EC50s.

In Table 79, concentrations are reported for each insecticide associated with either the 50^{th}, 10^{th}, or 5^{th} percentiles derived from probability distribution plots with all study results (plots not shown) or with species geometric means. Percentiles from the species geometric means show higher concentrations compared to the "all studies" plots. This is likely a reflection that when individual study results are considered separately, the species with the greater number of EC50 study results is found at the lower end of the distribution.

Table 79. Carbaryl, carbofuran, and methomyl survival EC50 concentrations at 50th , 10th , and 5th percentiles from probability distribution plots.

	tration of EC50 at each percentile (μg/L)		
	50%	10%	5%
All studies probability distribution plot			
Carbaryl	45.23	2.29	0.98
Carbofuran	58.95	0.94	0.29
Methomyl	128.9	12.93	6.74
Species geometric means probability distribution plot			
Carbaryl	69.53	4.33	1.97
Carbofuran	89.95	1.22	0.37
Methomyl	150.76	20.74	11.82

We selected the 10th percentile from each of the pesticide plots to represent the survival EC50 for salmonid prey. The associated 10th percentile for carbaryl (4.33 μg/L), carbofuran (1.22 μg/L), and methomyl (20.74 μg/L) was used as input for the population growth model exercises. The 10th percentile is a reasonable selection because the data included in the meta-analysis were limited to concentrations that caused mortality or immobilization within a short period of time (1-4 days). A growing number of studies on a variety of insecticides have reported that concentrations well below LC50s can cause delayed mortality or sublethal effects that may scale up to affect aquatic invertebrate populations, especially in scenarios with multiple exposures and/or other stressors. Evidence for ecologically significant sublethal or delayed effects to aquatic invertebrates includes reduced growth rates (Schulz and Liess 2001; Forbes and Cold 2005), altered behavior (Johnson, Jepson et al. 2008), reduced emergence (Schulz and Liess 2001; Johnson, Jepson et al. 2008), reduced reproduction (Cold and Forbes 2004; Forbes and Cold 2005), and reduced predator defenses (Sakamoto, Chang et al. 2006; Johnson, Jepson et al. 2008).

Additionally, the available toxicity data – and therefore the data included for these analyses– are from studies using taxa hearty enough to survive laboratory conditions. Studies specifically examining salmonid prey that are more difficult to rear in the laboratory have documented relatively low survival EC50 values when exposed to current use insecticides (Johnson, Jepson et al. 2008), including carbaryl (Peterson, Jepson et al. 2001; Johnson, Jepson et al. 2008).

The selection of the 10th percentile is a reasonable choice in keeping with general risk management practices of protecting the aquatic community. The standard procedure the U.S. EPA Office of Water uses in establishing Aquatic Life Criteria is to protect to the 5th percentile of a species sensitivity distribution that includes all genera of aquatic species for which valid data are available (EPA 1995). Their procedure involves combining species data into a genus mean value so that multiple tests on one sensitive species do not unduly influence the outcome, and evaluating a specified group of taxa. As we had already selected for the sensitive end of the spectrum by using salmonid prey items, and did not collapse species data into a genus mean, we used the 10th percentile. In the probabilistic risk assessment for carbofuran (EPA 2005), OPP evaluated effects on the 5th, 50th, and 95th percentiles of separate species sensitivity distributions for fish and invertebrates, although it was not clear in the document which was the preferred percentile for risk management.

Modeling availability of unaffected prey

Reductions in benthic invertebrate densities can lead to long-term reductions in prey availability and reductions in fish growth (Davies and Cook 1993). That said, prey densities are not usually reduced to zero (Wallace, Lugthart et al. 1989). Therefore, it is assumed that regardless of the exposure scenario, prey abundance would not drop below a specific “floor” of prey availability. This floor is included in the model to reflect an assumption that a minimal yet constant terrestrial subsidy of prey and/or an aquatic community with tolerant individuals would be available as prey, regardless of pesticide exposure and in addition to the constant recovery rate (see below).

Therefore, even in extreme exposure scenarios, some prey will be available, as determined by the value assigned to the floor; in some highly degraded systems this may or may not be the case. No studies have quantified this floor for the purpose of estimating prey availability, but several studies have documented reductions in overall benthic insect densities of 75-98% (Wallace, Lugthart et al. 1989; Anderson, Hunt et al. 2003; Anderson, Phillips et al. 2006). Because benthic densities are typically correlated with drift densities (Hildebrand 1974; Waters and Hokenstrom 1980), these reductions likely result in similar reductions of prey. Therefore, assuming there is also some constant rate of terrestrial invertebrate subsidy in addition to a residual aquatic community, a floor of 0.20, or 20% of fish ration, is reasonable. The model does not include any additional impacts to fish via dietary exposure from contaminated prey, or any potential synergistic or additive effects to the aquatic invertebrates that may be result from multiple stressors (Schulz and Liess 2001).

Modeling spikes in invertebrate drift following insecticide exposure

"Catastrophic drift" of invertebrates, due to acute mortality and/or emigration of benthic prey into the water column is frequently observed following exposure to insecticides (Davies and Cook 1993; Schulz and Liess 2001; Schulz 2004). Drift rates within hours of exposure can be more than 10,000 times the natural background drift (Cuffney 1984), and fish have been found to exploit this by feeding beyond satiation (Haines 1981; Davies and Cook 1993). The duration and magnitude of the spike in drift of prey is dependent in part on the physical properties and dose of the pesticide; however, the spike is generally ephemeral and returns to natural, background levels within hours to days (Haines 1981; Kreutzweiser and Sibley 1991). Likewise, the magnitude of the spike is dependent in part on the benthic density of prey; the spike in drift from communities that have been reduced by previous exposures is smaller than the spike from previously undisturbed communities (Cuffney 1984; Wallace, Huryn et al. 1991). To reflect this temporary increase in prey availability, the model includes a one-day prey spike for the day following an exposure (Appendix 1). The model also accounts for this short-term increase in prey availability by allowing fish to feed at a maximum rate of 1.5 times their normal, optimal ration.

Modeling recovery of salmonid prey

We selected a 1% recovery in prey biomass per day. Reports of recovery of invertebrate prey populations, once pesticide exposure has ended, range from within days to more than a year (Cuffney 1984; Kreutzweiser and Sibley 1991; Pusey, Arthington et al. 1994; Ward, Arthington et al. 1995; Van den Brink, van Wijngaarden et al. 1996; Liess and Schulz 1999; Colville, Jones et al. 2008). The dynamics of recovery are complicated by several factors, including the details of the pesticide exposure(s) as well as habitat and landscape conditions (Liess and Schulz 1999) (Van den Brink, Baveco et al. 2007). In watersheds with undisturbed upstream habitats, recovery can be rapid due to a healthy source of invertebrates that can immigrate via drift and/or aerial colonization (for adult insects) (Heckmann and Friberg 2005). However, in watersheds dominated by agricultural or urban land uses, healthy upstream or nearby habitats may be limited and consequently, recolonization by salmonid prey is likely reduced (Liess and Von der Ohe 2005; Schriever, Ball et al. 2007). Additionally, many large, high-quality prey take a year or more to develop (Merritt and Cummins 1995) indicating that recovery of biomass (as compared to prey density) is likely a limiting factor (Cuffney 1984). Recovery to pre-disturbance levels is unlikely in aquatic habitats where invertebrate abundances are repeatedly reduced by stressors. We consider a 1% (control prey abundance per day) recovery rate as ecologically realistic to represent recolonization by invertebrates in salmonid habitats (Ward, Arthington et al. 1995; Van den Brink, van Wijngaarden et al. 1996; Colville, Jones et al. 2008).

Growth model results

Exposure to single insecticides for 4-, 21- , and 60 day exposure durations

Population model outputs for the four salmon populations are summarized as dose-response curves in Figures 44-47. As expected, greater reductions in population growth resulted from longer exposures to the insecticides. The primary factor driving the magnitude of change in lambda was the Prey Abundance parameter for each insecticide *i.e.*, the 10th percentile survival EC50 for salmonid prey. The AChE parameter for each of the insecticides was a secondary factor compared to Prey Abundance. This is largely because the salmonid EC50s for AChE were much higher, typically by an order of magnitude, than the prey survival EC50s.

Similar trends in effects were seen for each pesticide across all four life history strategies modeled. This is apparent by the similar shape of the dose-response curves across species. The curves plateau when there is no more reduction possible in the aquatic community *i.e.,* the 20% biomass of the aquatic invertebrate community is reached. Once that plateau is achieved, further reductions in lambda are minimal with increasing concentrations. Clearly, the most toxic of the pesticides affected salmon populations at lower concentrations, as is observed with carbaryl and carbofuran where concentrations in the low μg/L range are sufficient to reduce populations' growth rates compared to methomyl where 20 μg/L or greater are needed to reduce lambdas. One factor that contributed to the similar responses observed was the use of the same surrogate toxicity values for all four life history strategies. The stream-type Chinook salmon (Figure 45) and sockeye salmon **(**Figure 46) models produced very similar results as measured as the final output of percent change in population growth rate. The ocean-type Chinook salmon model output (Figure 44) produced the next most extreme response, and coho salmon output (Figure 46) showed the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output **(**Appendix 1**)**. Combining these factors into the transition matrix for each life history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. While some life history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

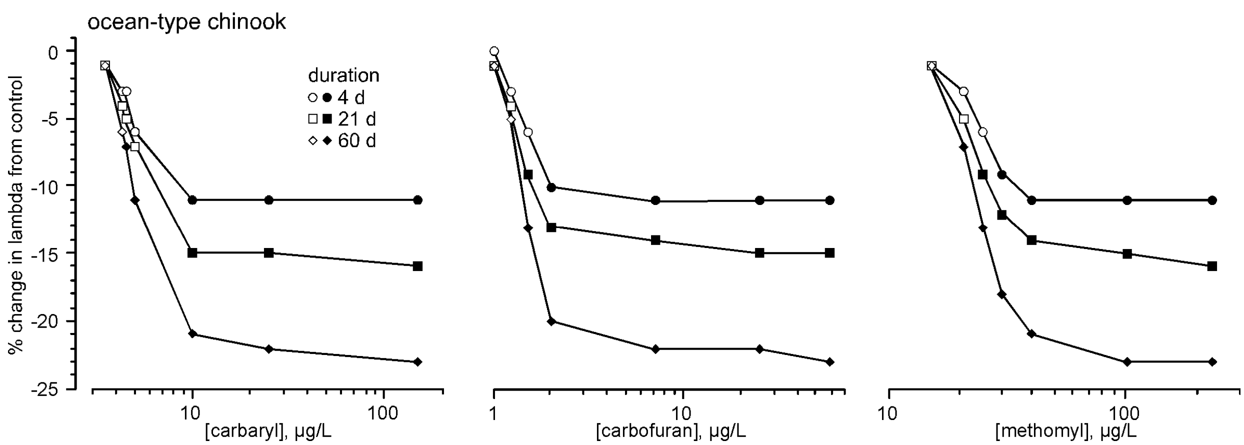

Figure 44. Percent change in lambda for Ocean-type Chinook salmon following 4 d, 21 d, and 60 d exposures to carbaryl, carbofuran, and methomyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

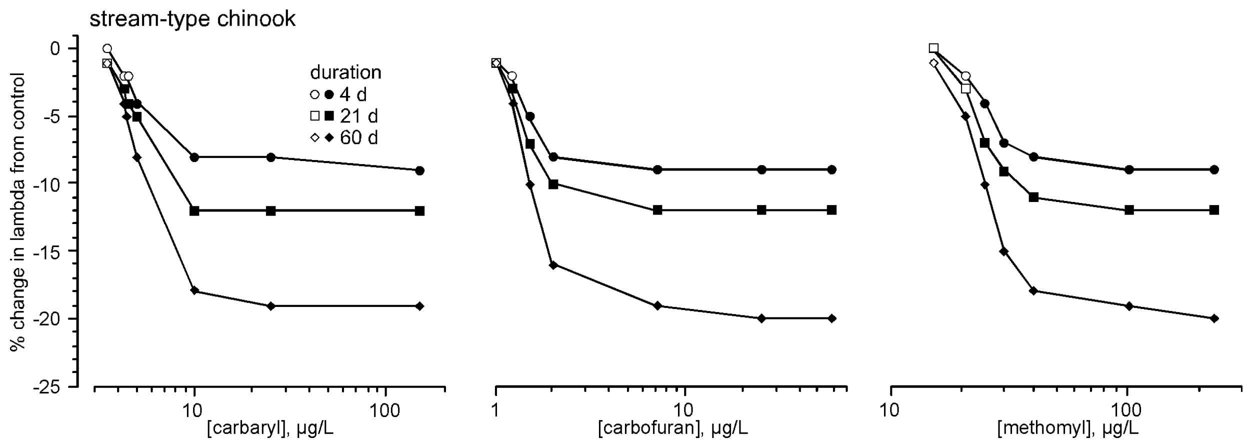

Figure 45. Percent change in lambda for Stream-type Chinook salmon following 4 d, 21 d, and 60 d exposures to carbaryl, carbofuran, and methomyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

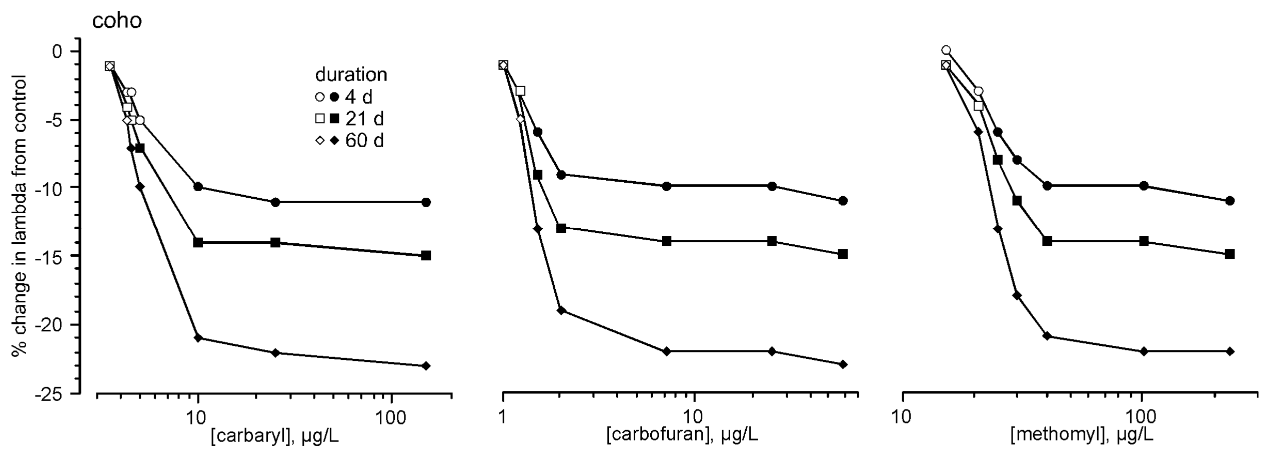

Figure 46. Percent change in lambda for Coho salmon following 4 d, 21 d, and 60 d exposures to carbaryl, carbofuran, and methomyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

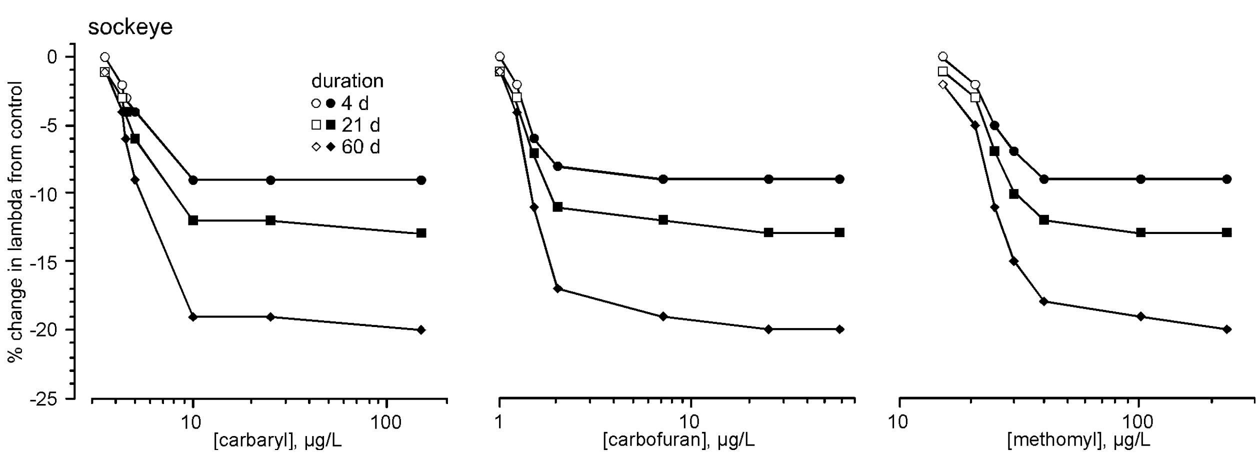

Figure 47. Percent change in lambda for sockeye salmon following 4 d, 21 d, and 60 d exposures to carbaryl, carbofuran, methomyl. Open symbols denote a percent change in lambda of less than one standard deviation from control population. Closed symbols represent a percent change in lambda of more than one standard deviation from control population.

By applying some of these changes in lambda to known threatened and endangered populations' lambdas from Appendix 2, significant reductions in population viabilities are anticipated. For example, if the Puget Sound Chinook salmon Green River population with a lambda of 0.67 is exposed to carbaryl at 10.0 μg/L for 4 d, a concentration attainable in off-channel habitats, we would expect a reduction in lambda by 11% (Table 74) or 8% (Table 75) depending whether the individuals exhibit ocean-type or stream-type life histories. These reductions would be severe to the population and are primarily a result of reductions in juvenile growth from impacts to salmonid prey. Even for those lambdas that are well above one such as Central Valley Chinook salmon Spring Runs' Butte Creek population (lambda = 1.3), reductions of 11% would have major consequences to a population's viability from reduced growth of juveniles. The repercussions to these populations' viabilities are increased with increasing concentrations, durations, multiple applications, and when mixtures are incorporated.

Exposure to multiple applications of carbaryl or methomyl

We constructed two scenarios to evaluate the potential population effects from multiple applications of carbaryl and methomyl (Table 80). We used concentrations from AgDrift model outputs to estimate drift contributions of carbaryl into an off-channel habitat (Table 52). Populations exposed to carbaryl drift within off-channel habitats following four applications would experience severe declines in population growth rate ranging from 15-19%. Interestingly, a single application would result in notable reductions in the four modeled population's lambdas as well, ranging from 8-11%. Similar to the carbaryl modeled scenario, methomyl applied to sweet corn ten times (the label allows for 14 applications at 1 d intervals) resulted in significant reductions in lambdas ranging from 6-8%. A single application did not result in a significant reduction in lambda for any of the four populations.

Table 80. Multiple application scenarios for carbaryl and methomyl and predicted percent change in lambdas for salmon populations

	Carbaryl	Methomyl
Crop examples	Almonds, chestnuts, pecans, filberts, walnuts, pistachios	Sweet corn
Application rate	5 lbs a.i./acre	0.45 lbs a.i./acre
Number of applications/yr	4	10
Application interval	14 days	3 days
Method of application	Aerial (fine-medium droplet distribution)	aerial (fine-medium droplet distribution)
No-application Buffer	none	100 ft
Off-channel habitat characteristics	water depth = 0.5 m Initial average concentration 335 μg/L; 24 h exposure	water depth = 0.5 m Initial average concentration 8.55 μg/L; 96 h exposure
% change in Lambda		
Ocean-type Chinook	-19%	-8%
Stream-type Chinook	-15%	-6%
Sockeye	-16%	-7%
Coho	-18%	-8%

Population-level consequences from other affected salmonid assessment endpoints and other stressors of the action

In this section we present the population-level consequences from individual effects that are not amenable to population modeling. In most cases we lack the empirical data to conduct population modeling for these endpoints. Thus, we use qualitative methods to infer population-level responses. We focus on the population metrics of abundance and productivity. Both are metrics used by NMFS to assess a population's viability and both can be compromised by the chemicals assessed in this Opinion. Individual fitness consequences that reduce survival, growth, reproduction, or migration can lead to reduced salmonid population viability if sufficient numbers of individuals comprising a population are affected, and are more pronounced when affected over multiple generations. If the reductions in fitness result in reducing a population's survival or recovery potential, then we consider whether the ESU or DPS is impacted (See *Integration and Synthesis* section).

With the proposed action it is difficult to place an exact number on the percentage of a population that is affected or how frequently a population is affected because of the lack of information on and wide variability in the spatial and temporal uses of the registered formulations containing carbaryl, carbofuran, and methomyl, compounded by the imperfect data on where salmonids are at any given time. However, NMFS has sufficient information to make reasonable inferences from the available use, exposure, and response data on the likelihood of population-level consequences. Below we address whether the remaining fitness level consequences identified from the risk hypotheses affect the viability of salmonid populations. As mentioned earlier, we focus on the potential for reduced population abundance and productivity.

Impaired swimming and olfactory–mediated behaviors

All life stages of salmonids rely on their inherent ability to smell, to swim, and to navigate through a variety of habitats over their life span and to ultimately spawn successfully in natal waters and complete their life cycle. We have shown that exposure to carbaryl and carbofuran, compared to effects concentrations necessary to impair swimming behaviors, appears sufficient to do so in some environments, especially in rearing locations for the juveniles. Although no data were available to evaluate the effect of methomyl on swimming behavior, we find it reasonable to apply the conclusions drawn for carbaryl and carbofuran due to the chemicals sharing a similar mechanism of toxic action. Specifically, we expect that salmonids with impaired swimming behaviors from AChE inhibition will show reduced feeding, delayed or interrupted migration, reduced survival, and reduced reproductive success. We conclude that exposed populations are likely to have reduced abundance and productivity as a result of impaired swimming.

Based on the information we reviewed for carbaryl, carbofuran, and methomyl on salmonid olfaction, we find differences in expected responses. For carbaryl, we conclude that it is unlikely to affect salmonid olfaction at estimated concentrations. For methomyl, we located no information on its effects to fish olfaction, and given the variation in olfactory responses measured from other AChE inhibitors (OPs and carbamates) it is uncertain whether methomyl will affect olfaction. For carbofuran, definitive evidence shows that olfaction in fish is affected at low μg/L concentrations.

Because olfaction plays an important role in a suite of ecologically relevant behaviors that are affected when an individual salmonid's olfaction is impaired, we include this endpoint in our analysis. Lack of predator avoidance behaviors by juvenile and adult salmonids reduces the probability of surviving predation events. Juvenile salmonids with impaired olfaction may fail to properly imprint on their natal waters, which later in life leads to adults straying *i.e.*, migrating into and spawning in streams other than their natal stream. Adults that do not return to natal waters are a functional loss to recruitment of a population. Adult male salmonids that do find their way back to natal stream or river reaches and are subsequently exposed to the three insecticides may lose some or all of their olfactory capacity, even from a short-term exposure. Female salmonids release odorants to trigger male priming hormones and to alert males of a female's spawning condition. However, male fish with reduced olfactory capacity may not detect these cues, as demonstrated in a study on carbofuran (Bretaud, Saglio et al. 2002). Thus spawning synchronization could be compromised and recently laid eggs may go unfertilized. Unfertilized eggs may result in reduced productivity and abundance for a population if sufficient numbers of spawning events are missed. Again, we find it difficult to accurately predict when these impairments and missed spawning opportunities occur, primarily as a result of incomplete pesticide use information, difficulty in conducting field experiments with adult salmonids, and uncertainties surrounding extent of effects and concentrations which may trigger them. Because imprinting, avoiding predators, homing, and spawning are likely affected when exposed to carbofuran, we conclude these additional effects cannot be dismissed. Therefore, we expect populations exposed to carbofuran will show reduced reproductive rates, reduced return rates, and reduced intrinsic rates of growth when sufficient numbers of individuals are affected.

Starvation during a critical life stage transition

Salmonids emerge from redds (nests) with a yolk-sac as their initial food source (yolk-sac fry). Once the yolk-sac has been completely absorbed, they must begin exogenous feeding. Fry have limited energy reserves, and if they are unable to properly swim or detect and capture prey the onset of starvation occurs rapidly. Because juvenile salmon are limited by gape width, prey at this stage are limited to very small aquatic invertebrates. The stressors of the action likely affect this critical life stage transition in several ways, leading to increased early life stage mortality. Impaired swimming and olfaction affects the fry's ability to detect and capture prey. Prey may be killed outright by the stressors of the action leading to reduced prey availability or the complete absence of prey. These same areas also have off-channel habitats where fry seek shelter and food and those areas are highly susceptible to the highest concentrations of the three insecticides. Therefore, we expect reductions in a population's abundance where transitioning yolk-sac fry are exposed to the stressors of the action. All salmonid ESUs share this common life stage transition and therefore are at risk.

Death of returning adults

We discussed and analyzed with models the importance of juveniles to population viability. However, we did not address possible implications of returning adults dying from exposure to the stressors of the action. An adult returning from the ocean to natal freshwaters is important to a population's survival and recovery for many reasons. Notably, less than one percent of adults generally complete their life cycle. For populations with lambdas well below 1, every adult is crucial to a population's viability. We expect that some sensitive adults will die from short-term exposures before they spawn within some of the populations, particularly those that spawn in intensive agricultural watersheds and urban/suburban environments where elevated temperatures, other AChE-inhibiting insecticides, and low pHs are present. The lower end concentrations that kill 50% of exposed salmonids fall in the hundreds of μg/L. For methomyl, we expect that fewer adults would die compared to carbofuran and carbaryl based on differences in toxic potencies. We still expect that sensitive individuals exposed to concentrations below the LC50 will die. These concentrations are expected in habitats receiving drift from aerial applications. The persistence of these concentrations will vary with habitat, but for those habitats with lower pHs and minimal flow sensitive adult salmonids are expected to be killed. We cannot quantify the number of adults lost to a given population in a given year. For those populations where each adult salmonid is important to viability, we expect reductions in both productivity and abundance.

Synergistic toxicity

With certain combinations and specific concentrations of carbaryl and carbofuran, synergism occurs, translating into increased inhibition of AChE and in some cases increased rates of mortality among exposed salmonids. We have no data either supporting or refuting synergism for methomyl. We have no predictive models for synergistic toxicity. However, where we expect co-occurrence of the three insecticides, we expect synergism may occur if high enough concentrations exist. Generally, these concentrations must exceed individual LC50s for the three compounds, which is most likely to occur in areas with extensive crop uses where applications overlap in space and time. In these areas, even more fish would die from synergistic effects than predicted based on the additive toxicity for carbaryl, carbofuran, and methomyl. Juveniles and returning adults may experience synergistic toxicity. Whether or not death occurs is dependent on exposure duration and concentrations of the three insecticides. Typically, adults are less sensitive than early lifestages, however it is very difficult to conduct toxicity assays with pre-spawn adult salmoninds. Pre-spawn adults have used up most of their accumulated fat, converting it into gamete production, and will die soon after spawning. We are unsure how sensitive these adults are to toxic pesticides, but expect in their physiological state that they will be as or more sensitive than juveniles. We expect that some adults that occupy or spawn in shallow, low flow systems will be impacted from synergistic toxicity. Therefore, population-level effects could be more pronounced, depending upon the number of individuals and the importance of those individuals to the survival and recovery of the population. We conclude that based on the expected environmental concentrations of the three insecticides, synergism is likely in many off-channel habitats resulting in increased rates of death to juveniles and to adults.

Toxicity from other stressors of the action

We identified inert ingredients, adjuvants (NP), tank mixtures (recommended on pesticide product labels), degradates (1-napthol, 3-hydroxycarbofuran), and other pesticide a.i.s (malathion, bifenthrin, rotenone, metaldehyde, captan, and cupric sulfate) as toxic to salmonids and their prey. There remain substantial data gaps on the expected concentrations of these chemicals in salmonid habitats. However, some chemicals are detected at concentrations that pose substantial risk to listed salmonids and their prey (*e.g.,* malathion, NP). The risk posed by these other stressors to salmonid populations is complicated by the same factors we discussed for carbaryl, carbofuran, and methomyl (*i.e.,* the numbers of individuals exposed, the uncertainty surrounding the temporal and spatial uses of these chemicals, etc.). Severe population reductions of Atlantic salmon in Canada were attributed specifically to the use of NP within a pesticide formulation (Fairchild, Swansburg et al. 1999; Brown and Fairchild 2003). We conclude that given the use and co-application of these chemicals with carbaryl, carbofuran, and methomyl, exposed individuals are at increased risk of the suite of toxic effects expected from these substances. Substantial uncertainty also exists regarding the identity of other ingredients in formulations further complicating our ability to predict overall toxicity to salmonids and their prey as a result of pesticide use. Exposed populations are at increased risk of reduced abundance and productivity from these chemicals. However, NMFS is unable to accurately describe the level of risk.

Conclusion on population-level effects

We conclude that many of the populations of threatened and endangered salmonids covered by this consultation will likely show reductions in viability, particularly those that are comprised of juvenile life histories that rear for weeks to years in freshwater habitats found in intensive agricultural and residential/urban areas. Juvenile coho salmon, steelhead, and ocean- and stream-type Chinook salmon use these types of rearing areas for extended periods which overlaps with pesticide applications. Of greatest concern are for those independent populations for each ESU or DPS distributed in high use areas of carbaryl, carbofuran, and methomyl.

Based on these facts, we expect that where the geographic range of listed populations overlaps with intensive cropping patterns and urban/residential areas severe effects to abundance and productivity are anticipated from exposure to the three pesticides. However, due to the impending phase out of carbofuran, we expect the likelihood of exposure and therefore toxicity to decline rapidly as existing products stocks are depleted. Population effects are largely a result of reduced salmonid prey abundances and subsequent reduced growth of juveniles. Although less of a factor, individual fitness consequences are also likely in some areas due to impaired swimming.

Effects to Designated Critical Habitat: Evaluation of Risk Hypotheses

Presently, critical habitat has been designated for 26 of the 28 listed salmonid ESUs, all of which is located within the action area. Designated critical habitat within the action area includes spawning and rearing areas, freshwater migratory corridors, and nearshore and estuarine areas, and includes essential physical and biological features. When NMFS designated critical habitat for the 26 salmonid ESU/DPSs, NMFS identified PCEs essential for the conservation of the species. The PCEs NMFS identified are those sites and habitat components that function to support one or more lifestages. For this opinion, the primary PCEs potentially affected are prey availability and water quality degradation in spawning and rearing areas, freshwater migratory corridors, and nearshore and estuarine areas.

The effects of the proposed action on prey and water quality PCEs are addressed below by weighing the available evidence either supporting or refuting the critical habitat-related risk hypotheses. NMFS reviewed and presented the toxicological information available for habitat assessment endpoints within the *Response Analysis* section. Included are the PCEs prey availability and water quality. If these PCEs are likely impacted by the stressors of the action, we address the potential for reductions to the associated conservation value of the designated critical habitat in the *Effects of the Proposed Action to Designated Critical Habitat* Section.

Critical Habitat Risk Hypotheses based on potential effects to PCEs:

> *Risk Hypothesis 1. Exposure to the stressors of the action is sufficient to reduce abundances of aquatic prey items of salmonids.*

We evaluated two lines of evidence to determine whether this hypothesis is supported by the available information. The first is whether data support the occurrence of adverse effects to salmonid prey items from the stressors of the action. The second is whether reductions in abundances of salmonid prey items occur in areas of documented exposure to the stressors of the action. We found overwhelming evidence in support of the first line of evidence. The stressors of the action are expected to kill large numbers and types of aquatic species that serve as prey to salmonids, especially when carbaryl, carbofuran, and methomyl are present together and/or co-occur with other insecticides. The concentrations we summarized indicate that alone each of the insecticides can also kill prey at expected environmental concentrations.

The IBI and other metrics of aquatic community health were reviewed to evaluate the second line of evidence. In areas of intensive agriculture, where we expect use of the stressors of the action, biological integrity is often significantly reduced (Cuffney et al 1997). Many of the preferred salmonid prey items are present only in low numbers or absent altogether in these areas. We see similar depauperate communities in urban areas. We recognize many other limiting factors contribute to poor condition of these aquatic communities. However, these insecticides and their formulations may be responsible for a substantial portion. In fact, several studies have shown toxicity to salmonid prey items from field collected waters and sediment due to pesticide residues (Cuffney, Meador et al. 1997; Hall, Killen et al. 2006).

We therefore expect that spawning, rearing, and migratory areas will be affected by the stressors of the action resulting in reduced abundances of aquatic prey items. The magnitude in reduction of prey is difficult to ascertain, however where designated critical habitats overlap with high potential use areas such as agricultural or urban dominated flood plains, we expect pronounced reductions. These areas support fry and juvenile growth and development which are predicated on an abundant and diverse source of prey. Reductions in prey availability may affect the conservation value and the ability of the habitat features to support salmonids. In summary, the available information shows prey items of ESA-listed salmonids are affected by the stressors of the action to an extent warranting an analysis of whether the conservation value of designated critical habitat is negatively affected.

Risk hypothesis 2. Exposure to the stressors of the action is sufficient to degrade water quality in designated critical habitat.

We addressed this hypothesis by comparing exposure concentrations to toxicity data. The results of the comparison indicate that expected concentrations from the proposed action trigger adverse effect levels for salmonids and their prey (see *Exposure Analysis* and *Response Analysis* sections). Based on pesticide uses described in the *Description of the Action* section coupled with expected concentrations presented in the *Exposure Analysis* section (*i.e.,* concentrations derived from surface water monitoring data, EPA and NMFS modeling estimates), we expect these concentrations to be present in designated critical habitat and therefore to degrade water quality. In many of the watersheds containing designated critical habitats, water quality is identified as a major limiting factor to salmonid production. The proposed action is likely to further degrade water quality in spawning, rearing, and migratory areas. Collectively, this information supports the conclusion that designated critical habitats are likely degraded throughout the four states and further analysis is warranted to determine the potential to reduce the conservation value of designated critical habitats.

We attempted to compare expected concentrations of these pesticides to U.S. Water Quality Criteria designated pursuant the Clean Water Act, however none are published. We then evaluated whether any of the state waters within designated critical habitat are listed as impaired by carbaryl, carbofuran, and/or methomyl by searching state 303(d) lists (discussed in the *Environmental Baseline* section). Currently, state water quality standards have been developed for carbaryl and carbofuran, but not for methomyl. Unfortunately, surface water sampling is not conducted on all freshwater habitats (including designated critical habitats) within the action area and as a result we do not discount areas not sampled. For these reasons, it is also not possible to determine from 303(d) lists the magnitude of water quality degradation to designated critical habitat. We also do not rely on 303(d) lists as the sole source of information for degradation of water quality. In California, Oregon, and Washington some streams have been placed on the 303(d) list for carbaryl and carbofuran (see *Environmental Baseline* section) which indicates that water quality can be degraded by the stressors of the action in salmonid habitats. We located no listings for methomyl, and we located no listings for the three pesticides in the state of Idaho.

Areas of Uncertainty

In this section we list the predominant uncertainties and data gaps uncovered by our analysis of the effects of the proposed action. We do not discuss the entire suite of uncertainties, but highlight those likely have the most influence on the present analysis.

- Description of the action. We lacked a complete description of EPA-authorized uses of pesticides containing carbaryl, carbofuran, and methomyl as described in labeling of all pesticide products containing these a.i.s.

- Exposure resulting from non-agricultural uses. We lacked exposure estimates of stressors of the action associated with non-agricultural uses of these pesticides.

- Exposure to and toxicity of pesticide formulations and adjuvants. Minimal information was found on formulations, adjuvants, and on other/inert ingredients within registered formulations.

- Exposure to mixtures. We lacked information on permitted tank mixtures. Additionally, given that relatively few tank mix combinations are prohibited, it was not feasible to evaluate all potential combinations of tank mixtures. Pesticide mixtures are found in freshwater throughout the listed-salmonid habitat areas. However, mixture constituents and concentrations are highly variable.

- Toxicity of mixtures. The toxicity of most environmental mixtures is unknown.

- Synergistic responses. Exposure to combinations of carbaryl, carbofuran, methomyl, and/or other combinations of OP and carbamate insecticides can result in synergistic responses. However, we are not aware of a method to predict synergistic responses.

Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered by this Opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, NMFS searched for information on future state, tribal, local, or private actions that were reasonably certain to occur in the action area. NMFS conducted electronic searches of business journals, trade journals, and newspapers using Google and other electronic search engines. Those searches produced reports on projected population growth, commercial and industrial growth, and global warming. Trends described below highlight the effects of population growth on existing populations and habitats for all 28 ESUs/DPSs. Changes in the near-term (five-years; 2014) are more likely to occur than longer-term projects (10-years; 2019). Projections are based upon recognized organizations producing best available information and reasonable rough-trend estimates of change stemming from these data. NMFS analysis provides a snapshot of the effects from these future trends on listed ESUs.

States along the Pacific west coast, which also contribute water to major river systems, are projected to have the most rapid growth of any area in the U.S. within the next few decades. This is particularly true for coastal states. California, Idaho, Oregon, and Washington are forecasted to have double digit increases in population for each decade from 2000 to 2030 (USCB 2005). Overall, the west coast region [which also includes four additional states (Arizona, Utah, Nevada, and Alaska) beyond the action area] had a projected population of 65.6 million people in 2005. This figure will eventually grow to 70.0 million in 2010 and 74.4 million in 2015. At this rate, such growth will make the Pacific coast states the most populous region in the nation.

Although general population growth stems from development of metropolitan areas, growth in the western states is projected from the enlargement of smaller cities rather than from major metropolitan areas. Of the 42 metropolitan areas that experienced a 10% growth or greater between 2000 and 2007, only seven have populations greater than one million people. Of these major cities, one and two cities are from Oregon and California, respectively. They include Portland-Vancouver-Beaverton, OR (1.83%/year), Riverside-San Bernadino-Ontario, CA (3.63%/year), and Sacramento-Arden-Arcade-Roseville, CA (2.34%/year).

Urban Growth

As these cities border coastal or riverine systems, diffuse and extensive growth will increase overall volume of contaminant loading from wastewater treatment plants and sediments from sprawling urban and suburban development into riverine, estuarine, and marine habitats. Urban runoff from impervious surfaces and roadways may also contain oil, heavy metals, PAHs, and other chemical pollutants and flow into state surface waters. Inputs of these point and non-point pollution sources into numerous rivers and their tributaries will affect water quality in available spawning and rearing habitat for salmon. Based on the increase in human population growth, we expect an associated increase in the number of NPDES permits issued and the potential listing of more 303(d) waters with high pollutant concentrations in state surface waters.

Mining

Mining has historically been a major component of western state economies. With national output for metals increasing at 4.3% annually (little oil, but some gas is drawn from western states), output of western mines should increase markedly (Woods and Figueroa 2007). Increases in mining activity will add to existing significant levels of mining contaminants entering river basins. Given this trend, we expect existing water degradation in many western streams that feed into or provide spawning habitat for threatened and endangered salmonid populations will be exacerbated.

Agriculture

As the western states have large tracts of irrigated agriculture, a rise in agricultural output is anticipated. Impacts from heightened agricultural production will likely result in two negative impacts on listed Pacific salmonids (Woods and Figueroa 2007). The first impact is the greater use and application of pesticide, fertilizers, and herbicides and their increased concentrations and entry into freshwater systems. Carbaryl, carbofuran, and methomyl, and other pollutants from agricultural runoff may further degrade existing salmonid habitats. Second, increased output and water diversions for agriculture may also place greater demands upon limited water resources. Water diversions will reduce flow rates and alter habitat throughout freshwater systems. As water is drawn off, contaminants will become more concentrated in these systems, exacerbating contamination issues in habitats for protected species.

Recreation

The western states are widely known for scenic and natural beauty. Increasing resident and tourist use will place additional strain on the natural state of park and nature areas that are also utilized by protected species. Hiking, camping, and recreational fishing in these natural areas is unlikely to have any extensive effects on water quality.

The above non-federal actions are likely to pose continuous unquantifiable negative effects on listed salmonids addressed in this Opinion. Each activity has undesirable negative effects on water quality. They include increases in sedimentation, loss of riparian shade (increasing temperatures), increased point and non-point pollution discharges, decreased infiltration of rainwater (leading to decreases in shallow groundwater recharge, decreases in hyporheic flow, and decreases in summer low flows).

Nevertheless, there are also non-federal actions likely to occur in or near surface waters in the action area that may have beneficial effects on the 28 ESUs. They include implementation of riparian improvement measures, fish habitat restoration projects, and BMPs (*e.g.*, associated with timber harvest, grazing, agricultural activities, urban development, road building, recreational activities, and other non-point source pollution controls).

NMFS expects many of the current anthropogenic effects described in the *Environmental Baseline* will continue. Listed Pacific salmonids are exposed to harvest, hatchery, hydropower, and habitat degradation activities. With regard to water quality, fish are continually exposed to pesticides, contaminants, and other pollutants during their early life history phase and during adult migratory returns to their natal streams for spawning.

NMFS also expects the natural phenomena in the action area (*e.g.*, oceanographic features, ongoing and future climate change, storms, natural mortality) will continue to influence listed Pacific salmonids as described in the *Environmental Baseline*. Climate change effects are expected to be evident as alterations to water yield, peak flows, and stream temperature. Other effects, such as increased vulnerability to catastrophic wildfires, may occur as climate change alters the structure and distribution of forest and aquatic systems.

Coupled with EPA's registration of carbaryl, carbofuran, and methomyl, climate change, and the effects from anthropogenic growth on the natural environment will continue to affect and influence the overall distribution, survival, and recovery of Pacific salmonids in California, Idaho, Oregon, and Washington.

Integration and Synthesis

The Integration and Synthesis section is the final step of NMFS' assessment of the risk posed to listed Pacific salmonids and critical habitat as a result of EPA's registration of carbaryl, carbofuran, and methomyl. In this section, we perform two evaluations: whether it is reasonable to expect the proposed action is likely to (1) reduce both survival and recovery of the species in the wild and (2) result in the destruction or adverse modification of designated critical habitat.

To address species survival and recovery, we discuss the likelihood of the proposed action to reduce the viability of the salmonid species to such an extent that increased extinction rates are likely. Specifically, we address whether we expect reductions in a population's abundance and productivity from the stressors of the action to affect a species (ESU or DPS). We also address whether the conservation value of the physical and biological features of critical habitat is retained. We address whether we expect reductions in the PCEs for prey availability and degradation of water quality from the stressors of the action to reduce the conservation value of designated critical habitat. Conclusions for each ESU/DPS and associated designated critical habitat are found in the *Conclusion* section.

Effects of the Proposed Action at the Species Level

In our *Effects of the Proposed Action* section we assessed the effects of the federal action to listed Pacific salmonids. We discussed the exposure, response, and risks to individuals when they co-occur with the allowable uses of carbaryl, carbofuran, and methomyl. In addition to the a.i.s, we also considered their metabolites and degradates, other active and inert ingredients included in their product formulations, tank mixtures, and adjuvants authorized on their product labels within the action area. Our analysis also considered the possibility of co-occurrence of carbaryl, carbofuran, or methomyl with other AChE-inhibiting pesticides resulting in additive or synergistic effects to salmonids within freshwater habitats. We assessed the stressors of the action in the context of differing life histories, declining populations, and degraded water quality and habitat conditions within the distribution of listed species.

The life history of listed salmonids plays a large role in determining exposure. Adult salmon and steelhead spend weeks to several months in freshwater habitats during their migration and spawning. Immediately after emerging from the gravel substrate, salmonid fry move to habitat where they can swim freely and forage. Chinook salmon, coho salmon, and steelhead fry typically select off-channel habitats associated with their natal rivers and streams. Off-channel habitats provide fry with protection from the primary water flow within rivers and streams. Diverse invertebrate communities also populate off-channel habitats, adding to juvenile salmonids' reliance on these areas. Coho salmon and steelhead rear in freshwater for more than a year. Sockeye salmon fry most frequently distribute to shallow beach areas in the littoral zones of lakes before moving into deeper water and taking on a more pelagic existence. In contrast, chum salmon migrate downstream to estuaries near the mouth of their parent stream almost immediately. Fry reside in estuaries for as little as one or two weeks before moving offshore or into deeper habitats within the nearshore environment.

Based on general knowledge of field runoff and drift from pesticide applications, we expect higher pesticide concentrations in edge-of-field, low-flow, and shallow aquatic systems. Therefore, Chinook salmon, coho salmon, and steelhead have an increased risk of exposure due to their preference for off-channel habitats. Coho salmon and steelhead are at greater risk of exposure because of their extended residency periods.

We expect carbaryl, carbofuran, and methomyl found in surface water runoff and pesticide drift will cause acute lethality and sublethal effects to juvenile salmonids. Based on exposure for four days at the reported LC50s, a severe consequence to a salmonid population's growth rate is expected. The most pronounced effects are within off-channel habitats based on NMFS exposure estimates. The likelihood of population-level effects as a result of mortalities of juvenile salmonids increases as populations spend longer periods in freshwater adjacent to agricultural or developed areas. Species at the greatest risk include steelhead, coho, stream-type Chinook, and ocean-type Chinook salmon.

Methomyl concentrations required to result in population-level impacts are not expected to occur in the majority of aquatic habitats used by juveniles. However, we expect that carbaryl and carbofuran applications could cause population-level effects through acute mortality of juveniles, particularly in areas of intensive agriculture and large urban or residential areas. We also expect acute mortalities of juveniles from applications of carbaryl to forests and from exposure to 24(c) carbaryl applications in estuaries in Washington State. However, we do not know how many individuals are exposed each year due to these uses.

Sublethal effects caused by AChE inhibition, such as impaired swimming and olfactory-mediated behaviors, are also expected to result in adverse population-level effects. Swimming is a critical function for salmonids. Individuals with impaired swimming ability will show reduced feeding, delayed or interrupted migration, reduced survival, and reduced reproductive success. We expect that carbaryl, carbofuran, and methomyl will occur at concentrations high enough to impair swimming ability in occupied shallow water habitats. These effects may lead to population-level consequences that hinder the survival and recovery of listed species.

Olfactory-mediated behaviors are also vital for salmonid survival. These behaviors include detecting and avoiding predators, recognizing kin, locating natal waters for spawning, and reproduction. Juvenile salmonids with impaired olfactory-mediated behaviors can lose their ability to imprint on natal streams and avoid predators. Spawning is triggered by olfactory-mediated cues given and received by both sexes. If olfaction is inhibited, salmon may not recognize these cues, resulting in reduced reproduction. The available information suggests that carbaryl does not inhibit olfaction. Therefore, we do not expect to see effects to olfactory mediated behaviors from carbaryl exposure. Currently there is not enough information to determine the effect of methomyl on olfaction. However, exposure to carbofuran at low μg/L concentrations will likely result in the impairment of olfactory-mediated behaviors. That in turn will lead to increased predation and ultimately reduced juvenile and adult survivorship. As pesticide exposure increases we expect to see more pronounced effects due to additive and synergistic effects.

We also expect indirect effects from the a.i.s through reduced salmonid prey abundance and subsequent reduced juvenile growth and probability of survival. Feeding during the period immediately following emergence is vital for the survival of all ESA-listed Pacific salmonid fry.

Once fry have completely absorbed the yolk-sac, they are dependent on the local invertebrate population and may starve if sufficient food is not available. As salmonid prey are sensitive to the expected exposures to each of the insecticides, as well as from mixtures containing the three a.i.s, we expect death and a variety of sublethal effects to salmonid prey. If the salmonid prey base is limited while fry are in this life stage, juvenile growth will be significantly hindered. Many of the invertebrate communities are already degraded due to poor habitat conditions and poor water quality. These conditions make it more difficult for the prey base to recover following an exposure event. Thus, juvenile survivorship will be low and may consequently lead to a decrease in lambda for the affected population.

Based on the NMFS growth models, a four-day exposure to expected concentrations of carbaryl, carbofuran, and methomyl will substantially reduce a population's growth rate due to prey base reduction. All four life history types modeled demonstrated this effect. As exposure duration increases, we expect more pronounced effects on salmonid prey abundance and a longer prey recovery period leading to further reduced salmonid growth and decreased survival.

We expect that all effects will be more pronounced in drainage basins that include high use areas, *i.e.*, major agricultural, urban, or residential centers. To determine which ESUs/DPSs will be affected by the action, we compare the range of each species to the locations of high use land classes. Species that occupy major agricultural basins, such as the Willamette basin or California Central Valley, have a much higher risk of exposure than species found in uninhabited, non-agricultural areas.

California Coastal Chinook Salmon

The CC Chinook salmon ESU consists of 10 historically functionally independent populations in California from Humboldt County in the north to Sonoma County in the south. Historic annual escapement was around 73,000 fish with most of these being produced in the Eel River. Today, annual returns to the Eel River system is estimated at 150 to 2,800 fish. Runs in the Russian River may be viable, though the short-term trend is negative. Lack of data precludes development of population growth rates or trends for all other populations.

The *Status of Listed Resources* and *Environmental Baseline* sections indicate that fisheries, timber harvest, vineyards and other agriculture, non-native fish species, and migration barriers negatively affect this ESU. Adverse effects on Chinook salmon habitat include a high percentage of fines in the streams' bottom substrate, lack of large instream woody debris, reduced riparian vegetation, elevated water temperatures, degraded water quality, increased predation, and barriers that limit access to tributaries. The combined impacts from these multiple threats continue to affect the CC Chinook salmon. Accumulation of fine sediment in the bottom substrate, contamination of waters, lack of riparian vegetation, and non-natives fish species are expected to impact invertebrate abundance and composition in streams within the ESU.

Most of the agriculture and urban developments are concentrated in the Russian River valley and on alluvial plains of coastal rivers. Vineyards and smaller centers are dispersed along coastal watersheds. Based on the crop types, we expect carbaryl and methomyl are commonly applied throughout the growing season in the Russian River basin. We also expect use of the a.i.s on the alluvial plain of the lower Eel River and its estuary, and to a lesser extent along the lower portions and estuaries of other coastal streams. Carbaryl is expected to be used the entire year within urban/residential areas. Existing stocks of carbofuran are also expected to be applied throughout the growing season. In California, there are 61 pesticides that inhibit AChE approved for use and we expect that application of other AChE inhibiting pesticides will co-occur with the a.i.s, exacerbating adverse effects from AChE inhibition. However, monitoring data for pesticides in streams within the CC Chinook salmon ESU is lacking. Adults within Willapa Bay during 24(c) carbaryl application to oyster beds are likely exposed to this compound and may experience additional mortalities.

CC Chinook spawning occurs on gravel beds in the mainstem of rivers and larger tributaries. Fry use the floodplain, stream margins, and side channels to rear after emergence from the redds in December through mid-April. Downstream migration of juveniles starts as early as February and most enter marine waters before mid-summer.

We expect the proposed uses of carbaryl and carbofuran pesticide products that contaminate aquatic habitats will lead to individual fitness level consequences and subsequent population-level consequences. Land use and crop type data indicate that, while not a major agricultural center, the CC Chinook will be exposed to all three a.i.s. We expect that exposure to methomyl may lead to individual fitness consequences, but not to an extent that would affect population growth rates. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl and carbofuran, but low for methomyl.

Central Valley Spring-run Chinook Salmon

The CV Spring-run Chinook salmon ESU includes four populations of Spring-run populations in the upper Sacramento River and its tributaries. The distribution within the Sacramento River basin is now mostly restricted to accessible areas below dams in the mainstem river and in three of its tributaries: Deer, Mill, and Butte Creeks. Abundance remains far below the estimated 700,000 once entering the Sacramento-San Joaquin Rivers system. The number of Spring-run Chinook salmon spawning in the Sacramento River has averaged about 9,800 annually since 2000. While most populations within the ESU are at or above replacement, the Sacramento River population has been steadily decreasing.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include impaired or loss of habitat, predation, contamination, and water management. Reservoir dams in the Sacramento River have prevented the ESU from using its historic spawning locations. Physical channel habitat has been altered through sediment input from mining, levee construction, and removal of riparian vegetation for levee maintenance. Detected pesticides in the Sacramento River include thiobencarb, carbofuran, molinate, simazine, metolachlor, and dacthal, chlorpyrifos, carbaryl, and diazinon. State and federal water diversions in the south Sacramento-San Joaquin Delta (Delta) have resulted in increased mortality through prolonged migration and entrainment at the water diversion facilities. We also expect that application of other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters within the ESU and exacerbate adverse effects from AChE inhibition.

With the high density of agriculture in the Sacramento River valley and the Sacramento-San Joaquin Delta (Delta), application of all three a.i.s is expected. Large urban centers occur along the Sacramento River and San Francisco Bay. Young and adult migrating Chinook salmon are also exposed to poor water quality from agricultural runoff that enters the Delta from the San Joaquin River. Carbaryl and methomyl can be applied to several crops throughout the growing season in the Central Valley. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Carbaryl is approved for use throughout the year within urban/residential areas

The CV Spring-run Chinook salmon is categorized as an ocean-type fish. The young salmon emerge from the gravel from November through May and outmigration starts within four months. While the majority emigrates soon after emergence, some juveniles emigrate as yearlings with the onset of fall storms. Juveniles' seaward migration follows the mainstem Sacramento River, through the Delta, and across San Francisco Bay. Off-channel habitats within the lower Sacramento River floodplains, especially the Yollo Bypass, are particularity important for CV Chinook salmon juveniles during rearing and migration (Sommer, Nobriga et al. 2001; Sommer, Harrell et al. 2005).

The widespread uses of these pesticides have substantial overlap with rearing and migratory habitat of the CV Spring-run Chinook salmon ESU. The CV Spring-run Chinook salmon exhibit high variation in abundance, have restricted distribution, and are exposed to pesticides and other pollutants during rearing and downstream migration. We expect the proposed uses of the three carbamates may contaminate aquatic habitats and lead to both individual fitness and subsequentr population-level consequesnces. Therefore, the risk to this species' survival and recovery from the stressors of the action is high from carbaryl, carbofuran, and methomyl.

Lower Columbia River Chinook salmon

The LCR Chinook salmon ESU includes 32 historical populations in tributaries from the ocean to the Big White Salmon River, Washington and Hood River, Oregon. The ESU also includes 17 artificial propagation programs. LCR Chinook salmon numbers began to decline by the early 1900s from habitat degradation and excessive harvest rates. Many of these populations have low abundance. The annual population growth rates for 14 independent populations range from 0.93 to 1.037.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include hydromorphological changes from hydropower development, loss of tidal marsh and swamp habitat, and reduced or eliminated access to subbasin headwaters from by the construction of non-federal dams. LCR Chinook salmon spawning and rearing habitats in tributary mainstems have been adversely affected by sedimentation, elevated water temperature, and reduced habitat diversity. The survival of yearlings in the ocean is also affected by habitat conditions in the estuary, such as changes in food availability and the presence of contaminants.

NAWQA sampling in surface waters within the ESU range detected more than 50 pesticides in streams. Concentrations of ten pesticides, including carbaryl and carbofuran, also exceeded EPA's chronic toxicity aquatic life criteria (Wentz, Bonn et al. 1998). The combined impacts from these multiple threats continue to affect this ESU.

Most of the highly developed land and agricultural areas in this ESU's range are adjacent to salmonid habitat. Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. Based on land use patterns, we expect the highest exposure to these chemicals to occur in the Lewis River basin, Clackamas River basin, and Hood River basin.

Mature LCR Fall-run Chinook salmon enter freshwater in August through October to spawn in large river mainstems. After emergence, fry typically select off-channel habitats associated with their natal rivers and streams. Juveniles eventually emigrate from freshwater, usually within six months of hatching. LCR Spring-run Chinook salmon enter freshwater in March through June to spawn in upstream tributaries. These fish generally emigrate from freshwater as yearlings. As juveniles overwinter in shallow, freshwater habitats, they are likely to experience higher exposure to pesticides, other contaminants, and elevated temperature. In northern rivers, juveniles may rear in freshwater for two years or more. Given their long residence time in shallow freshwater habitats, LCR Chinook salmon are vulnerable to high pesticide exposures. Also, if adults and juveniles are within Willapa Bay during 24(c) carbaryl application to oyster beds they are likely exposed to this compound and may likely experience additional mortalities.

Given the life history of LCR Chinook salmon, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may contaminate aquatic habitats and lead to individual fitness and subsequent population-level consequences. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Upper Columbia River Spring-run Chinook salmon

The UCR Spring-run Chinook salmon ESU includes 11 populations and 7 artificial propagation programs in the state of Washington. The four known annual population growth rates range from 0.99 to 1.1. Based on 1980-2004 returns, the average annual population growth rate for this ESU is estimated at 0.93. One historical population is considered extinct.

The major threats to UCR Chinook salmon identified in the *Status of Listed Resources* and *Environmental Baseline* sections include reduced tributary stream flow and impaired fish passage from hydroelectric dams. Additionally, degradation of the tributary habitat and impaired water quality from development negatively affect this ESU. Pesticide detections in UCR Chinook salmon freshwater habitats are well documented. NAWQA sampling from 1992-1995 in the Central Columbia Plateau detected numerous pesticides in surface water (Williamson, Munn et al. 1998). Carbaryl was detected in 5% of samples, and carbofuran was detected in 6%. Methomyl was not detected. While detections of these three chemicals did not exceed water quality standards, concentrations of six other pesticides exceeded EPA criteria for the protection of aquatic life. Listed salmonids are commonly exposed to combinations of carbaryl, carbofuran, and methomyl, other compounds, and other mixtures of cholinesterase-inhibiting insecticides.

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses in Washington State include carbofuran application to potatoes and spinach grown for seed. About 3% of the land has been developed and 3.5% has been cultivated for crops. Areas with the potential for high exposure in the Upper Columbia River basin are the Methow, Entiat, and Wenatchee River basins.

UCR Spring-run Chinook salmon begin returning from the ocean in the early spring. After migration, they hold in freshwater tributaries until spawning in mid- to late August. Fish spawn in the major tributaries leading to the Columbia River between Rock Island and Chief Joseph dams. UCR Spring-run Chinook salmon fry typically select off-channel habitats associated with their natal rivers and streams to rear. Juveniles spend a year in freshwater before migrating to the ocean in the spring of their second year of life. The duration of juvenile rearing in shallow freshwater habitats increases their susceptibility to higher exposures of pesticides, contaminants, and elevated temperature.

Given the life history of UCR Spring-run Chinook salmon, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products that contaminate aquatic habitats will lead to individual fitness and subsequent population-level consequences, *i.e.*, reductions in population viability. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Puget Sound Chinook Salmon

The Puget Sound ESU includes all runs of Chinook salmon in the Puget Sound region from the North Fork Nooksack River to the Elwha River on the Olympic Peninsula. This includes 31 historic quasi-independent populations and 26 artificial propagations programs. Of the historic populations, only 22 are considered extant. The estimated total run size for this ESU in the early 1990s was 240,000 fish. During a recent five-year period, the geometric mean of natural spawners in populations of this ESU ranged from 222 to just over 9,489 fish (Good, Waples et al. 2005). Recent five-year and long-term productivity trends remain below replacement for the majority of the 22 extant populations of Puget Sound Chinook salmon. The annual population growth rate known for these populations ranged from 0.75 to 1.17.

The major threats to the Puget Sound Chinook salmon identified in the *Status of Listed Resources* and *Environmental Baseline* sections include degraded freshwater and marine habitat from agricultural activities and urbanization. Poor forestry practices have also reduced water quality in the upper river tributaries for this ESU. Elevated temperature, water diversions, and poor water quality across land use categories pose significant threats to the status of Puget Sound

Chinook salmon. Furthermore, there has been extensive urbanization in this region. Well over two million people inhabit the area, with most development occurring along rivers and coastline.

Pesticide use and detections in the ESU's watershed are well documented. NAWQA sampling conducted in 2006 in the Puget Sound basin detected numerous pesticides and other synthetic organic chemicals in streams and rivers. However, mixtures of chemicals found in agricultural and urban settings differ. Urban streams sampled in Puget Sound showed the highest detections for carbaryl, diazinon, and malathion. Carbaryl was detected at 60% of urban sampling sites (Ebbert, Embrey et al. 2000). Diazinon was detected at all urban sites, frequently at concentrations that exceeded EPA guidelines for protecting aquatic life (Bortleson and Ebbert 2000).

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. In addition, Washington has 24(c) registrations for carbofuran use on spinach grown for seed and potatoes. Land classes indicate high-use of the three a.i.s in the following drainage basins: Nooksack, Skagit River, Stillaguamish, Skykomish, Snoqualimie, Snohomish, Lake Washington, Duwamish, and Puyallup. Roughly 7% of the lowland areas (below 1,000 ft elevation) in the Puget Sound region are covered by impervious surfaces, which increase urban runoff containing pollutants and contaminants into streams. Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, PBDEs, PAHs, pharmaceuticals, nutrients, and sediments. As such, we expect salmonid populations within the ESU to be exposed to carbaryl, carbofuran, and methomyl. Further, monitoring data shows that we can expect concurrent exposure with other AChE inhibiting chemicals, which will exacerbate the severity of effects. Adults within Willapa Bay during 24(c) carbaryl application to oyster beds are likely exposed to this compound and may experience additional mortalities.

Puget Sound stream-type Chinook salmon fry typically rear in shallow off-channel habitats associated with their natal rivers and streams. Juveniles generally have long freshwater residences of one or more years before migrating to the ocean.

Given the life history of the Puget Sound Chinook salmon, we expect that the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may lead to individual fitness consequences and subsequent population-level consequences, *i.e.*, reductions in population viability. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Sacramento River Winter-run Chinook salmon

The Sacramento River Winter-run Chinook salmon ESU includes only one population in Sacramento River, California. The current spawning distribution of Sacramento River Winter-run salmon is restricted to a short portion of the mainstem Sacramento River below Keswick Dam. Historic run estimates for the Sacramento River are as large as 200,000 fish (Brown et al. 1994). Estimated natural production has fluctuated greatly over the past two decades. In 2007, estimated natural production was 4,461 fish. The population's annual growth rate ranged from 0.870 to 1.090.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections indicate impaired or loss of habitat, predation, contamination, and water management negatively affect this ESU. Reservoir dams in the Sacramento River have eliminated the ESU from its historic spawning locations. Today, the ESU depends on the ability of the BOR to manage cold water through reservoir storage and releases to support adult holding, spawning, incubation, and rearing. The physical channel habitat has been altered through sediment input from mining, levee construction, and removal of riparian vegetation for levee maintenance. Pesticides are frequently detected in the Sacramento River including thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Carbofuran is commonly detected in agricultural areas but less so in urban sites while carbaryl is commonly detected in all urban sites. Modification of hydrology has resulted in increased mortality through stranding, increased predation, prolonged migration, and entrainment at water diversion facilities.

About 10% of the land within this ESU is developed; large areas of urban centers occur along the Sacramento River and San Francisco Bay. As about 21% of land within the ESU is cultivated, all three a.i.s are expected to be applied to several crops within the ESU. Agriculture activity is prominent in the lower Sacramento River and within the Delta. Chinook salmon are also exposed to poor water quality from agricultural runoff that enters the Delta from the San Joaquin River. The Mediterranean climate in California, with dry summers and fall storms, may result in high concentration of these contaminants in run-off during the onset of the rainy season. These high concentrations can overlap with juvenile presence in the river system and movement into floodplains for rearing.

Winter-run adults enter the Sacramento River in early spring with spawning peaking in May and June. Spawning occurs in the Sacramento River downstream of the Keswick Dam. Fry rear in the Sacramento River for a few weeks to months before starting outmigration in late July, peaking in November and December. During outmigration, the young salmon migrate down the Sacramento River, through the Delta and San Francisco Bay. Juvenile winter-run Chinook salmon appear in the Delta from October to early May where they may rear in the fresher upstream portions for up to two months.

We expect that the proposed uses of carbaryl, carbofuran, and methomyl pesticide products will lead to both individual fitness level and subsequent population-level consequences. Registered uses of these a.i.s indicate overlap with the spawning, rearing, and migratory habitat of the one extant population in this ESU. Thus, the risk to this species' survival and recovery from the stressors of the proposed action is high for carbaryl, carbofuran, and methomyl.

Snake River Fall-run Chinook salmon

The SR Fall-run Chinook salmon ESU is comprised of a single population that spawns and rears in the mainstem Snake River and its tributaries below Hells Canyon Dam. Only 10 to 15% of the historical range of this ESU remains. Estimated historical returns from 1938 to 1949 were 72,000 fish annually (Bjornn and Horner 1980). The average abundance (1,273) of SR Fall-run Chinook salmon over the most recent 10-year period is below the 3,000 natural spawner average abundance thresholds identified as a minimum for recovery. The annual population growth rate for this single population is 1.02. Two historical populations are considered extirpated.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include impaired stream flows and barriers to fish passage in tributaries from hydroelectric dams. During the 1960s and 1970s, approximately 80% of the ESU's historic habitat was eliminated or severely degraded by the construction of the Hells Canyon complex and the lower Snake River dams. Additionally, degraded freshwater habitats in the estuary, mainstem, and tributaries from development and land use activities negatively affect this ESU. Agricultural activities, urban communities, and industries are concentrated along the Snake River and near the mouths of major tributary valleys. Thus, stream water quality and biological communities in the downstream portion of the upper Snake River basin are degraded. The combined impacts from these multiple threats continue to affect SR Fall-run Chinook salmon.

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses within this ESU include carbofuran application to potatoes and spinach grown for seed in Washington. The remaining spawning and rearing area of the SR Fall-run Chinook salmon is potentially a high-use area for these chemicals, as it is adjacent to agricultural and residential areas. About 14% of the land has been developed. Dryland agriculture occurs in the lower Clearwater Basin, with crops including wheat, peas, and lentils. Alfalfa, hay, and grasses are also grown in the lower Clearwater as well as the lower Salmon River basin. Thus, ESU exposure to these pesticides is likely. Adults within Willapa Bay during 24(c) carbaryl application to oyster beds are likely exposed to this compound and may experience additional mortalities.

Adult SR Fall-run Chinook salmon enter the Columbia River in July and August. Spawning occurs from October through November above Lower Granite Dam in the mainstem Snake River and in the lower reaches of the larger tributaries, such as the Salmon and Clearwater Rivers. Fry emerge from redds beginning in March or April, then rear for two months or more in the sandy littoral zone along the river margin. Juveniles begin migration in June along the edges of rivers, where they are at risk of exposure to higher concentrations of pesticides from drift and runoff. Their duration in shallow freshwater habitats increases their chances of higher exposure to pesticides, contaminants, and elevated temperature. Most fish exhibit an ocean-type life cycle, completing migration to the Pacific during the first year of life, though some overwinter inlow-velocity pools created by the hydroelectric dams.

Given the life history of SR Fall-run Chinook salmon, we expect the proposed uses of carbaryl and carbofuran pesticide products may lead to individual fitness level consequences and subsequent population-level consequences. We expect that exposure to methomyl may lead to individual fitness consequences, but not to an extent that would affect population growth rates. Land class data indicate that exposure of the carbamates to SR Fall-run Chinook salmon is possible. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl and carbofuran, but low for methomyl.

Snake River Spring/Summer-run Chinook salmon

This ESU includes 32 historical populations in the Snake River basin which drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. Historically, the Salmon River system may have supported more than 40% of the total return of Spring/Summer-run Chinook salmon to the Columbia system (Fulton 1968). The long-term trends in productivity indicate a shrinking population. However, recent trends in productivity, buoyed by the last five years, are approaching replacement levels. The annual population growth known for 18 populations ranged from 0.97 to 1.1. Historical populations above Hells Canyon are considered extinct.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include degraded water quality in the freshwater estuary, tributaries, and coastal habitats from land use activities and hydroelectric dams. Significant threats to SR Spring/Summer-run Chinook salmon include elevated temperature, water diversions, and poor water quality.

Some spawning and rearing areas of the SR Spring/Summer Run Chinook are in forested areas, where carbaryl may be applied. The likelihood of exposure in forested areas is unknown. Registered 24(c) uses within this ESU include carbofuran application to potatoes and spinach grown for seed in Washington. Dryland agriculture occurs in lower Clearwater Basin, with crops including wheat, peas, and lentils. Alfalfa, hay, and grasses are also grown in the lower Clearwater as well as the lower Salmon River basin. All three a.i.s are registered for use on alfalfa. Agricultural activities and urban communities, are concentrated along the Snake River and near the mouths of major tributary valleys, thus stream water quality and biological communities in the downstream portion of the upper Snake River basin are degraded.

SR Spring/Summer-run Chinook salmon spawn at high elevations in the headwater tributaries of the Clearwater, Grande, Ronde, Salmon, and Imnaha Rivers. Spawning is complete by the second week of September. Eggs incubate and hatch in late winter and early spring of the following year. The fry typically overwinter in shallow off-channel habitats associated with their natal rivers and streams. Juveniles become active seaward migrants during the following spring as yearlings (Connor, Sneva et al. 2005).

Given the life history of SR Spring/Summer-run Chinook salmon, we expect the proposed uses of carbaryl and carbofuran pesticide products that contaminate aquatic habitats will lead to individual fitness level consequences and subsequent population-level consequences. The uses of these materials along the migratory route of SR Spring/Summer-run Chinook salmon, may cause acute lethality or temporary AChE inhibition. We expect that exposure to methomyl may lead to individual fitness consequences, but not to an extent that would affect population growth rates. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl and carbofuran, but low for methomyl.

Upper Willamette River Chinook Salmon

The UWR Chinook salmon ESU includes all eight naturally spawned populations residing in the Clackamas River and the Upper Willamette River above Willamette Falls. It also includes seven artificial propagation programs. The population in the McKenzie River is the only population that is naturally producing, and current estimates indicate a negative growth rate. Historically, the Upper Willamette River supported large numbers (exceeding 275,000 fish) of UWR Chinook salmon. Current abundance of natural-origin fish is estimated at less than 10,000.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include habitat loss due to blockages from hydroelectric dams and irrigation diversions, and degraded water quality within the Willamette mainstem and the lower reaches of its tributaries. Elevated water temperature also poses a significant threat to the status of UWR Chinook salmon. Fifty pesticides were detected in streams that drain both agricultural and urban areas. Forty-nine pesticides were detected in streams draining agricultural land, while 25 were detected in streams draining urban areas. Ten of these pesticides, including carbaryl and carbofuran, exceeded EPA criteria for the protection of freshwater aquatic life from chronic toxicity (Wentz, Bonn et al. 1998). The combined impacts from these threats continue to affect UWR Chinook salmon.

Based on the crop types, we expect that carbaryl and methomyl are commonly applied in the Willamette Valley throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. The Willamette Basin is the largest agricultural area in Oregon. In 1992 the Willamette Basin accounted for 51% of Oregon's total gross farm sales and 58% of Oregon's crop sales. About one-third of the agricultural land is irrigated and most of it is adjacent to the mainstem Willamette River. Urban developments are also located primarily in the valley along the mainstem Willamette River. We expect carbaryl to be used throughout the year in urban and residential areas. Given that major urban and agricultural areas are located adjacent to the mainstem Willamette, ESU exposure to these pesticides is likely. We also expect that other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters of the Willamette Valley and exacerbate adverse effects from AChE inhibition.

Chinook salmon fry typically select shallow off-channel habitats associated with their natal rivers and streams. Juveniles generally rear in freshwater for several months to more than one year before migrating to the ocean. Their duration in shallow freshwater habitats increases their susceptibility to higher exposures of pesticides, contaminants, and elevated temperature. UWR Chinook salmon exhibit an earlier time of entry into the Columbia River and estuary than other spring Chinook salmon ESUs (Meyers, Kope et al. 1998). Although most juveniles from interior spring Chinook salmon populations reach the mainstem migration corridor as yearlings, some juvenile Chinook salmon in the lower Willamette River are sub-yearlings (Friesen, Vile et al. 2004). Off-channel habitats within the Willamette Valley floodplain are particularity important for rearing fry and are actively being identified, reconnected and restored. We expect fry to be exposed to the three insecticides when applications overlap with fry occurrence and will further depress abundances of the available prey.

Based on land use and the life history of UWR Chinook salmon, we expect that this ESU will be compromised by reduced lambdas of affected populations. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Columbia River Chum Salmon

This ESU includes two remaining populations of 16 historical populations in the lower reaches (the Lower Gorge tributaries and Grays River) of the Columbia River. Thus, about 88% of the historic populations are extirpated or nearly so. In the early 1900s, the run numbered in the hundreds of thousands to a million returning adults. The size of the Lower Gorge population is estimated at 400-500 individuals, down from a historical level of greater than 8,900 (Good, Waples et al. 2005). Previous estimates of the Grays River population range from 331 to 812 individuals. However, the population increased in 2002 to as many as 10,000 individuals (Good, Waples et al. 2005). Overall, the lambda values indicate a long-term downward trend at 0.954 and 0.984, respectively.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections are hydromodification and habitat loss. Of the salmonids, chum salmon are most averse to negotiating obstacles in their migratory pathway. Thus, they are more highly impacted by the Columbia River hydropower system – specifically the Bonneville Dam (Johnson, Grant et al. 1997). The water quality in the lower Columbia River is poor. Recent USGS studies have demonstrated the presence of 25 pesticide compounds in surface waters, including carbaryl (Ebbert and Embry 2002). Although the habitat restoration project for the Grays River will likely provide some benefit to the population, we are unable to quantify the overall net effect for salmonids at this time.

Land use data indicate that the Columbia River chum salmon may be at risk of pesticide exposure. In addition to general uses, registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. The locations of high-pesticide use areas and the preferential use of river-edge habitat by chum salmon indicate that the species is at risk of pesticide exposure. The developed area surrounding the cities of Portland and Vancouver occurs along the migratory route of the Lower Gorge chum.

Columbia River chum salmon fry emerge between March and May and emigrate shortly thereafter to nearshore estuarine environments (Salo 1991). This is in sharp contrast to other salmonid behavior and indicates that chum salmon are less dependent on freshwater conditions for survival. After emergence, juvenile Columbia River chum salmon spend around 24 days feeding in the estuary. Adults return to spawn in the lower reaches of the Columbia River between the ages of two and five from mid-October through December. An average of ten days is spent in the freshwater by the spawning adults.

Given the life history of the Columbia River chum salmon, we expect that the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may lead to individual fitness level consequences. We expect that exposure will occur, resulting in both acute lethality and sublethal olfactory-mediated effects. However, we do not expect these effects to occur at a scale that would have population-level effects. As chum fry are more precocious and quickly leave natal streams, they are less reliant on the local invertebrate population. Given the life history of Columbia River chum salmon, the risk to this species' survival and recovery from the proposed stressors of the action is low for carbaryl, carbofuran, and methomyl.

Hood Canal Summer-run Chum Salmon

This ESU includes 16 historical, naturally spawned populations of summer-run chum salmon in Olympic Peninsula Rivers between Hood Canal and Dungeness Bay, Washington, as well as eight artificial propagation programs. Of the historically existing populations, seven are believed to be extirpated. Most of the extirpated populations occurred on the eastern side of the canal. Only two of the remaining populations have long-term trends above replacement; long-term lambda values of the nine existing populations range from 0.85 to 1.39 (Good, Waples et al. 2005). The Hood Canal Summer-run chum salmon populations are the subject of an intense hatchery program intended to bolster numbers. As much as 60% of the spawning populations are hatchery-raised fish.

The major threat to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections is habitat degradation. The combined effects of degraded floodplains, estuarine, and riparian habitats, along with reduced stream flow and sedimentation, have had a profound negative impact on this ESU.

The land use and environmental data indicate that the Hood Canal Summer-run chum may be exposed to carbaryl, carbofuran, and methomyl. There is no cultivated crop land and less than 6% of the ESU is developed. Washington’s 24(c) registrations for carbofuran use are not expected to be employed in this region. We do expect some exposure due to residential uses.

The Hood Canal Summer-run chum spawn from mid-September through mid-October (Tynan 1997). Emergence generally occurs from early February through mid April. Upon emerging, fry immediately commence downstream migration to estuaries (Tynan 1997). Upon arrival in the estuary, salmon fry inhabit nearshore areas in shallow water. In Puget Sound, they have been observed to reside in the top 6 inches of surface water and extremely close to the shoreline (Tynan 1997). This behavior increases the likelihood of acute exposure to drift and runoff events.

Given the life history of the Hood Canal Summer-run chum, we expect the proposed uses of carbofuran, carbaryl, and methomyl pesticide products may contaminate aquatic habitats and lead to individual fitness level consequences. We expect that exposure will occur, resulting in lethality and sublethal olfactory-mediated effects. We do not, however, expect that these effects will have population-level consequences. As chum fry are more precocious and quickly leave natal streams, they are less reliant on the local invertebrate population. Therefore, risk to this species’ survival and recovery from the stressors of the action is low for carbaryl, carbofuran, and methomyl.

Central California Coast Coho Salmon

The CCC coho salmon ESU includes 11 historical independent populations within counties from Mendocino to Santa Cruz in California. Coho populations in three larger watersheds, as well as some in smaller watersheds, have been extirpated or are nearly so. Historical escapement has been estimated between 200,000 and 500,000 fish. Current escapements are not known, though a minimum of 6,570 adult coho salmon are estimated to return to coastal streams within the ESU. Long-term population trends do not exist for any of the populations in this ESU. More fish enter northern streams but variation in abundance between cohorts can be large with one cohort often dominating. Southern streams produce few naturally spawned fish of all cohorts.

CCC coho salmon populations have been adversely affected by loss of riparian cover, elevated water temperatures, alteration of channel morphology, loss of winter habitat, and siltation. High water temperatures prevent coho salmon from inhabiting several streams within the ESU. Pesticides are expected to enter rivers as drift during application in agricultural areas. Highly contaminated runoff into the Russian River, San Francisco Bay, and into rivers south of the Golden Gate Bridge is expected during the first fall storms. We expect that application of other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters within the ESU and exacerbate adverse effects from AChE inhibition.

The majority of agricultural land use is concentrated in the Russian River watershed and watersheds south of the Golden Gate Bridge. High density urban development and urban centers occur in the San Francisco Bay and in the Russian River basin. All three a.i.s are expected to be applied to several crops within the Russian River basin and the Santa Cruz stratum during the growing season. Carbaryl is expected to be used within urban/residential areas throughout the entire year.

The movements of mature adults are influenced by stream flow; entry often occurs during the first large winter storms. Similarly, CCC coho salmon adults must wait in the lower river for sufficient flow to be able to reach spawning grounds. Fry emerge in spring and remain in the stream for up to 18 months. Newly emerged fry use backwater, side channels, and shallow channel edges. During winter, juveniles inhabit side channels, sloughs, backwater, and other protected channel features. The off-channel floodplain habitats provide feeding and growth opportunities that are important before smoltification and seaward migration in the spring. The presence of discrete brood years (BYs) makes coho salmon more vulnerable to environmental perturbations than other salmonids because a failed BY is unlikely to be replaced by other BYs.

Land use data indicate substantial overlap between high-use areas and fish runs in the Russian River, San Francisco Bay, and the Santa Cruz area. In several streams, one or more BYs are at the verge of extinction, and heavy pesticide exposure may result in the loss of a BY. We expect the proposed use of the three carbamates will lead to both individual fitness level consequences and subsequent population-level consequences. Therefore, the risk to this species' survival and recovery from the proposed action is high for carbaryl, carbofuran, and methomyl.

Lower Columbia River Coho Salmon

The LCR coho salmon ESU includes all naturally spawned coho salmon populations in streams and tributaries to the Columbia River in Washington and Oregon from the mouth of the Columbia up to and including the White Salmon and Hood rivers, and along the Willamette to Willamette Falls, Oregon. The ESU includes 26 historical populations and 25 artificial propagation programs. Over 90% of the historic populations of LCR coho salmon are considered extirpated. Most populations have very low numbers and have been replaced by hatchery production. Only two populations have a degree of natural spawning – the Sandy River and the Clackamas River. The annual population growth rates known for the Sandy River and Clackamas River are 1.102 and 1.028, respectively.

The major threats to LCR coho salmon identified in the *Status of Listed Resources* and *Environmental Baseline* sections include reduced water flow in the mainstem and estuary from irrigation diversions and hydroelectric dams. Additionally, degraded water quality in freshwater and tributary habitats negatively affects this ESU. Among the various types of habitat threats, elevated temperature, water diversions, and poor water quality have significant influence on the status of LCR coho salmon.

Pesticide use and detections in LCR coho salmon freshwater habitats are well documented. NAWQA sampling in surface waters within the ESU range detected more than 50 pesticides in streams within this ESU. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity, including carbaryl. The combined impacts from these multiple threats continue to affect this ESU.

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. Agricultural and urban development occurs along the Columbia River and the basins of its major tributaries. Both the Sandy and Clackamas Rivers drain parts of the Willamette Basin – a major agricultural area. Thus, ESU exposure to these pesticides is likely.

LCR coho salmon enter freshwater from August through December. Coho salmon spawn in November and December: emergence from the redds occurs between March and July. Fry typically select off-channel habitats associated with their natal rivers and streams to rear. The juvenile coho salmon reside in shallow freshwater habitats for more than one year. The long residence in these habitats increases their likelihood of experiencing significant exposure to pesticides and other contaminants.

Given the life history of LCR coho salmon, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may lead to both individual fitness level and subsequent population-level consequences. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

SONCC *coho salmon*

The SONCC coho salmon ESU includes coho salmon in streams between Cape Blanco, Oregon, and Punta Gorda, California, and three artificial propagation programs in the Klamath, Trinity, and Rogue Rivers. Little information on abundance trends exists for the streams within the ESU. However, available information indicates that Eel River and Southern populations are at high risk from critically low abundances. Northern populations may have larger runs. Recent estimated escapement in the Klamath River is about 2,000 fish. The Rogue River spawner run ranged from 7,800 to 12,213 coho salmon from 1999 to 2001, though many are of hatchery origin.

The threats to this ESU include road crossings and other migration barriers, timber harvest and agricultural activities, and water management. Adverse effects on the SONCC coho salmon consist of barriers that limit access to tributaries, lack of large instream woody debris, reduced riparian vegetation, and elevated water temperature. The Klamath River, one of the largest rivers, is 303(d) listed because aquatic habitat has been degraded due to excessively high water temperatures and algae blooms associated with high nutrient loads, water impoundments, and agricultural water diversions (USEPA 1993). Pesticides used in land management activities are expected to enter stream mainstems and tributaries during application as drift and as runoff.

A large portion of the agricultural and municipal land uses within this ESU is concentrated in the upper Klamath and Rogue Rivers. A diverse array of crops is produced in these areas each year. Some agriculture and smaller urban centers are dispersed on the lower alluvial coastal plains and in rivers valleys of coastal rivers. Active forest management occurs throughout the watersheds within this ESU, and application of pesticide products is anticipated.

Spawning occurs from November through January, depending on the occurrence of fall and winter storms. Fry incubate for four to eight weeks before emergence in the spring. Newly emerged fry rear in backwater, side channels, and shallow channel edges for up to 18 months. During winter, juveniles inhabit side channels, sloughs, backwater, and other protected channel features. The off-channel and floodplain habitats provide feeding and growth opportunities that are important before smoltification and seaward migration in spring. The three-year life cycle of coho salmon, with limited exchange between cohorts, makes them more vulnerable to environmental perturbations than other salmonids.

Agricultural and urban land uses overlap substantially with this ESU, particularly in the Interior Rogue and Klamath rivers. Because of this overlap, we expect that the proposed uses of carbaryl, carbofuran, and methomyl will result in both individual fitness and population-level effects. In several streams, one or more BYs are at the verge of extinction. Exposure to the three a.i.s can result in loss of weak BYs. Low population abundance levels and the discrete cohort life history of the coho salmon make the SONCC ESU more vulnerable to environmental stressors. Therefore the risk to the survival and recovery of the SONCC ESU from the stressors of the action is high.

Oregon Coast Coho Salmon

The OC coho salmon ESU includes 11 naturally spawned populations in Oregon coastal streams south of the Columbia River and north of Cape Blanco and one hatchery stock. While none of the populations have become extinct, the ESU's current abundance levels are less than 10% of historic populations. OC coho salmon abundance estimates from 2000 to 2007 ranged from a low of 51,875 to a high of 260,000 naturally produced spawners. Long-term trends in ESU productivity remain strongly negative.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections are habitat degradation from logging, road construction, urban development, mining, agriculture, and recreation. Within the various types of habitat, elevated temperature, water diversions, and poor water quality also affect the status of OC coho salmon. The combined impacts from these multiple threats continue to affect OC coho salmon.

Crop rotation patterns and crop types influence the distribution and frequency of pesticides within an area. Based on limited agricultural activities, we expect that carbaryl and existing stocks of carbofuran may be applied on a limited scale throughout the growing season. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. We also expect carbaryl use within urban and residential areas throughout the entire year. However, we expect no to very low applications of methomyl throughout the growing season. Although carbaryl and carbofuran may be applied to pine seedlings, we expect no to low applications in these forests. As runoff from urban and agricultural areas may drain into adjacent streams, ESU exposure to these pesticides is likely. However, given the above land uses and expected low application of the three carbamates, we expect lower incidences of fish exposure to these compounds in surface waters following application.

OC coho salmon enter rivers in September or October; spawning occurs in December. Fry emerge between March and July, then move to shallow areas near the stream banks. Juvenile coho salmon are often found in small streams less than five ft wide, and may migrate considerable distances to rear in lakes and off-channel ponds. Generally, coho salmon spend 18 months rearing in freshwater before moving out into the ocean. Given this duration spent in shallow freshwater habitats, they are more likely to experience higher pesticide exposure and contaminants.

Given the limited applications of the three carbamates and low incidences within land uses that overlap with the range of OC coho, exposure is possible, although not likely to be extensive. We do not expect that exposure will have effects at the population level. Therefore, the risk to this species' survival and recovery from the stressors of the action is low for carbaryl, carbofuran, and methomyl.

Ozette Lake Sockeye Salmon

This ESU is made up of only one historic population. Natural spawning aggregations remain on two beaches of Ozette Lake. Two tributary spawning groups were initiated in1992 through hatchery programs. Peak run size in the 1940s has been estimated to be between 3,000 and 18,000 fish, and actual production (*i.e.*, including harvest) may have been as high as 50,000. Recent estimates put the population at 3,600 – 4,600 individuals (Haggerty, Ritchie et al. 2007). The supplemental hatchery program began with out-of-basin stocks and make up an average of 10% of the run. The proportion of beach spawners originating from the hatchery is unknown but likely low. Uncertainty in past population counts coupled with poorly documented historical abundance prevents calculation of population growth rates and trends.

Major threats to this population identified in the *Status of Listed Resources* and *Environmental Baseline* sections are siltation of spawning habitat from logging activities within the watershed and genetic effects from past interbreeding with kokanee. Almost 80% of the land cover for this

ESU is evergreen forest. Between 1940 and 1984, 85% of the basin was clear-cut logged (Blum 1988). Roughly 77% of the land in Ozette Basin is managed for timber production (Jacobs, Larson et al. 1996). The extent to which pesticide products are currently used by these companies is unknown.

Ozette Lake is in a sparsely populated area, with less than 1% of land developed. No crop land was identified in NLCD data (Table 32). The land use and environmental data indicate that the Ozette Lake sockeye salmon may be exposed to carbaryl or carbofuran if applied in the watershed. Carbaryl is registered for a number of non-agricultural applications which may take place in forested and residential areas. Additionally, the area has a small population, making residential use of carbaryl within the ESU a possibility. We do not expect 24(c) registrations of carbofuran will be used within this ESU, though it may be applied to pine seedlings. However, there are few data available on use and no monitoring data are currently available. We do not expect methomyl to be used within the boundaries of this ESU.

Ozette Lake sockeye salmon enter the lake between April and August, and spawning occurs late October through February. Fry emerge from gravel redds in the spring and emigrate to the open waters of the lake where they remain for a full year. They then smolt as 1-year olds and migrate to the open ocean. The majority of Ozette Lake sockeye salmon return to spawn as four-year old fish after spending two full years at sea.

Given the life history of the Ozette Lake sockeye, we expect that that the proposed uses of carbaryl and carbofuran pesticide products may contaminate aquatic habitats used by sockeye in a way that might lead to individual fitness level consequences. While the uses of these materials may lead to some overlap with the Ozette Lake sockeye, the existing and likely future land uses should limit the applications of pesticides containing the three carbamates. Consequently, the risk posed by the proposed action to Ozette Lake sockeye salmon's survival and recovery is low for carbaryl, carbofuran, and methomyl.

Snake River Sockeye Salmon

The SR sockeye salmon ESU historically includes populations in five Idaho lakes as well as artificially propagated sockeye salmon from the Redfish Lake Captive Broodstock Program. Only one hatchery-sustained population remains and is found in Redfish Lake. This population is listed as endangered and has an extremely high risk of extinction. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely greater than 0.3% (Hebdon, Kline et al. 2004). No natural origin adults have returned to Redfish Lake to spawn since 1998; the population is maintained entirely by propagation efforts. Around 30 fish of hatchery origin return to spawn each year (FCRPS 2008).

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include impaired tributary flow and passage, migration barriers, degraded water quality, and hydromodification of the Columbia and Snake Rivers. Like the Ozette Lake ESU, the SR sockeye occupy a relatively undeveloped area with very little cropland (Table 37) However, the SR sockeye have the longest migration of any sockeye salmon, traveling 900 miles inland. These waters are contaminated by drift and runoff from both agricultural and urban areas. Exposure during migration likely adds to the low survivorship of smolts. The land use and environmental data indicate that the SR sockeye may be exposed to carbaryl, carbofuran, and methomyl during migration. Registered 24(c) uses within this ESU include carbofuran application to potatoes and spinach grown for seed in Washington.

Historically, sockeye salmon entered the Columbia River system in June and July, and arrived at Redfish Lake between August and September (FCRPS 2008). Spawning occurred in lakeshore gravel and generally peaked in October. Fry emerged in the spring (April and May) then migrated to open waters of the lake to feed. Juvenile sockeye remained in the lake for one to three years before migrating through the Snake and Columbia Rivers to the ocean. Adult sockeye spent two or three years in the open ocean before returning to Redfish Lake to spawn.

During adult and juvenile migrations the sockeye are at their greatest risk of exposure to the stressors of the action. Sockeye salmon making the 900 mile journey each way pass along many miles where agricultural crops are at the river's edge. Drift and runoff occurring in conjunction with sockeye salmon migration is expected to cause adverse effect.

Given the life history of the SR sockeye, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may lead to individual fitness level consequences. However, we do not expect fry to be exposed to the a.i.s, although some exposure may occur during adult migration and lead to acute lethality or temporary AChE inhibition. However, this exposure is not expected to occur at a frequency that would cause effects at the population level. Therefore, the risk to this species' survival and recovery from the stressors of the action is low for carbaryl, carbofuran, and methomyl.

Central California Coast steelhead

The CCC steelhead DPS includes all naturally spawned steelhead in streams from the Russian River south to Aptos Creek. This area includes streams entering the San Francisco Bay. The DPS consists of nine historic independent populations. There is limited information on the abundance of these populations, but all are in severe decline.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are dams and other migration barriers, urbanization and channel modification, agricultural activities, predators, hatcheries, and water diversions. Throughout the species' range, habitat conditions and quality have been degraded by a lack of channel complexity, eroded banks, turbid and contaminated water, low summer flow and high water temperatures, an array of contaminants found at toxic levels, and restricted access to cooler head waters from migration barriers. There is limited monitoring data from streams within this DPS. The combined impacts of these threats continue to affect the CCC steelhead.

Crop farming is concentrated in low laying areas and floodplains along the estuaries and stream valleys of the Russian River and Santa Cruz drainage basins. Christmas trees production occurs in these areas in addition to a variety of food crops. Carbaryl and methomyl are registered for a variety crops grown in the Russian River valley and in the San Mateo and Santa Cruz counties. Registered 24(c) uses for carbaryl in California include application to fruits and nuts, prickly pear cactus, ornamental plants, and non-food crops. Methomyl has 24(c) registrations for insect control on ornamentals, beans, soybeans, radishes, sweet potatoes, Chinese broccoli, broccoli raab, and pumpkins. The a.i.s are therefore expected to be used throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Further, 16% of the DPS' range consists of developed urban and residential land, mostly within the San Francisco Bay and Russian River basin. Carbaryl is expected to be used the entire year within the urban/residential areas.

The CCC steelhead populations are all of the winter-type. Fry emerge in spring and rear in smaller tributaries and off-channel habitats. During periods with high flows, they move into floodplain habitat. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to smolt.

The registered uses of the three carbamates indicate substantial overlap with the CCC steelhead populations, especially in the San Francisco Bay, Russian River, and Santa Cruz area. We expect that the proposed uses of the three carbamates may contaminate spawning and rearing habitats and lead to both individual fitness level and population -level consequences. Thus, the risk to this species' survival and recovery from the stressors of the proposed action is high for carbaryl, carbofuran, and methomyl.

California Central Valley Steelhead

The CCV steelhead DPS includes all naturally spawned steelhead in the Sacramento River, San Joaquin River, and their tributaries. This area includes streams entering the Sacramento-San Joaquin Delta (Delta) east of Chipps Island. All populations rear and migrate through the Sacramento River, San Joaquin River, Delta, and San Francisco Bay. The current distribution is severely reduced and fragmented compared to historical distributions. About 6,000 river miles of river habitat have been reduced to 300 miles. Historical returns within the DPS may have approached two million adults annually. Current annual run size for the entire Sacramento-San Joaquin system today is estimated at less than 10,000 returning adults.

The *Status of Listed Resources* and *Environmental Baseline* sections indicate that dams and other migration barriers, urbanization and channel modification, agricultural activities, introduced non-native predators, hatcheries, and large scale water management and diversions negatively affect this DPS. Steelhead habitat has been highly degraded by reduced channel complexity, eroded banks, increased water temperature, migration barriers restricting access to cooler head waters, and decreased water quality from contaminants. Numerous NAWQA, CDPR, and other assessments found high concentration of contaminants in both the San Joaquin and Sacramento Rivers and their tributaries. In the San Joaquin Basin, seven pesticides exceeded EPA criteria for aquatic life. These pesticides include diuron, trifluralin, azinphos-methyl, carbaryl, chlorpyrifos, diazinon, and malathion. The combined impacts from these threats continue to affect the CCV steelhead.

Approximately 27% of the land within the DPS is developed for cultivation of crops (Table 38). High densities of crop farming occur throughout the San Joaquin Basin, in the Sacramento-San Joaquin Delta, and along lower Sacramento River. Further, 9.2% of the DPS consists of urban development. Based on the crop types, we expect carbaryl, carbofuran, and methomyl are commonly applied in the Central Valley throughout the growing season. In urban and residential areas, we expect carbaryl to be applied throughout the year. Monitoring in the San Joaquin basin found 48 of 83 pesticides tested for. Seven pesticides (diuron, trifluralin, azinphos-methyl, carbaryl, chlorpyrifos, diazinon, and malathion) exceeded criteria for the protection of aquatic life. Pesticides detected in the Sacramento River included thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Carbofuran was most often detected in agricultural sites while carbaryl was also found in all urban sites tested.

All of the steelhead populations within this DPS exhibit the winter-type life history, though detailed information about the CCV steelhead life history is not available. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to enter the ocean. The CCV steelhead use the lower Sacramento River and the Delta for rearing and as migration corridor. Some may utilize tidal marshes, non-tidal freshwater marshes, and other shallow areas in the Delta for rearing areas for short periods during outmigration to the ocean.

The registered uses of the three carbamates indicate substantial overlap with the CCV steelhead populations, especially in the lower Sacramento River and Delta. We expect that the proposed uses may contaminate spawning and rearing habitats and lead to both individual fitness and population-level consequences. Thus, the risk to this species' survival and recovery from the stressors of the proposed action is high for carbaryl, carbofuran, and methomyl.

Lower Columbia River Steelhead

The LCR Steelhead DPS includes 23 historical, naturally-spawned steelhead populations in Columbia River tributaries on the Washington side between the Cowlitz and Wind Rivers in Washington and on the Oregon side between the Willamette and Hood Rivers. Historical counts from the Cowlitz, Kalama, and Sandy Rivers suggest the ESU probably exceeded 20,000 fish. During the 1990s, fish abundance dropped to 1,000 to 2,000 fish. Many of the populations in this DPS are small, and the long- and short-term trends in abundance of all individual populations are negative. The annual population growth rate known for nine independent populations ranged from 0.945 to 1.06.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections include dams; water diversion; destruction or degradation of riparian habitat; and land use practices (logging, agriculture, and urbanization). Tributary hydropower development has created barriers and reduced fish access, ultimately affecting the spatial structure within the independent populations. Within the various types of habitat, water diversions, elevated temperature, and poor water quality also affect the status of LCR steelhead. Pesticides have also been detected in LCR steelhead habitats. NAWQA sampling from 1991-1995 in surface waters within the DPS range detected more than 50 pesticides in streams. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity. These pesticides include carbaryl, carbofuran, atrazine, chlorpyrifos, and malathion. The combined impacts from these multiple threats continue to affect this DPS.

Most of the highly developed land and agricultural areas in this DPS's range are adjacent to salmonid habitat. Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. Based on the crop types, we expect that carbaryl and methomyl are commonly applied in the Lewis River subbasin throughout the growing season and within urban and residential areas throughout the entire year for carbaryl. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Thus, DPS exposure to these pesticides is likely.

This DPS includes winter- and summer-run types. Summer-run steelhead return to freshwater between May and November. They enter the Columbia River in a sexually immature condition and require several months in freshwater before spawning. Winter-run steelhead enter freshwater from November to April. These fish are close to sexual maturation and spawn shortly after arrival in their natal streams. Steelhead fry typically rear in off-channel habitats associated with their natal rivers and streams for more than a year. Given this duration, juveniles are likely to experience pesticide exposure.

Given the life history of LCR steelhead, we expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products may contaminate rearing habitats and lead to individual fitness and subsequent population-level consequences. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Middle Columbia River Steelhead

The MCR steelhead DPS includes 19 populations in Oregon and Washington subbasins upstream of the Hood and Wind River systems to and including the Yakima River. Historical run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby, Wainwright et al. 1996), whereas only 1,000 – 4,000 currently spawn. The most recent 10-year period indicated trends in abundance were positive for approximately half of the independent populations and negative for the remainder. Growth rates ranged from 0.97 to 1.02. Two historical populations are considered extinct.

Agricultural development is high within the range of MCR steelhead. Based on crop types and allowable uses, we expect that carbaryl and methomyl are commonly applied in the Yakima River basin, Walla Walla River basin, Deschutes River basin, and the John Day River basin throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. In urban and residential areas, carbaryl will likely be applied throughout the entire year. Registered 24(c) uses within this ESU include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons in Oregon and to potatoes and spinach grown for seed in Washington. Given the high amount of agriculture in these areas, we also expect other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl and exacerbate adverse effects from AChE inhibition. Thus, DPS exposure to these pesticides is expected to be high

Mature adults (three to five years old) may enter rivers any month of the year and spawn in late winter or spring. Swim–up fry usually inhabit shallow water along banks of streams or aquatic habitats on stream margins. Steelhead rear in a variety of freshwater habitats and most remain in freshwater for two to three years. Some individuals, however, have stayed for as many as six to seven years. Most MCR steelhead smolt at two years and spend one to two years in the ocean prior to re-entering the freshwater to spawn.

Intense agricultural development has a high degree of overlap with the spawning and rearing areas of the MCR steelhead DPS. Given the long residency periods in freshwater, we expect that the proposed uses of of the three carbamates may lead to both individual fitness and population-level consequences. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Northern California Steelhead

The NC steelhead DPS includes all naturally spawned steelhead in California coastal river basins from Redwood Creek southward to, but not including, the Russian River, as well as two artificial propagation programs. Historical estimates of annual production for this DPS are upwards of 200,000 fish. Exact information on abundance is lacking for most of the streams within the DPS, though most populations are in decline. The DPS includes 15 historically independent populations of winter steelhead and 10 populations of summer steelhead. None of the winter steelhead populations are viable due to low abundance and production.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are timber harvest, forest roads, vineyard developments, and road crossings and dams. Stressors to the NC steelhead include lack of large instream woody debris, reduced riparian vegetation, elevated water temperature, increased predation, and barriers that limit access to tributaries. The combined impacts from these multiple threats continue to affect the NC steelhead.

About 85% of the land use for this DPS consists of forests and chaparral (Table 26). Atlhough carbaryl may be applied in forested areas, the extent of exposure is unknown. There are areas of Mendocino County with a high density of vineyards, and there is some crop farming in low laying areas and floodplains along the estuaries and lower reaches of coastal streams. Coastal communities are located at the mouths of several streams within the NC steelhead DPS. Based on the crop types, we expect carbaryl, carbofuran, and methomyl to be commonly applied throughout the growing season in these areas. Carbaryl is expected to be used the entire year within urban and residential areas. However, because agricultural and urban development is not concentrated in any one basin, the risk of exposure to any given population is decreased. Additionally, the majority of steelhead juveniles rear at higher elevations in the watersheds with less agricultural influence. The fry, then, are less likely to be exposed during critical life stages.

The winter-type steelhead enters rivers as mature adults between November and April to spawn. The summer-type enters the stream in immature condition between May and October. Early arriving summer steelehead move high into the upper watersheds where they hold in deep pools throughout the summer before spawning in fall. In northern California, juvenile steelhead remain in freshwater for two or more years before migrating downstream to smolt. Juvenile steelheads spend a variable amount of time in the estuary before entering the open ocean.

We expect the proposed uses of carbaryl, carbofuran, and methomyl pesticides products within this DPS may lead to individual fitness effects, ranging from acute lethality to temporary AChE inhibition. However, exposure of NC steelhead populations to the three carbamates will be limited given the amount of overlap between agriculture, residential and urban development and spawning and rearing habitat. As such, we do not expect exposure to result in population-level consequences. Therefore, the risk to the survival and recovery of NC steelhead from the proposed action is low for carbaryl, carbofuran, and methomyl.

Puget Sound Steelhead

The Puget Sound steelhead is comprised of 21 populations, some of which have both summer and winter runs. Steelhead occur in all major watersheds within the Sound and out to the Elwha River. Of the winter runs, 17 had declining and four had increasing population trends. Summer run abundance trends are not available. Estimated run size for Puget Sound steelhead in the early 1980s is approximately 100,000 winter-run steelhead and 20,000 summer-run steelhead.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are habitat degradation from logging, road construction, urban development, mining, agriculture, and recreation; water diversions; and poor water quality. In particular, elevated temperature, water diversions, and poor water quality have the most significant influences on the status of Puget Sound steelhead. Furthermore, there has been extensive urbanization in this region. Well over two million people inhabit the area, with most development occurring along rivers and coastline.

Pesticide use and detections in the DPS's watershed are also well documented. 2006 NAWQA sampling in the Puget Sound basin detected 26 pesticides and 74 other synthetic organic chemicals in streams and rivers, with different mixtures of chemicals linked to agricultural and urban settings. Urban streams sampled in Puget Sound showed the highest detections for carbaryl, diazinon, and malathion. Diazinon was also frequently detected in urban streams at concentrations that exceeded EPA guidelines for protecting aquatic life (Bortleson and Ebbert 2000).

Registered uses of carbaryl, carbofuran, and methomyl include applications to crop agricultural sites, residential sites, and urban sites. In addition, Washington has 24(c) registrations for carbofuran use on spinach grown for seed and potatoes. Land classes indicate high use of the three a.i.s in the following drainage basins: Nooksack, Skagit River, Stillaguamish, Skykomish, Snoqualimie, Snohomish, Lake Washington, Duwamish, and Puyallup. Roughly 7% of the lowland areas (below 1,000 ft elevation) in the Puget Sound region are covered by impervious surfaces, which increase urban runoff containing pollutants and contaminants into streams. Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, PBDEs, PAHs, pharmaceuticals, nutrients, and sediments. Thus, we expect salmonid populations within the ESU to be exposed to carbaryl, carbofuran, and methomyl. Further, monitoring data shows that we can expect concurrent exposure with other AChE inhibiting chemicals, leading to a pronounced severity of effects.

Summer steelhead enter freshwater between May and October, while winter steelhead enter between November and April. Spawning generally occurs in late winter or spring. Immediately after leaving the gravel, swim-up fry usually inhabit shallow water along banks of stream or aquatic habitats on stream margins. Steelhead rear in a wide variety of freshwater habitats, generally for two to three years. However, they may possibly reside up to six to seven years in freshwater environments. Following the rearing period, they smolt and migrate to sea in the spring.

We expect the proposed uses of carbaryl, carbofuran, and methomyl pesticide products that contaminate aquatic habitats may lead to both individual fitness level and subsequent population-level consequences. The risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Snake River Basin Steelhead

The SR steelhead DPS includes 23 naturally spawned populations below impassable natural and man-made barriers in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the U.S.-Canada border. The Snake River supports about 63% of the natural-origin production of steelhead in the Columbia River Basin. The 10-year average for natural-origin steelhead passing Lower Granite Dam between 1996 and 2005 is 28,303 adults. Annual population growth rates show mixed long- and short-term trends in abundance and productivity. The annual population growth rate is known for eight independent populations and range 0.89 to 1.08. One historical population is likely extirpated.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections include hydrosystem mortality, water diversions, excessive sediment, and degraded water quality. Elevated temperature also impacts the status of SR steelhead. Pesticides have been detected in SR steelhead freshwater habitats. NAWQA sampling in 1992-1995 in the DPS's watersheds detected Eptam, atrazine, desethylatrazine, metolachlor, and alachlor. Carbaryl and carbofuran were detected in only 1% of samples. The combined impacts from these multiple threats continue to affect SR steelhead.

Some spawning and rearing habitat of the Snake River steelhead is in forested areas, where carbaryl may be applied. However, the likelihood of exposure in forested areas is unknown. Registered 24(c) uses within this ESU include carbofuran application to potatoes and spinach grown for seed in Washington. Dryland agriculture occurs in lower Clearwater Basin, with crops including wheat, peas, and lentils. Alfalfa, hay, and grasses are also grown in the lower Clearwater as well as the lower Salmon River basin. All three a.i.s are registered for use on alfalfa. Agricultural activities and urban communities are concentrated along the Snake River and near the mouths of major tributary valleys, thus stream water quality and biological communities in the downstream portion of the upper Snake River basin are degraded. Given that urban and agricultural areas are located adjacent to streams in the Lower Clearwater River drainage, DPS exposure to these pesticides is likely.

Sexually immature adult Snake River summer steelheads enter the Columbia River from late June to October. Adults migrate upriver until they reach Snake River tributaries where they spawn between March and May of the following year. Emergence occurs by early June in low elevation streams and as late mid-July at higher elevations. After hatching, juvenile SR steelhead typically select off-channel habitats associated with their natal rivers and streams for rearing. They spend two to three years in freshwater before they smolt and migrate to the ocean. Juvenile steelhead are more likely to be exposed to pesticides and other contaminants because of their long freshwater residency period.

Given the life history of SR steelhead, we expect the proposed uses of carbaryl and carbofuran pesticide products that contaminate aquatic habitats will lead to individual fitness level consequences and subsequent population-level consequences. The uses of these materials in some natal areas and along migratory routes may cause acute lethality, temporary AChE inhibition, and reduced growth. We expect that exposure to methomyl may lead to individual fitness consequences, but not to an extent that would affect population growth rates. The risk to SR steelhead survival and recovery from the stressors of the action is high for carbaryl and carbofuran, but low for methomyl.

South-Central California Coast Steelhead

The S-CCC steelhead DPS includes all naturally spawned steelhead in streams from the Pajaro River to the Santa Maria River. The major basins in the S-CCC steelhead range are the Pajaro River system and the Salinas River system. Carmel River, a smaller basin, also supports a stable run of steelhead. Historic abundance estimates for the DPS imply an annual return may have been upwards of 20,000 fish. The estimated production in five of the major rivers indicates a return of less than 500 adults.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are dams and other migration barriers, urbanization and channel modification, agricultural activities, and wildfires. These activities has resulted in reduced channel complexity, eroded banks, increased water temperature, migration barriers restricting access to cooler head waters, and decreased water quality from contaminants.

About 7% of the DPS is developed for cultivation of crops; a large portion of this land is in the Salinas River valley. Crops are concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams. Based on the crop types, we expect carbaryl, carbofuran, and methomyl are commonly applied within the DPS throughout the growing season. Approximately 8% of the DPS has been developed for urban and residential purposes. Developed areas are located at the mouth of several streams within the S-CCC steelhead DPS. We expect carbaryl use in urban/residential areas throughout the entire year.

All S-CCC steelhead are a winter-run life history. Juvenile steelhead remain in freshwater for one or more years before migrating downstream to smolt. Steelhead in this area consists of two life history groups: one where the juveniles rear for one or two years mainly in freshwater and another where the juveniles rear at the upper end of coastal lagoons for the first or second summer.

In conclusion, based on the long freshwater residence time of steelhead and the considerable overlap between S-CCC steelhead distribution and expected pesticide use, we expect that the S-CCC may experience high levels of exposure. We expect that proposed uses of the three carbamates that contaminate spawning and rearing habitats may lead to both individual fitness and population-level consequences. Therefore, the risk to the survival and recovery of S-CCC from the proposed action is high for carbaryl, carbofuran, and methomyl.

Southern California Steelhead

The SC steelhead DPS includes populations in five major and several small coastal river basins in California from the Santa Maria River to the U.S. – Mexican border. It is estimated that the species' current distribution constitutes about 1% of the historical distribution. Current abundance is considerably reduced with an estimated escapement of 500 fish for four of the larger rivers. Long-term estimates and population trends are lacking for the streams within the DPS.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections are dams and other migration barriers, urbanization and channel modification, agricultural activities, and wildfires. As a result of these activities, the stream substrate contains a high proportion of fines, stream channels lack complexity, banks are eroding, migration barriers restrict fish access to cooler head waters and tributaries, and the water is turbid and contaminated.

Agriculture is concentrated in low laying areas and floodplains along the estuaries and lower reaches of streams. Carbaryl, carbofuran, and methomyl are expected to be commonly used for several crops during the growing season. Large areas of dense development exist in the counties of Santa Barbra to San Diego. Carbaryl is expected to be used within urban/residential areas throughout the entire year. The NAWQA analysis detected more than 58 pesticides in ground and surface waters within the heavily populated Santa Ana basin, including multiple AChE inhibitors. The invertebrate community in the basin is heavily altered by pesticide influence. Carbaryl was detected in 42% or urban samples but few detections exceeded standards for protection of aquatic life. Carbofuran was also detected but not in levels exceeding standards. The application of other AChE inhibiting pesticides is expect to co-occur with the a.i.s, and exacerbate adverse effects from AChE inhibition.

All steelhead populations in this DPS are winter-type. Juvenile steelheads remain in freshwater for one or more years before migrating downstream to smolt. SC steelhead consists of two life history groups: one where the juveniles rear for one or two years mainly in freshwater and another where the juveniles rear at the upper end of coastal lagoons for the first or second summer.

We expect the proposed uses of carbaryl, carbofuran, and methomyl pesticides products that contaminate aquatic habitat will lead to both individual fitness consequences and subsequent population-level consequences. There is substantial overlap between urban and agricultural areas and the spawning and rearing habitats of the SC steelhead populations. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Upper Columbia River Steelhead

The UCR steelhead DPS includes all naturally spawned populations below natural and man-made impassable barriers in the Columbia River basin upstream from the Yakima River, Washington, to the U.S.-Canada border. The DPS is comprised of four naturally-spawned populations and six artificial propagation programs in Washington State. The historical independent populations are located within the Wenatchee, Entiat, Methow, and Okanogan river systems. Abundance data indicate that all populations are below the minimum threshold for recovery. The annual population growth rates range between 1.067 and 1.086. Overall adult returns are dominated by hatchery fish.

The major threats to this DPS identified in the *Status of Listed Resources* and *Environmental Baseline* sections include dams that block fish migration and alter river hydrology; water diversions; destruction or degradation of riparian habitat; and land use practices (logging, agriculture, urbanization). Elevated water temperature and poor water quality also affect the status of UCR steelhead. Pesticides have also been detected in UCR steelhead freshwater habitats. NAWQA sampling from 1992-1995 in the Central Columbia Plateau detected numerous pesticides in surface water (Williamson et al. 1998). Carbaryl was detected in 5% of samples, and carbofuran was detected in 6%. Methomyl was not detected. While detections of these three chemicals did not exceed water quality standards, concentrations of six other pesticides exceeded EPA criteria for the protection of aquatic life. The co-occurrence of atrazine with carbaryl and other OPs in aquatic habitats increases the likelihood of adverse responses in salmonids and their aquatic prey. The combined impacts from these multiple threats continue to affect UCR steelhead.

Land uses within this DPS range include agricultural, urban and residential development. Based on the crop types, we expect that carbaryl and methomyl are commonly applied in the Wenatchee, Methow, and Entiat subbasins throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. As runoff containing pesticides and other contaminants from the above land uses drain into the Wenatchee, Entiat, and Methow Rivers, DPS exposure to these pesticides is likely. We also expect other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters of the Wenatchee, Entiat, Methow, and Lower Crab Creek subbasins and exacerbate adverse effects from AChE inhibition.

UCR adults return to the Columbia River in the late summer and early fall. UCR steelhead spawn and rear in the major tributaries to the Columbia River between Rock Island and Chief Joseph dams. Adults reach spawning areas in late spring of the calendar year following entry into the river. Newly emerged fry move about considerably and seek suitable rearing habitat, such as stream margins or cascades. Steelhead fry typically select off-channel habitats associated with their natal rivers and stream for extended periods of rearing. Fry move downstream in the fall in search of suitable overwintering habitat (Chapman, Hillman et al.

1994). Most juvenile steelhead spend two or three years in freshwater before migrating to the ocean. Some juvenile steelhead may spend up to seven years rearing in freshwater before migrating to sea. Given the long duration in shallow freshwater habitats, juveniles are more likely to experience higher pesticide exposure, contaminants, and elevated temperature.

We expect that the proposed uses of carbaryl, carbofuran, and methomyl may contaminate the rearing habitat of UCR steelhead and lead to individual fitness and population-level effects. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Upper Willamette River Steelhead

The UWR steelhead DPS includes all naturally spawned late-fall populations below natural and man-made impassable barriers in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River. The DPS is comprised of four historical populations. Steelhead populations within this DPS have been declining, on average, since 1971 and have exhibited large fluctuations in abundance. Long-term trends in the annual population growth rate are less than one. The annual population growth rate of the four independent populations ranged from 0.97 to 1.023.

The major threats to this ESU identified in the *Status of Listed Resources* and *Environmental Baseline* sections include habitat loss due to blockages from hydroelectric dams and irrigation diversions, and degraded water quality within the Willamette mainstem and the lower reaches of its tributaries. Additionally, pesticide use and detections in UWR steelhead freshwater habitats are well documented. Fifty pesticides were detected in streams that drain both agricultural and urban areas. Forty-nine pesticides were detected in streams draining agricultural land, while 25 pesticides were detected in streams draining urban areas. Ten of these pesticides, including carbaryl and carbofuran, exceeded EPA criteria for the protection of freshwater aquatic life for chronic toxicity (Wentz, Bonn et al. 1998). The combined impacts from these multiple threats continue to affect UWR steelhead.

Based on the crop types, we expect that carbaryl and methomyl are commonly applied in the Willamette Valley throughout the growing season. Existing stocks of carbofuran are also expected to be applied throughout the growing season. Registered 24(c) uses in Oregon include carbofuran application to potatoes, nursery stock, sugar beets, and watermelons. The Willamette Basin is the largest agricultural area in Oregon. In 1992 the Willamette Basin accounted for 51% of Oregon's total gross farm sales and 58% of Oregon's crop sales. About one-third of the agricultural land is irrigated and most of it is adjacent to the mainstem Willamette River. Urban developments are also located primarily in the valley along the mainstem Willamette River. We expect carbaryl to be used throughout the year in urban and residential areas. Given that major urban and agricultural areas are located adjacent to the mainstem Willamette, DPS exposure to these pesticides is likely. We also expect that other AChE inhibiting pesticides will co-occur with carbaryl, carbofuran, and methomyl in the waters of the Willamette Valley and exacerbate adverse effects from AChE inhibition.

UWR steelhead enter the Willamette River in January and February and ascend to their spawning areas in late March or April. After emergence, steelhead fry typically rear in off-channel habitats associated with their natal rivers and streams for two to three years. Smolt migration past Willamette Falls begins in early April and extends through early June, with peak migration in early to mid-May. Most spend two years in the ocean before re-entering freshwater to spawn.

Much of the Willamette Valley has been converted to agricultural purposes. Based on the co-occurrence of high use land classes with spawning and rearing habitats, we expect that the proposed uses of the three carbamates may contaminate spawning and rearing habitats and lead to both individual fitness and population-level effects. Therefore, the risk to this species' survival and recovery from the stressors of the action is high for carbaryl, carbofuran, and methomyl.

Summary of Species-Level Effects

In the preceding section NMFS described expected population-level effects in terms of reductions in annual growth rate, productivity (reproduction), and abundance (numbers of salmonids). We concluded that all but Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon populations will likely show reductions in viability due to the reregistration of carbaryl and carbofuran. We also concluded that all but Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, Snake River fall-run Chinook salmon, Snake River spring-summer-run Chinook salmon, California Coastal Chinook salmon, and Snake River steelhead populations will likely show reductions in viability due to the reregistration of methomyl. The effects of EPA's proposed action are first manifested at the individual-level where reductions in individual fitness are expected. We showed that an individual's survival, migration, and growth are all significantly reduced by the proposed action. We also showed that these reductions are likely intensified by co-occurring stressors in the action area including the presence of other carbamates and OP insecticides in the action area.

Therefore, given the severity of expected changes in the annual population growth rate for affected populations, it is likely that California Coastal Chinook salmon, Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Central California Coast coho salmon, Southern Oregon and Northern Coastal California coho salmon, Central California Coast steelhead, California Central Valley steelhead, LCR steelhead, MCR steelhead, Snake River Basin steelhead, South-Central California Coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead will experience reductions in viability, which ultimately reduces the likelihood of survival and recovery of these species. The Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon will not likely experience reductions in viability.

Effects of the Proposed Action to Designated Critical Habitat

NMFS' critical habitat analysis determines whether the proposed action will destroy or adversely modify critical habitat for ESA-listed species by examining any change in the conservation value of the essential features of critical habitat. Our analysis does not rely on the regulatory definition of 'adverse modification or destruction' of critical habitat. Instead, this analysis focuses on statutory provisions of the ESA, including those in Section 3 that define "critical habitat" and "conservation," those in Section 4 that describe the designation process, and those in Section 7 setting forth the substantive protections and procedural aspects of consultation.

NMFS has designated critical habitat for all listed Pacific salmonids except for LCR coho salmon and Puget Sound steelhead. The action area encompasses all designated critical habitat areas considered in this Opinion. The PCEs for each listed species, where they have been designated, are described in the *Status of Listed Resources* section of this Opinion. The PCEs identify those physical or biological features that are essential to the conservation of the species that may require special management considerations or protections. As the species addressed in this Opinion have similar life history characteristics, they share many of the same PCEs. These PCEs include sites essential to support one or more life stages (sites for spawning, rearing, migration, and foraging) and contain physical or biological features essential to the conservation of the ESU/DPS, such as:

1. freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development;
2. freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks;
3. freshwater migration corridors free of obstruction, along with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival;
4. estuarine areas free of obstruction, along with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation;
5. nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and

natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and

6. offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

At the time that each habitat area was designated as critical habitat, that area contained one or more PCEs within the acceptable range of values required to support the biological processes for which the species use that habitat. We identified PCEs of prey availability and water quality in freshwater spawning sites, freshwater rearing sites, migratory corridors, and estuarine and nearshore marine areas as susceptible to the effects of the proposed action. We evaluated whether prey availability and water quality are likely affected by the proposed action and formulated risk hypotheses for these PCEs. We relied upon pesticide use information (see the *Description of the Action* section), surface water monitoring data, EPA modeling estimates, and NMFS modeling estimates (see *Exposure Analysis* section) to determine likely aquatic concentrations. Based on our analysis within the *Effects of the Proposed Action* section, the proposed action will reduce prey availability and degrade water quality within freshwater spawning habitat, freshwater rearing habitat, and migration corridors.

Direct exposure to carbaryl, carbofuran, and methomyl and the other chemical stressors of the action within freshwater will have an effect on Pacific salmonid critical habitat that overlaps with intense agricultural and residential/urban land uses. The *Environmental Baseline* discusses the extent of anthropogenic alteration of salmonid habitat within the action area. As established in *Effects of the Action to Listed Species* Section, agricultural, urban, and residential areas overlap with salmonids' geographic range to differing degrees between ESUs/DPSs. As noted in the *Effects of the Proposed Action* section, pesticides most often occur in the aquatic environment as mixtures. Carbaryl, carbofuran, and methomyl are found in environmental mixtures. Based on evidence of additive and synergistic effects of these compounds, we expect mortality of large numbers and types of aquatic insects, which are prey items for salmon. A reduction in salmonid prey abundance poses subsequent impacts on the overall growth of salmonid juveniles, especially during their first year of survival.

Reduction in Prey

Freshwater spawning, rearing, and migratory habitats must provide forage areas that support juvenile development. A reduction in the abundance of prey items will decrease the conservation value of these habitats, as they will support fewer individuals, especially during a salmonid's first year of survival. Given the environmental baseline conditions of the aquatic systems, the existing invertebrate community is already depauperate in many of these areas and requires longer periods for recovery following each pesticide application event. The a.i.s in surface runoff and pesticide drift entering freshwater rearing habitats will exacerbate reductions in insect communities that are salmonid prey items in these degraded systems. We expect that designated critical habitats that support juvenile feeding and growth will be contaminated by the proposed uses of carbaryl, carbofuran, and methomyl to the extent that the habitat is precluded from serving its intended role in the survival and recovery of listed salmonids. The overall conservation value of designated critical habitat, whether low, medium, or high, will not meet its intended role to support the survival and recovery of the species.

Reduction in Water Quality

Drift and runoff from areas of intensive urban and agricultural development will likely contain carbaryl, carbofuran, and methomyl in addition to other pesticides - particularly other AChE-inhibiting pesticides, chemical pollutants, and sediments that also degrade water quality. Depending on the available water flow, amount of shade from LWD and intact riparian zones, and water temperature in aquatic habitats, the toxicity of carbaryl, carbofuran, and methomyl in tributary and stream waters may become more pronounced. Reductions in water quality may reduce the conservation value of designated habitats used for spawning, rearing, and migration. Furthermore restoration actions promoted in many of the salmonid recovery plans focus on increasing flood plain connectivity and creating new off-channel habitats. These actions are proposed in agricultural and urban flood plains that overlap with uses of the three insecticides. Water quality (as well as prey availability) may be degraded in these newly constructed habitats from the stressors of the action- effectively precluding intended benefits to rearing juvenile salmonids. We expect that proposed uses may contaminate these areas, thereby precluding habitat from its intended purpose in supporting the survival and recovery of listed Pacific salmonids.

The precise change in the conservation value of critical habitat within the ESU/DPS from the proposed action cannot be quantified and will likely vary according to the specific designated critical habitat. However, based on the effects described above, it is reasonably likely that the proposed action will have a large, local, negative reduction in that conservation value of designated critical habitat within highly developed agricultural, residential, and urban areas. The duration, frequency, and severity of these reductions will vary according to overall numbers and volume of applications of carbaryl, carbofuran, and methomyl in areas of designated critical habitat, among other variables.

Therefore, we expect that the registration of carbaryl and carbofuran will adversely affect designated critical habitat of all listed Pacific salmonids except for that of Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon. We also expect that the reregistration of methomyl will adversely affect designated critical habitat of all listed Pacific salmonids, except for that of Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, Snake River fall-run Chinook salmon, Snake River spring-summer-run Chinook salmon, California Coastal Chinook salmon, and Snake River steelhead.

Conclusion

Carbaryl and Carbofuran

After reviewing the current status of California Coastal Chinook salmon, Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Central California Coast coho salmon, LCR coho salmon, Southern Oregon and Northern Coastal California coho salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Puget Sound steelhead, Snake River Basin steelhead, South Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of carbaryl and carbofuran is likely to jeopardize the continued existence of these endangered or threatened species (Table 81).

It is NMFS' Opinion that the registration of carbaryl and carbofuran is not likely to jeopardize the continued existence of Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon (Table 81).

After reviewing the current status of designated critical habitat for California Coastal Chinook salmon, Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Central California Coast coho salmon, Southern Oregon/Northern Coastal California coho salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Snake River Basin steelhead, South-Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of carbaryl and carbofuran, is likely to result in the destruction or adverse modification of critical habitat of these endangered and threatened species (Table 82).

It is NMFS' Opinion that the registration of carbaryl and carbofuran is not likely to result in the destruction or adverse modification of critical habitat of Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, and Oregon Coast coho salmon (Table 82).

Methomyl

After reviewing the current status of Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Central California Coast coho salmon, LCR coho salmon, Southern Oregon and Northern Coastal California coho salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Puget Sound steelhead, South Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of methomyl is likely to jeopardize the continued existence of these endangered or threatened species (Table 81).

It is NMFS' Opinion that the registration of methomyl is not likely to jeopardize the continued existence of California Coastal Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, and Snake River steelhead (Table 81).

After reviewing the current status of designated critical habitat for Central Valley spring-run Chinook salmon, LCR Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, UCR spring-run Chinook salmon, Upper Willamette River Chinook salmon, Central California Coast coho salmon, LCR coho salmon, Southern Oregon and Northern Coastal California coho salmon, California Central Valley steelhead, Central California Coast steelhead, LCR steelhead, MCR steelhead, Puget Sound steelhead, South Central California coast steelhead, Southern California steelhead, UCR steelhead, and Upper Willamette River steelhead, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS' Opinion that the registration of methomyl is likely to result in the destruction or adverse modification of critical habitat of these endangered and threatened species (Table 82).

It is NMFS' Opinion that the registration of methomyl is not likely to result in the destruction or adverse modification of critical habitat of California Coastal Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Ozette Lake sockeye salmon, Snake River sockeye salmon, Northern California steelhead, Columbia River chum salmon, Hood Canal summer-run chum salmon, Oregon Coast coho salmon, and Snake River steelhead (Table 82).

Table 81. Jeopardy and Non-Jeopardy Determinations for Listed Species

<table>
<tr><th>Common Name
(Scientific Name)</th><th>Distinct Population Segment or Evolutionarily Significant Unit</th><th>Carbaryl</th><th>Carbofuran</th><th>Methomyl</th></tr>
<tr><td rowspan="9">Chinook salmon
Oncorhynchus tshawytscha</td><td>California Coastal</td><td colspan="2">Jeopardy</td><td>Non-Jeopardy</td></tr>
<tr><td>Central Valley Spring-run</td><td colspan="3" rowspan="5">Jeopardy</td></tr>
<tr><td>Lower Columbia River</td></tr>
<tr><td>Upper Columbia River Spring-run</td></tr>
<tr><td>Puget Sound</td></tr>
<tr><td>Sacramento River Winter-run</td></tr>
<tr><td>Snake River Fall-run</td><td colspan="2" rowspan="2">Jeopardy</td><td rowspan="2">Non-Jeopardy</td></tr>
<tr><td>Snake River Spring/Summer-run</td></tr>
<tr><td>Upper Willamette River</td><td colspan="3">Jeopardy</td></tr>
<tr><td rowspan="2">Chum salmon
Oncorhynchus keta</td><td>Columbia River</td><td colspan="3" rowspan="2">Non-Jeopardy</td></tr>
<tr><td>Hood Canal Summer-run</td></tr>
<tr><td rowspan="4">Coho salmon
Oncorhynchus kisutch</td><td>Lower Columbia River</td><td colspan="3">Jeopardy</td></tr>
<tr><td>Oregon Coast</td><td colspan="3">Non-Jeopardy</td></tr>
<tr><td>Southern Oregon & Northern California Coast</td><td colspan="3" rowspan="2">Jeopardy</td></tr>
<tr><td>Central California Coast</td></tr>
<tr><td rowspan="2">Sockeye salmon
Oncorhynchus nerka</td><td>Ozette Lake</td><td colspan="3" rowspan="2">Non-Jeopardy</td></tr>
<tr><td>Snake River</td></tr>
<tr><td rowspan="11">Steelhead
Oncorhynchus mykiss</td><td>Central California Coast</td><td colspan="3" rowspan="4">Jeopardy</td></tr>
<tr><td>California Central Valley</td></tr>
<tr><td>Lower Columbia River</td></tr>
<tr><td>Middle Columbia River</td></tr>
<tr><td>Northern California</td><td colspan="3">Non-Jeopardy</td></tr>
<tr><td>Puget Sound</td><td colspan="3">Jeopardy</td></tr>
<tr><td>Snake River</td><td colspan="2">Jeopardy</td><td>Non-Jeopardy</td></tr>
<tr><td>South-Central California Coast</td><td colspan="3" rowspan="4">Jeopardy</td></tr>
<tr><td>Southern California</td></tr>
<tr><td>Upper Columbia River</td></tr>
<tr><td>Upper Willamette River</td></tr>
</table>

Table 82. Adverse Modification Determinations for Designated Critical Habitat of Listed Species

<table>
<tr><th>Common Name
(Scientific Name)</th><th>Distinct Population Segment or Evolutionarily Significant Unit</th><th>Carbaryl</th><th>Carbofuran</th><th>Methomyl</th></tr>
<tr><td rowspan="9">Chinook salmon
Oncorhynchus tshawytscha</td><td>California Coastal</td><td colspan="2">Adverse Modification</td><td>No Adverse Modification</td></tr>
<tr><td>Central Valley Spring-run</td><td colspan="3" rowspan="5">Adverse Modification</td></tr>
<tr><td>Lower Columbia River</td></tr>
<tr><td>Upper Columbia River Spring-run</td></tr>
<tr><td>Puget Sound</td></tr>
<tr><td>Sacramento River Winter-run</td></tr>
<tr><td>Snake River Fall-run</td><td colspan="2" rowspan="2">Adverse Modification</td><td rowspan="2">No Adverse Modification</td></tr>
<tr><td>Snake River Spring/Summer-run</td></tr>
<tr><td>Upper Willamette River</td><td colspan="3">Adverse Modification</td></tr>
<tr><td rowspan="2">Chum salmon
Oncorhynchus keta</td><td>Columbia River</td><td colspan="3" rowspan="2">No Adverse Modification</td></tr>
<tr><td>Hood Canal Summer-run</td></tr>
<tr><td rowspan="4">Coho salmon
Oncorhynchus kisutch</td><td>Lower Columbia River</td><td colspan="3">No Designated Critical Habitat</td></tr>
<tr><td>Oregon Coast</td><td colspan="3">No Adverse Modification</td></tr>
<tr><td>Southern Oregon & Northern California Coast</td><td colspan="3" rowspan="2">Adverse Modification</td></tr>
<tr><td>Central California Coast</td></tr>
<tr><td rowspan="2">Sockeye salmon
Oncorhynchus nerka</td><td>Ozette Lake</td><td colspan="3" rowspan="2">No Adverse Modification</td></tr>
<tr><td>Snake River</td></tr>
<tr><td rowspan="11">Steelhead
Oncorhynchus mykiss</td><td>Central California Coast</td><td colspan="3" rowspan="4">Adverse Modification</td></tr>
<tr><td>California Central Valley</td></tr>
<tr><td>Lower Columbia River</td></tr>
<tr><td>Middle Columbia River</td></tr>
<tr><td>Northern California</td><td colspan="3">No Adverse Modification</td></tr>
<tr><td>Puget Sound</td><td colspan="3">No Designated Critical Habitat</td></tr>
<tr><td>Snake River</td><td colspan="2">Adverse Modification</td><td>No Adverse Modification</td></tr>
<tr><td>South-Central California Coast</td><td colspan="3" rowspan="4">Adverse Modification</td></tr>
<tr><td>Southern California</td></tr>
<tr><td>Upper Columbia River</td></tr>
<tr><td>Upper Willamette River</td></tr>
</table>

Reasonable and Prudent Alternatives

This Opinion has concluded that EPA's proposed registration of pesticides containing carbaryl and carbofuran is likely to jeopardize the continued existence of 22 endangered and threatened Pacific salmonids and is likely to destroy or adversely modify designated critical habitat for 20 threatened and endangered salmonids. The Opinion further concluded that EPA's proposed registration of pesticides containing methomyl is likely to jeopardize the continued existence of 18 endangered and threatened Pacific salmonids and is likely to destroy or adversely modify designated critical habitat for 16 threatened and endangered salmonids. "Jeopardize the continued existence of" means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR §402.02).

Regulations (50 CFR §402.02) implementing section 7 of the ESA define reasonable and prudent alternatives as alternative actions, identified during formal consultation, that: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) NMFS believes would avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat.

NMFS reached this conclusion because measured and predicted concentrations of the three a.i.s in salmonid habitats, particularly in off-channel habitats[5], are likely to cause adverse effects to listed Pacific salmonids including significant reductions in growth and survival. For carbaryl and carbofuran, 22 ESUs/DPSs of listed Pacific salmonids are likely to suffer reductions in

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[5] Off-channel habitat – water bodies and/or inundated areas that are connected (accessible to salmonid juveniles) seasonally or annually to the main channel of a stream including but not limited to features such as side channels, alcoves, ox bows, ditches, and floodplains.

Main channel –the stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel.

viability given the severity of expected changes in abundance and productivity associated with the proposed action. Similarly for methomyl, 18 ESUs/DPSs of listed Pacific salmonids are likely to suffer reductions in viability. These adverse effects are expected to appreciably reduce the likelihood of both the survival and recovery of these listed Pacific salmonids. EPA's proposed registration of carbaryl and carbofuran is also likely to result in the destruction or adverse modification of critical habitat for 20 affected ESUs/DPSs because of adverse effects on salmonid prey and water quality in freshwater rearing, spawning, and foraging areas. EPA's proposed registration of methomyl is also likely to result in the destruction or adverse modification of critical habitat for 16 affected ESUs/DPSs because of adverse effects on salmonid prey and water quality in freshwater rearing, spawning, and foraging areas.

The Reasonable and Prudent Alternative (RPA) accounts for the following issues: (1) the action will result in exposure to other chemical stressors in addition to the a.i. that may increase the risk of the action to listed species, including unspecified inert ingredients, adjuvants, and tank mixes; (2) exposure to chemical mixtures containing carbaryl, carbofuran, and methomyl and other cholinesterase-inhibiting compounds result in additive and synergistic responses; and (3) exposure to other chemicals and physical stressors (*e.g.*, pH and temperature) in the baseline habitat will likely intensify response to carbaryl, carbofuran, and methomyl.

The action as implemented under the RPA will remove the likelihood of jeopardy and of destruction or adverse modification of critical habitat. In the proposed RPA, NMFS does not attempt to ensure there is no take of listed species. NMFS believes take will occur, and has provided an incidental take statement exempting that take from the take prohibitions, so long as the action is conducted according to the RPA and reasonable and prudent measures (RPM). Avoiding take altogether would most likely entail canceling registration, or prohibiting use in watersheds inhabited by salmonids. NMFS recognizes that carbofuran's registration is currently in the process of being cancelled. However, NMFS is uncertain when and if cancellation will occur. Furthermore, existing stocks under older labels would remain and are currently allowed to be applied. The RPA and RPMs therefore apply to carbofuran as well. The goal of the RPA is to reduce exposure to ensure that the action is not likely to jeopardize listed species or destroy or adversely modify critical habitat.

The RPA is comprised of six required elements that must be implemented in its entirety within one year of the receipt of EPA's Opinion to ensure the proposed registration of these pesticides is not likely to jeopardize endangered or threatened Pacific salmonids under the jurisdiction of NMFS or destroy or adversely modify critical habitat that has been designated for these species. All elements of the RPA apply only to those ESUs/DPSs where there is jeopardy or the destruction or adverse modification of critical habitat. These elements rely upon recognized practices for reducing drift and runoff of pesticide products into aquatic habitats.

Specific Elements of the Reasonable and Prudent Alternative

Elements 1-4 shall be specified on FIFRA labels of all pesticide products containing carbaryl, carbofuran, and methomyl used in California, Idaho, Oregon, and Washington. Alternatively, the label could direct pesticide users to the EPA's Endangered Species Protection Program (ESPP) bulletins that specify elements 1-4. For purposes of this RPA salmonid habitats are defined as freshwaters, estuarine habitats, and nearshore marine habitats including bays within the ESU/DPS ranges including migratory corridors. The freshwater habitats include intermittent streams and other temporally connected habitats to salmonid-bearing waters. Freshwater habitats also include all known types of off-channel habitats as well as drainages, ditches, and other man-made conveyances to salmonid habitats that lack salmonid exclusion devices (*e.g.*, screens).

Element 1.

Do not apply pesticide products[6] within specified buffers of salmonid habitats (Table 83). Buffers only apply to those salmonid habitats where NMFS concluded jeopardy or the destruction or adverse modification of designated critical habitat for listed Pacific salmonids. Buffers also only apply when water exists in the stream or habitat and shall be measured from the water's edge of salmonid habitat, including off-channel habitat, to the point of deposition (below spray nozzle).

Pesticide buffers are recognized tools to reduce pesticide loading into aquatic habitats from drift. EPA, USFWS, NMFS, courts, and state agencies routinely enlist buffers as pesticide load

[6] Use of the term "pesticide products" in the Reasonable and Prudent Alternative section of the Opinion refers to pesticide products containing carbaryl, carbofuran, and methomyl.

reduction measures. EPA requires the use of buffers on end-use product labels for ground and/or aerial applications for some products that pose risk to aquatic systems. For example, many methomyl containing end-use products have mandated buffers of 25, 100, and 450 ft for ground, aerial, and Ultra Low Volume applications, respectively. CDPR has pesticide use limitations of 120 and 600 ft buffers for carbaryl, carbofuran, and methomyl-containing pesticides when the wind is blowing toward sensitive areas. On June 14, 1989, USFWS issued a Biological Opinion for 165 listed species and 112 pesticide a.i.s. Prescribed buffers under species-specific RPAs ranged from 60 ft (ground applications) to one half mile (aerial applications). Many of EPA's historical county bulletins for endangered species referenced a 60 ft buffer for ground applications and a 300 ft buffer for aerial spraying. One court decision prescribed mandatory 60 ft (ground) and 300 ft (aerial) buffers for applications within the ranges of ESA-listed Pacific salmonids. NMFS has prescribed a range of buffers in ESA consultations for herbicide and insecticide application actions by agencies such as the U.S. Forest Service and Bureau of Land Management overlapping with ESA-listed salmonid habitats. Herbicide buffers ranged from 0 ft to 500 ft depending on application type, rate, and frequency. Insecticide buffers ranged from 0 ft to 200 ft depending on application type, rate, and frequency.

NMFS generated estimated environmental concentrations for the three *N*-methyl carbamates for off-channel habitats using the AgDrift model (set to EPA Tier 1 simulation defaults). NMFS generated values for a range of buffer sizes in 100 ft increments for ground applications (0 – 1,000 ft), and aerial applications (0 – 1,000 ft). The dimensions of the off-channel habitat modeled were 32.8 ft (10 m) wide and 0.328 ft (0.1 m) deep. Key model assumptions include:

- Drift as the only pathway of exposure;
- A single application of each a.i.; and
- The estimates represent instantaneous average concentrations across the entire habitat modeled.

Table 83. Mandatory pesticide no application buffers for ground and aerial applications

Rate	Carbaryl	Carbofuran	Methomyl
lbs/ acre	No Application Buffer (ft)		
Ground Applications			
0 -1	200	200	50
≥1-3	300	300	NA
≥3-5	400	400	NA
≥5-10	500	500	NA
≥10	600	600	NA
Aerial Applications			
All rates	1000	1000	600

NA – Not Applicable. Methomyl has no current registrations permitting maximum applications ≥ 1 lb ai/A

The estimated concentrations decline as buffer size increases **(**Table 84**).** We note the disparity between the concentrations predicted by the two models. For example, at 100 ft. the predicted concentrations are 4.4 µg/L from a ground application and 92.9 µg/L from an aerial application. The two results are not directly comparable because the models use different methods to predict amount of drift.

Table 84. Estimated environmental concentrations of carbaryl, carbofuran, and methomyl applied at the rate of 1 lb per acre for ground and aerial applications.

Ground application, low boom, ASAE very fine-fine droplet distribution, 50th percentile estimates EPA Tier 1 simulations	
Buffer (ft)	Off-Channel (10 m * 0.1 m) (µg/L)
0	76.427
10	20.168
100	4.406
200	2.568
300	1.813
400	1.392
500	1.122
600	0.933
700	0.794
800	0.688
900	0.604
997	0.583
Aerial application, fine-medium droplet distribution. EPA Tier 1 simulation	
Buffer (ft)	Off-Channel (10 m * 0.1 m) (µg/L)
0	333.566
10	260.482
100	92.888
200	48.985
300	33.096
400	25.289
500	20.902
600	18.010
700	16.035
800	14.692
900	13.719
997	12.983

Based on the population growth modeling exercises, a four-day exposure to expected concentrations of carbaryl, carbofuran, and methomyl will substantially reduce a population's growth rate due to prey base reduction. All four life history types modeled demonstrated this effect. As exposure duration and application rate increases, we expect more pronounced effects on salmonid prey abundance and recovery timing leading to further reduced salmonid growth. We expect that some juvenile salmonids are likely to experience reductions in growth. However, the prescribed buffers are intended to avoid population-level effects. We also expect that prey items will die from these exposures. The likelihood of impacting salmonid prey availability, an identified PCE, is substantially reduced by these buffers.

The majority of buffers described earlier are smaller than the buffers prescribed in this element. Concentrations expected with smaller buffers would lead to a greater probability of affecting populations and PCEs especially in habitats that are compromised from a variety of stressors (described in the *Environmental Baseline s*ection).

The scenario we modeled with AgDrift in this RPA element is expected to occur when all of the modeled variables are present *e.g.*, specific wind speed, wind direction, release height, size of off-channel habitat, droplet size distribution, etc. The input variables are relevant to field conditions and the frequency of this exact scenario occurring remains unknown. We selected this scenario to represent off-channel habitats used by a sensitive salmonid life stage *i.e.*, juveniles. NMFS believes that these buffers will remove a substantial portion of risk attributed to pesticide drift.

*Element 2***.** Do not apply when wind speeds are greater than or equal to 10 mph as measured using an anemometer immediately prior to application. Because wind conditions may change during application of pesticide products, commence applications on the side nearest the aquatic habitat and proceed away from the aquatic habitat.

Element 3. For all uses do not apply pesticide products when soil moisture is at field capacity, or when a storm event likely to produce runoff from the treated area is forecasted by NOAA/NWS (National Weather Service), to occur within 48 h following application.

Element 4. Report all incidents of fish mortality that occur within four days of application and within the vicinity of the treatment area to EPA OPP (703-305-7695).

*Element 5***.** In addition to the labeling requirements above, EPA shall develop and implement a NMFS-approved effectiveness monitoring plan for off-channel habitats with annual reports. The plan shall identify representative off-channel habitats within agricultural areas prone to drift and runoff of pesticides. The number and locations of off-channel habitat sampling sites shall include currently used off-channel habitats by threatened and endangered Pacific salmonids identified by NMFS biologists and will include at least two sites for each general species (ESU or DPS; *i.e.*, coho salmon, chum salmon, steelhead, sockeye salmon, and ocean-type Chinook and stream-type Chinook salmon). The plan shall collect daily surface water samples targeting at least three periods during the application season for seven days. Collected water samples will be analyzed for current-use OPs and carbamates following USGS schedule for analytical chemistry. The report shall be submitted to NMFS OPR and will summarize annual monitoring data and provide all raw data.

*Element 6***.** This element is specific to Washington State’s 24(c) for carbaryl applications on estuarine mudflats. As this use is specific for application on mudflats within estuaries, elements 1, 3, and 4 are not applicable for this use. Elements 2 and 5 are required. A monitoring program approved by NMFS OPR shall be established in Willapa Bay and Grays Harbor to determine the presence, AChE activity, and genetic source populations of captured juvenile salmonids. For example, a monitoring program shall be implemented to determine salmonid health at five locations using beach seines with the first incoming tide after carbaryl application. Additionally, fyke nets placed in five channels proximate to sprayed mudflats shall be used to determine and health at first outgoing tide after carbaryl application. All dead or dying salmonids, if any, shall be collected and genetically analyzed to determine source population origin. Additionally, muscle and brain AChE activity shall be measured in collected dead or dying salmonids. A subset of the live salmonids captured in seines and nets shall be genetically analyzed to determine source population origin. Annual reports shall be submitted to NMFS and include the raw data and summaries of the raw data including number of fish caught, species of salmonids, genetic analysis results, AChE activity results of brain and muscle samples, and number of dead or dying salmonids collected. The sampling effort shall be conducted annually as long as carbaryl is allowed for use in Willapa Bay and Grays Harbor.

Although NMFS has concluded that EPA's action for carbaryl and carbofuran is likely to jeopardize 22 listed ESUs/DPSs and destroy or adversely modify 20 designated critical habitats during the 15-year duration of the proposed action, NMFS does not believe that the effects of the action will attain this level in the year between issuance of this Opinion and EPA's implementation of the RPA. NMFS has further concluded that EPA's action for methomyl is likely to jeopardize 18 listed ESUs/DPSs and destroy or adversely modify designated critical habitats for 16 listed species during the 15-year duration of the proposed action. As with carbaryl and carbofuran, NMFS does not believe that the effects of the action for methomyl will attain this level in the year between issuance of this Opinion and EPA's implementation of the RPA. Products containing these three a.i.s have been in use for some time. NMFS believes that these products have contributed to ESU/DPS declines, but not to the extent that one year of additional use as now authorized would lead to likely jeopardy or adverse modification.

Based on the above conclusions on the effects of EPA's proposed registration of pesticide products containing carbaryl, carbofuran, and methomyl, the EPA is required to notify NMFS OPR of its final decision on implementation of the RPA.

Incidental Take Statement

Section 9 of the ESA and federal regulation pursuant to section 4(d) of the ESA prohibits the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Take Anticipated

As described earlier in this Opinion, this is a consultation on the EPA's registration of pesticide products containing carbaryl, carbofuran, and methomyl, and their formulations as they are used in the Pacific Northwest and California and the impacts of these applications on listed ESUs/DPSs of Pacific salmonids. The EPA authorizes use of these pesticide products for pest control purposes across multiple landscapes. The goal of this Opinion is to evaluate the impacts to NMFS' listed resources from the EPA's broad authorization of applied pesticide products. This Opinion is a partial consultation because pursuant to the court's order, EPA sought consultation on only 26 listed Pacific salmonids under NMFS' jurisdiction. However, even though the court's order did not address the two more recently listed ESUs and DPSs, NMFS analyzed the impacts of EPA's actions to them because they belong to the same taxon and the analysis requires consideration of the same information. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species under NMFS' jurisdiction and consults with NMFS as necessary.

For this Opinion, NMFS anticipates the general direct and indirect effects that would occur from EPA's registration of pesticide products across the states of California, Idaho, Oregon, and Washington to 28 listed Pacific salmonids under NMFS' jurisdiction during the 15-year duration of the proposed action. Recent and historical surveys indicate that listed salmonids occur in the action area, in places where they will be exposed to the stressors of the action. The RPAs are designed to reduce this exposure but not eliminate it. Pesticide runoff and drift of carbaryl, carbofuran, and methomyl are most likely to reach streams and other aquatic sites when they are applied to crops and other land use settings located adjacent to wetlands, riparian areas, ditches, off-channel habitats, and intermittent streams. These inputs into aquatic habitats are especially high when rainfall immediately follows applications. The effects of pesticides and other contaminants found in urban runoff especially from areas with a high degree of impervious surfaces may also exacerbate degraded water quality conditions of receiving waters used by salmon. Urban runoff is also generally warmer in temperature and elevated water temperature pose negative effects on certain life history phases for salmon, increasing susceptibility to chemical stressors. The range of effects of the three a.i.s on salmonids include reductions in growth, prey capture, and swimming ability, impaired olfaction affecting homing and reproductive behaviors, and increased susceptibility to predation and disease. Thus, we expect some exposed fish will respond to these effects by changing normal behaviors. In some cases, fish may die, be injured, or suffer sublethal effects. These results are not the purpose of the proposed action. Therefore, incidental take of listed salmonids is reasonably certain to occur over the 15-year duration of the proposed action.

Given the variability of real-life conditions, the broad nature and scope of the proposed action, and the migratory nature of salmon, the best scientific and commercial data available are not sufficient to enable NMFS to estimate a specific amount of incidental take associated with the proposed action. As explained in the *Description of the Proposed Action* and the *Effects of the Proposed Action* sections, NMFS identified multiple uncertainties associated with the proposed action. Areas of uncertainty include:

1. Incomplete information on the proposed action (*i.e.*, no master label summarizing all authorized uses of pesticide products containing carbaryl, carbofuran, and methomyl);
2. Limited use and exposure data on stressors of the action for non-agricultural uses of these pesticides;
3. Minimal information on exposure and toxicity for pesticide formulations, adjuvants, and other/inert ingredients within registered formulations;
4. No information on permitted tank mixtures and associated exposure estimates;
5. Limited data on toxicity of environmental mixtures;
6. No known method to predict synergistic responses from exposure to combinations of the three a.i.s;
7. Annual variable conditions regarding land use, crop cover, and pest pressure;
8. Variable temporal and spatial conditions within each ESU, especially at the population-level; and
9. Variable conditions of water bodies in which salmonids live.

NMFS therefore identifies as a surrogate for the allowable extent of take the ability of this action to proceed without any fish kills attributed to the use of carbaryl, carbofuran, or methomyl or any compounds, degradates, or mixtures in aquatic habitats containing individuals from any ESU/DPS. Because of the difficulty of detecting salmonid deaths, the fishes killed are not to be listed salmonids. Salmonids appear to be more sensitive to these compounds, so that if there are kills of other freshwater fishes attributed to use of these pesticides, it is likely that salmonids have also died, even if no dead salmonids can be located. In addition, if stream conditions due to pesticide use kill less sensitive fishes in certain areas, the potential for lethal and non-lethal takes in downstream areas increases. A fish kill is considered attributable to one of these three ingredients, its metabolites, or degradates, if measured concentrations in surface waters are at levels expected to kill fish, if AChE measurements were taken of the fish carcass and correlate to fish death, if pesticides were applied in the general area, and if pesticide drift or runoff was witnessed or apparent.

NMFS notes that with increased monitoring and study of the impact of these pesticides on water quality, particularly water quality in off-channel habitats, NMFS will be able to refine this incidental take statement, and future incidental take statements, to allow other measures of the extent of take.

Reasonable and Prudent Measures

The measures described below are non-discretionary, and must be undertaken by the EPA so that they become binding conditions of any grant or permit issued to the applicant(s), as appropriate, for the exemption in section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this incidental take statement. If the EPA (1) fails to assume and implement the terms and conditions or (2) fails to require the applicant(s) to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the EPA must report the progress of the action and its impact on the species to NMFS OPR as specified in the incidental take statement. [50 CFR§402.14(i)(3)].

To satisfy its obligations pursuant to section 7(a)(2) of the ESA, the EPA must monitor (a) the direct, indirect, and cumulative impacts of its long-term registration of pesticide products containing carbaryl, carbofuran, and methomyl; (b) evaluate the direct, indirect, or cumulative impacts of pesticide misapplications in the aquatic habitats in which they occur; and (c) the consequences of those effects on listed Pacific salmonids under NMFS' jurisdiction. The purpose of the monitoring program is for the EPA to use the results of the monitoring data and modify the registration process in order to reduce exposure and minimize the effect of exposure where pesticides will occur in salmonid habitat.

The EPA shall:

1. Minimize the amount and extent of incidental take from use of pesticide products containing carbaryl, carbofuran, and methomyl by reducing the potential of chemicals reaching the water;
2. Monitor any incidental take or surrogate measure of take that occurs from the action; and
3. Report annually to NMFS OPR on the monitoring results from the previous season.

Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, within one year following the date of issuance of this Opinion, the EPA must comply with the following terms and conditions. These terms and conditions implement the reasonable and prudent measure described above. These terms and conditions are non-discretionary.

1. a. EPA shall include the following instructions requiring reporting of fish kills either on the labels for all products containing carbaryl, carbofuran, and methomyl or in ESPP Bulletins:

   NOTICE: Incidents where salmon appear injured or killed as a result of pesticide applications shall be reported to NMFS OPR at 301-713-1401 and EPA at 703-305-7695. The finder should leave the fish alone, make note of any circumstances likely causing the death or injury, location and number of fish involved, and take photographs, if possible. Adult fish should generally not be disturbed unless circumstances arise where an adult fish is obviously injured or killed by pesticide exposure, or some unnatural cause. The finder may be asked to carry out instructions provided by NMFS OPR to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved.

   b. EPA shall report to NMFS OPR any incidences regarding carbaryl, carborfuran, or methomyl effects on aquatic ecosystems added to its incident database that it has classified as probable or highly probable.

   c. Do not apply pesticide products when wind speeds are greater than or equal to 10 mph as measured using an anemometer immediately prior to application. When applying pesticide products, commence applications on the side nearest the aquatic habitat and proceed away from the aquatic habitat.

2. For all uses do not apply pesticide products when soil moisture is at field capacity, or when a storm event likely to produce runoff from the treated area is forecasted by NOAA/NWS (National Weather Service), to occur within 48 h following application.

Conservation Recommendations

Section 7(a) (1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultations involving future authorizations of pesticide a.i.s that may affect listed species:

1. Conduct mixture toxicity analysis in screening-level and endangered species biological evaluations;
2. Develop models to estimate pesticide concentrations in off-channel habitats; and
3. Develop models to estimate pesticide concentrations in aquatic habitats associated with non-agricultural applications, particularly in residential and industrial environments.

In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the EPA should notify NMFS OPR of any conservation recommendations it implements in the final action.

Reinitiation Notice

This concludes formal consultation on the EPA's proposed registration of pesticide products containing carbaryl, carbofuran, and methomyl and their formulations to ESA-listed Pacific salmonids under the jurisdiction of the NMFS. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the extent of take specified in the *Incidental Take Statement* is exceeded; (2) new information reveals effects of this action that may affect listed species or designated critical habitat in a manner or to an extent not previously considered in this biological opinion; (3) the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the identified action. If reinitiation of consultation appears warranted due to one or more of the above circumstances, EPA must contact NMFS OPR. If none of these reinitiation triggers are met within the next 15 years, then reinitiation will be required because the Opinion only covers the action for 15 years.

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Appendix 1: Population Modeling

Introduction

To assess the potential for adverse impacts of the anticholinesterase insecticides on Pacific salmon populations, a model was developed that explicitly links impairments in the biochemistry, behavior, prey availability and somatic growth of individual salmon to the productivity of salmon populations. More specifically, the model connects known effects of the pesticides on salmon physiology and behavior with community-level effects on salmon prey to estimate population-level effects on salmon. The model used here is an extension of one developed for investigating the direct effects of pesticides on the biochemistry, behavior, and growth of ocean-type Chinook salmon (Baldwin et al., in press).

In the freshwater portion of their life, Pacific salmon may be exposed to insecticides that act by inhibiting acetylcholinesterase (AChE). AChEe is a crucial enzyme in the proper functioning of cholinergic synapses in the central and peripheral nervous systems of vertebrates and invertebrates. Of consequence to salmon, anticholinesterase insecticides have been shown to interfere with salmon swimming behavior (Beauvais et al. 2000, Brewer et al. 2001, Sandahl et al. 2005), feeding behavior (Sandahl et al. 2005), foraging behavior (Morgan and Kiceniuk 1990), homing behavior (Scholz et al. 2000), anti-predator behaviors (Scholz et al. 2000), and reproductive physiology (Moore and Waring 1996, Waring and Moore 1997, Scholz et al. 2000).

Anticholinesterase insecticides have also been found to reduce benthic densities of aquatic invertebrates and alter the composition of aquatic communities (Liess and Schulz 1999, Schulz and Liess 1999, Schulz et al. 2002, Fleeger et al. 2003, Schulz 2004, Chang et al. 2005, Relyea 2005). Spray drift and runoff from agricultural and urban areas can expose aquatic invertebrates to relatively low concentrations of insecticides for as little as minutes or hours, but populations of many taxa can take months or even years to recover to pre-exposure or reference densities (Wallace et al. 1991, Liess and Schulz 1999, Anderson et al. 2003, Stark et al. 2004). For example, when an aquatic macroinvertebrate community in a German stream was exposed to runoff containing parathion (an AChE inhibitor) and fenvalerate (another commonly used insecticide), eight of eleven abundant species disappeared and the remaining three were reduced in abundance (Liess and Schulz 1999). Long-term changes in invertebrate densities and community composition likely result in reductions in salmon prey availability. Therefore, in addition to the direct impacts that AChE inhibitors have on salmon, there may also be, independently, significant indirect effects to salmon via their prey (Peterson et al. 2001a). Wild juvenile salmon feed primarily on invertebrates in the water column and those trapped on the water's surface, actively selecting the largest items available (Healey 1991, Quinn 2005). Salmon are often found to be food limited (Quinn 2005), suggesting that a reduction in prey number or size due to insecticide exposure may further stress salmon. For example, Davies and Cook (1993) found that several months following a spray drift event, benthic and drift densities were still reduced in exposed stream reaches. Consequently, brown trout in the exposed reaches fed less and grew at a slower rate compared to those in unexposed stream reaches (Davies and Cook 1993). Although the insecticide in their study was cypermethrin (a pyrethroid), similar reductions in macroinvertebrate density and recovery times have been found in studies with AChE inhibitors (Liess and Schulz 1999, Schulz et al. 2002), suggesting indirect effects to salmon via prey availability may be similar.

One likely biological consequence of reduced swimming, feeding, foraging, and prey availability is a reduction in food uptake and, subsequently, a reduction in somatic growth of exposed fish. Juvenile growth is a critical determinant of freshwater and marine survival for Chinook salmon (Higgs et al. 1995). Reductions in the somatic growth rate of salmon fry and smolts are believed to result in increased size-dependent mortality (Healey 1982, West and Larkin 1987, Zabel and Achord 2004). Zabel and Achord (2004) observed size-dependent survival for juvenile salmon during the freshwater phase of their outmigration. Mortality is also higher among smaller and slower growing salmon because they are more susceptible to predation during their first winter (Healey 1982, Holtby et al. 1990, Beamish and Mahnken 2001). These studies suggest that factors affecting the organism and reducing somatic growth, such as anticholinesterase insecticide exposure, could result in decreased first-year survival and, thus, reduce population productivity.

Changes to the size of juvenile salmon from exposure to carbaryl, carbofuran, and methomyl were linked to salmon population demographics. We used size-dependent survival of juveniles during a period of their first year of life. We did this by constructing and analyzing general life history matrix models for coho salmon (*Oncorhynchus kisutch*), sockeye salmon (*O. nerka*), and ocean-type and stream-type Chinook salmon (*O. tshawytscha*). A steelhead (*O. mykiss*) life history model was not constructed due to the lack of demographic information relating to the proportions of resident and anadromous individuals, the freshwater residence time of steelhead, and rates of repeated spawning. The basic salmonid life history modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. Differences between the modeled strategies are lifespan of the female, time to reproductive maturity, and the number and relative contribution of the reproductive age classes (Figure 1). The coho females we modeled reach reproductive maturity at age 3 and provide all of the reproductive contribution. Sockeye females reach maturity at age 4 or 5, but the majority of reproductive contributions are provided by age 4 females. Chinook salmon females can mature at age 3, 4, or 5, with the majority of the reproductive contribution from ages 4 and 5. The primary difference between the ocean-type and stream-type Chinook salmon is the juvenile freshwater residence with ocean-type juveniles migrating to the ocean as sub-yearlings and stream-type overwintering in freshwater and migrating to the ocean as yearlings. The models depicted general populations representing each life history strategy and were constructed based upon literature data described below. Specific populations were not modeled due to the difficulty in finding sufficient demographic and reproductive data for a single population.

A separate acute toxicity model was constructed that estimated the population-level impacts of juvenile mortality resulting from exposure to lethal concentrations of carbaryl, carbofuran, and methomyl. These models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from an annual exposure of juveniles to a pesticide. The lethal impact was implemented as a change in first year survival for each of the salmon life history strategies.

The overall model endpoint used to assess population-level impacts for both the growth and acute lethality models was the percent change in the intrinsic population growth rate (lambda, λ) resulting from the pesticide exposure. Change in λ is an accepted population parameter often used in evaluating population productivity, status, and viability. The National Marine Fisheries Service uses changes in λ when estimating the status of species, conducting risk and viability assessments, developing endangered species recovery plans, composing biological opinions, and communicating with other federal, state and local agencies (McClure et al. 2003). While values of $\lambda<1.0$ indicate a declining population, negative changes in lambda greater than the natural variability for the population indicate a loss of productivity. This can be a cause for concern since the decline could make a population more susceptible to dropping below 1.0 due to impacts from multiple stressors.

The following models were developed to serve as a means to assess the potential effects on ESA-listed salmon populations from exposure to AChE inhibiting pesticides, including *N*-methyl carbamates and organophosphorus insecticides. The growth model focuses on the impacts to prey abundance and a salmon's ability to feed which are integrated into reductions in juvenile growth. Assessing the results from different pesticide exposure scenarios relative to a control (*i.e.*, unexposed) scenario can indicate the potential for sublethal pesticide exposures to lead to changes in the somatic growth and survival of individual sub-yearling salmon. Consequently, subsequent changes in salmon population dynamics as indicated by percent change in a population's intrinsic rate of increase assists us in forecasting the potential population-level impacts to listed populations. Also, the model helps us understand the potential influence of life history strategies that might explain differential results within the species modeled.

Methods

The model consists of two parts, an organismal portion and a population portion. The organismal portion of the model links AChE inhibition and reduced prey abundance from insecticide exposure to potential reductions in the growth of individual fish. The population portion of the model links the sizes of individual sub-yearling salmon to their survival and the subsequent growth of the population. Models were constructed using MATLAB 7.7.0 (R2008b) (The MathWorks, Inc. Natick, MA).

Organismal Model

For the organismal model a relationship between AChE activity and somatic growth of salmonid fingerlings was developed using a series of relationships between pesticide exposure, AChE activity, feeding behavior, food uptake, and somatic growth rate (Figures 2-4). The model incorporates empirical data when available. Since growth and toxicity data are limited, extrapolation from one salmon species to the others was done with the assumption that the salmon stocks would exhibit similar physiological and toxicological responses. Sigmoidal dose-response relationships based upon the AChE inhibition EC50 values and their slopes are used to determine the level of AChE activity (Figure 2A, 2B, 2C) from the exposure concentration of each pesticide exposure or pulse.

A linear relationship based on empirical data related AChE activity to feeding behavior (Sandahl et al. 2005, Figure 2D). Feeding behavior was then assumed to be directly proportional to food uptake, defined as potential ration (Figure 2E). The potential ration expresses the amount of food the organism can consume when prey abundance is not limiting. Potential ration over time (Figure 2F) depicts how the food intake of individual fish changes in response to the behavioral effects of the pesticide exposure over the modeled growth period. Potential ration is equal to final ration if no effects on prey abundance are incorporated (Figure 4). If effects of pesticide exposure on prey abundance are incorporated, final ration is the product of potential ration (relating to the fish's ability to capture prey, Figure 2) and the relative abundance of prey available following exposure (Figure 3). Next, additional empirical data (*e.g.*, Weatherley and Gill 1995) defined the relationship between final ration and somatic growth rate (Figure 4C). While the empirical relationship is more complex (*e.g.*, somatic growth rate plateaus at rations above maximum feeding), a linear model was considered sufficient for the overall purpose of this model. Finally, the model combines these linear models relating AChE activity to feeding behavior, feeding behavior to potential ration, and final ration to somatic growth rate to produce a linear relationship between AChE activity and somatic growth rate (Figure 4D). One important assumption of the model is that the relationships are stable, *i.e.*, do not change with time. The relationships would need to be modified to incorporate time as a variable if, for example, fish are shown to compensate over time for reduced AChE activity to improve their feeding behavior and increase food uptake.

Juvenile salmonids are largely opportunistic, feeding on a diverse community of aquatic and terrestrial invertebrate taxa that are entrained in the water column or on the surface (Higgs et al. 1995). As a group, these invertebrates are among the more sensitive taxa for which there is toxicity data, but within this group, there is a wide range of sensitivities. To determine a single effect concentration to use in the model analyses, a search was completed using the EPA's Ecotox database for each pesticide (http://cfpub.epa.gov/ecotox/). Several criteria were used to determine which reported effect concentrations were included in the final analysis. The data included were from studies on taxa that are known to be salmonid prey (or are functionally similar to salmonid prey); these include a diverse group of fresh and saltwater crustaceans, aquatic insects and worms. Studies with exposures of at least 24 hours (h) and not more than 96 h were included. Studies examining shorter and longer exposure times are known to affect invertebrates (*e.g.,* Peterson et al. 2001b), but these were excluded so that estimated EC50s would be comparable to previous analyses (NMFS 2008). Studies reporting invertebrate LC50s and EC50s in which mortality or immobilization of invertebrates was the recorded endpoint were included; the term "EC50" will be used in this report to describe all of these included data. Data derived for sublethal endpoints (*e.g.,* growth or reproduction) were not included. If specific data were represented more than once in the Ecotox output, duplicates were eliminated. Data from several recent peer-reviewed studies that are not yet included in the Ecotox data base, but report effect concentrations that caused mortality, were also included.

From the distributions of those data, a single effect concentration and slope were derived to best represent the diverse community of prey available in juvenile salmonid freshwater and estuarine habitats. The distributions of individual invertebrate EC50s and the geometric means of EC50s by taxa were analyzed to estimate the 50th, 10th, and 5th percentiles. Figure 43shows the distribution of geometric means of EC50s by taxa for the 10th percentile and Table 79shows the concentrations for the percentiles. Specifically, for each pesticide, a probability plot was used to graph the EC50 concentrations normalized to a normal probability distribution. For each plot, the X axis is scaled in probability (between zero and 100%) and shows the percentage of entire data whose value is less than the data point. The Y axis displays the range of the data on a log scale. The results of a linear regression of the log-transformed concentrations are shown and highlight the lognormal distribution of the data (Figure 43). In the regression equation, the normsinv() function returns the inverse of the standard normal cumulative distribution. The standard normal distribution has a mean of zero and a standard deviation of one. For example, given a percentile value of 50 (*i.e.*, a probability of 0.5), normsinv(50) returns a value of zero. The plots and regressions were performed using KaleidaGraph 4.03 (Synergy Software).

The decision to use the 10th percentile rather than the 50th percentile is consistent with previous designations by EPA, and is reasonable because of the relative sensitivity of invertebrates that are most likely consumed by juveniles. In addition, the 10th percentile is a reasonable threshold because the data included in the meta-analysis were limited to concentrations that caused mortality or immobilization within a short period of time (1-4 days). A growing number of studies on a variety of insecticides have reported that concentrations well below LC50s can cause delayed mortality or sublethal effects that may scale up to affect populations, especially in scenarios with multiple exposures and/or other stressors. Evidence for ecologically significant sublethal or delayed effects includes reduced growth rates (Schulz and Liess 2001b, Forbes and Cold 2005), altered behavior (Johnson et al. 2008), reduced emergence (Schulz and Liess 2001a), reduced reproduction (Cold and Forbes 2004, Sakamoto et al. 2006), and reduced predator defenses (Sakamoto et al. 2006, Johnson et al. 2008). Finally, the available toxicity data – and therefore the data included for these analyzes – are from studies using taxa hearty enough to survive laboratory conditions. Studies specifically examining salmonid prey that are more difficult to rear in the laboratory have documented relatively low LC50 or EC50 values when exposed to current-use pesticides (*e.g.*, Peterson et al. 2001a, Johnson et al. 2008). It is noteworthy that the most relevant invertebrate study for carbaryl – a study using a diverse group of seven salmonid prey taxa collected from streams in the Pacific Northwest – reports LC50s for carbaryl with a geometric mean of 26.28 µg/L (range 11.1 – 61.0) (Peterson et al. 2001a).

The models allow exposures that can include multiple AChE-inhibiting pesticides over various time pulses. Sigmoidal dose-response relationships, at steady-state, between each single pesticide exposure and 1) AChE activity and 2) relative prey abundance are modeled using specific EC50s and EC50s and slopes (Figure 2B and 3B). The timecourse for each exposure was built into the model as a pulse with a defined start and end during which the exposure remained constant (Figure 2A and 3A). The timecourse for AChE activity, on the other hand, was modeled using two single-order exponential functions, one for the time required for the exposure to reach full effect and the other for time required for complete recovery following the end of the exposure (time-to-effect$_{\text{AChE activity}}$ and time-to-recovery$_{\text{AChE activity}}$, respectively; Figure 2C). The apparent activity level was back-calculated to result in a relative concentration (concentration/ AChE inhibition EC50) for each day of the growth period for each pulse. The relative concentration for each day was summed across all the pulses to result in a total apparent concentration for each day. The sigmoid slope used in the calculation of AChE activity using the apparent concentration was the arithmetic mean of the sigmoid slopes for each pesticide present on each day. The timecourse for relative prey abundance was modeled incorporating a one day spike in prey drift relative to the toxicity and available prey base followed by a drop in abundance due to the toxic impacts (Figure 3C). Recovery is assumed to be due to a constant influx of invertebrates from connected habitats (aquatic and terrestrial) that are not exposed to the pesticide. Incoming organisms are subject to toxicity if pesticides are still present and this alters the rate of recovery during exposures. Incorporating dynamic effects and recovery variables allows the model to simulate differences in the pharmacokinetics (*e.g.,* the rates of uptake from the environment and of detoxification) of various pesticides and simulate differences in invertebrate community response and recovery rates (see below).

The relationship between final ration and somatic growth rate (Figure 4C) produces a relationship representing somatic growth rate over time (Figure 4D), which is then used to model individual growth rate and size over time. The growth models were run for 1,000 individual fish, with initial weight selected from a normal distribution with a mean of 1.0 g and standard deviation of 0.1 g. The size of 1.0 g was chosen to represent sub-yearling size in the spring prior to the onset of pesticide application. For each iteration of the model (one day for the organismal model), the somatic growth rate is calculated for each fish by selecting the parameter values from normal distributions with specified means and standard deviations (Table 1). The weight for each fish is then adjusted based on the calculated growth rate to generate a new weight for the next iteration. The length (days) to run the growth portion of the model was selected to represent the time from when the fish enter the linear portion of their growth trajectory in the mid to late spring until they change their growth pattern in the fall due to reductions in temperature and resources or until they migrate out of the system. The outputs of the organismal model that are handed to the population models consist of mean weights (with standard deviations) after the species-appropriate growth period (Table 2). A sensitivity analysis was run to determine the influence of the parameter values on the output of the growth model.

The parameter values defining control conditions that are constant for all the modeled species are listed in Table 1. Model parameters such as the length of the growth period and control daily growth rate that are species specific are listed in Table 2. Each exposure scenario was defined by a concentration and exposure time for each pesticide. The duration of time until full effect for the pesticides was assumed to be within a few days (Ferrari et al. 2004), with a half-life of 0.5 days.

For prey, it is assumed there is a constant, independent influx of prey from upstream habitats that will eventually (depending on the rate selected) return prey abundance to 1. As mentioned above, however, these invertebrates are subject to exposure once added to the system, and therefore prey recovery rate is a product of the influx rate as well as the exposure scenario. While recovery rates reported in the literature vary, it is assumed a 1% recovery rate is ecologically realistic (Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008). It was also assumed that regardless of the exposure scenario, relative prey abundance would not drop below a specific floor (Figure 3B). This assumption depends on a minimal yet constant terrestrial subsidy of prey and/or an aquatic community with tolerant individuals that would be available as prey, regardless of pesticide exposure and in addition to the constant recovery rate. No studies specify floors per se, but studies quantifying invertebrate densities following highly toxic exposures indicate a floor of 0.2 is ecologically realistic (*i.e.*, regardless of the exposure, 20% of a fish's ration will be available daily; *e.g.*, Cuffney et al. 1984). Finally, because prey availability has been found to increase dramatically albeit briefly following pesticide exposures (due to immediate mortality and/or emigration of benthic prey into the water column; Davies and Cook 1993, Schulz 2004), a one-day prey spike is included for the day following an exposure. The relative magnitude of the spike is calculated as the product of the standing prey availability the day prior to exposure (minus the floor), the toxicity of the exposure, and a constant of 20. This calculation therefore accounts for the potential prey that are available and the severity of the exposure. The spike will be greater when more prey are available and/or the toxicity of the exposure is greater; alternatively, the spike will be small when few prey are available and/or the exposure toxicity is low.

Below are the mathematical equations used to derive Figures 2, 3, and 4.

Figures 2A and 3A use a step function:

time < start; exposure = 0

start ≤ time ≤ end; exposure = exposure concentration(s)

time > end; exposure = 0.

Figures 2B and 3B use a sigmoid function:

y = bottom + (top – bottom)/(1 + (exposure concentration/EC50)^slope).

For 2B, y = AChE activity, top = Ac, bottom = 0.

For Figure 3B, y = prey abundance, top = Pc (in this case 1), bottom = Pf.

Figures 2D, 2E, and 4C use a linear function (the point-slope form of a line):

y = m*(x – x1) + y1.

For 2D, m = Mfa, x1 = Ac, and y1 = Fc.

For 2E, m = Mrf (computed as Rc/Fc), x1 = Fc, and y1 = Rc.

For 4C, m = Mgr, x1 = Rc, and y1 = Gc.

Figure 2C uses a series of exponential functions:

time < start; y = c

start ≤ time ≤ end; y = c – (c – i)*(1 – exp(-ke*(time – start)))

time > end; ye = c – (c – i)*(1 – exp(-ke*(end – start)))

y = ye + (c – ye)*(1 – exp(-kr*(time – end))).

For Figure 2C, c = Ac, i = Ai, ke = ln(2)/AChE effect half-life, kr = ln(2)/AChE recovery half-life. For Figure 2C the value of ye is calculated to determine the amount of inhibition that is reached during the exposure time, which may not be long enough to reach the maximum level of inhibition.

For Figure 3C, an exposure pulse would result in a 1-day spike followed by a decline to the impacted level based upon the prey toxicity. During exposures resulting in low prey toxicity, toxicity-limited recovery can occur. After exposure ends a constant rate of recovery proceeds until control drift is reached or another exposure occurs

preyavail=preydrift(day-1)-floor;

preytox=1/(1+(concentration)^preyslope);

preyrecrate=0.01;

preydriftrec = preyrecrate*preytox.

time=start; spike=(-1+10^(1.654*preyavail))*(1-preytox)

preydrift =preydrift+spike

start ≤ time ≤ end; preydrift=(preyavail*preytox)+preyrdriftrec+floor;

time>end; preydrift = preydrift(day-1)+preydriftrec

Figure 2F is generated by using the output of Figure 2C for a given time as the input for 2D and using the resulting output of 2D as the input for 2E. The resulting output of 2E produces a single time point in the relationship in 2F. Performing this series of computations across multiple days produces the entire relationship in 2F. 4D is generated by taking the outputs of 4A and 4B for the same day. Note the relationship of 4A is equivalent to 2F. The resulting outputs of 4A and 4B are multiplied to produce a final ration for a given day. The prey abundance (4B) available for consumption during a prey spike is capped at a maximum of 1.5*control drift to provide a limited benefit to the individual fish. The final ration is used as input for 4C to generate 4D.

Population Model

The weight distributions from the organismal growth portion of the model are used to calculate size-dependent first-year survival for a life history matrix population model for each species and life history type. This incorporates the impact that reductions in size could have on population growth rate and abundance. The first-year survival element of the transition matrix incorporates a size-dependent survival rate for a three- or four-month interval (depending upon the species) which takes the juveniles up to 12 months of age. This time represents the 4-month early winter survival in freshwater for stream-type Chinook salmon, coho, and sockeye models. For ocean-type Chinook salmon, it is the 3-month period the sub-yearling smolt spend in the estuary and nearshore habitats (*i.e.*, estuary survival). The weight distributions from the organismal model are converted to length distributions by applying condition factors from data for each modeled species (cf; 0.0095 for sockeye and 0.0115 for all others) as shown in Equation L.

Equation L: length(mm) = ((fish weight(g)/cf)^(1/3))*10

The relationship between length and early winter or estuary survival rate was adapted from Zabel and Achord (2004) to match the survival rate for each control model population (Howell et al. 1985, Kostow 1995, Myers et al. 2006). The relationship is based on the length of a sub-yearling salmon relative to the mean length of other competing sub-yearling salmon of the same species in the system, Equation D, and relates that relative difference to size-dependent survival based upon Equation S. The values for α and resulting size-dependent survival (survival ϕ) for control runs for each species are listed in Table 2. The constant α is a species-specific parameter defined such that it produces the correct control survival ϕ value when Δlength equals zero.

Equation D: Δlength = fish length(mm) – mean length(mm)

Equation S: Survival $\phi = (e^{(\alpha+(0.0329*\Delta\text{length}))}) / (1 + e^{(\alpha+(0.0329*\Delta\text{length}))})$

Randomly selecting length values from the normal distribution calculated from the organismal model output size and applying equations 1 and 2 generates a size-dependent survival probability for each fish. This process was replicated 1,000 times for each exposure scenario and simultaneously 1,000 times for the paired control scenario and results in a mean size-dependent survival rate for each population. The resulting size-dependent survival rates are inserted in the calculation of first-year survival in the respective control and pesticide-exposed transition matrices.

The investigation of population-level responses to pesticide exposures uses life history projection matrix models. Individuals within a population exhibit various growth, reproduction, and survivorship rates depending on their developmental or life history stage or age. These age specific characteristics are depicted in the life history graph (Figure 1A-D) in which transitions are depicted as arrows. The nonzero matrix elements represent transitions corresponding to reproductive contribution or survival, located in the top row and the subdiagonal of the matrix, respectively (Figure 1E). The survival transitions in the life history graph are incorporated into the n x n square matrix (A) by assigning each age a number (1 through n) and each transition from age i to age j becomes the element a_{ij} of matrix A (i = row, j = column) and represent the proportion of the individuals in each age passing to the next age as a result of survival. The reproductive element (a_{1j}) gives the number of offspring that hatch per individual in the contributing age, j. The reproductive element value incorporates the proportion of females in each age, the proportion of females in the age that are sexually mature, fecundity, fertilization success, and hatch success.

In order to understand the relative impacts of a short-term exposure of a single pesticide on exposed vs. unexposed fish, we used parameters for an idealized control population that exhibits an increasing population growth rate. All characteristics exhibit density independent dynamics. The models assume closed systems, allowing no migration impact on population size. No stochastic impacts are included beyond natural variability as represented by selecting parameter values from a normal distribution about a mean each model iteration (year). Ocean conditions, freshwater habitat, fishing pressure, and marine resource availability were assumed constant and density independent.

In the model an individual fish experiences an exposure once as a sub-yearling (during its first spring) and never again. The pesticide exposure is assumed to occur annually. All sub-yearlings within a given population are assumed to be exposed to the pesticide. No other age classes experience the exposure. The model integrates this as every brood class being exposed as sub-yearlings and thus the vital demographic rates of the transition matrix are continually impacted in the same manner. Regardless of the level of AChE inhibition due to the direct exposure, only the sublethal effects are incorporated in the models at this time.

The model recalculates first-year survival for each run using a size-dependent survival value selected from a normal distribution with the mean and standard deviation produced by Equation S. Population model output consists of the percent change in lambda from the unexposed control populations derived from the mean of two thousand calculations of both the unexposed control population and the pesticide exposed population. Change in lambda, representing alterations to the population productivity, was selected as the primary model output for reasons outlined previously.

A prospective analysis of the transition matrix, A, (Caswell 2001) explored the intrinsic population growth rate as a function of the vital rates. The intrinsic population growth rate, λ, equals the dominant eigenvalue of A and was calculated using matrix analysis software (MATLAB version 7.7.0 by The Math Works Inc., Natick, MA). Therefore, λ is calculated directly from the matrix and running projections of abundances over time is redundant and unnecessary. The stable age distribution, the proportional distribution of individuals among the ages when the population is at equilibrium, is calculated as the right normalized eigenvector corresponding to the dominant eigenvalue λ. Variability was integrated by repeating the calculation of λ 2,000 times selecting the values in the transition matrix from their normal distribution defined by the mean standard deviation. The influence of each matrix element, a_{ij}, on λ was assessed by calculating the sensitivity values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to a_{ij}, defined by $\delta\lambda / \delta a_{ij}$. Higher sensitivity values indicate greater influence on λ. The elasticity of matrix element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. One characteristic of elasticity analysis is that the elasticity values for a transition matrix sum to unity (one). The unity characteristic also allows comparison of the influence of transition elements and comparison across matrices.

Due to differences in the life history strategies, specifically lifespan, age at reproduction and first year residence and migration habits, four life history models were constructed. This was done to encompass the different responses to freshwater pesticide exposures and assess potentially different population-level responses. Separate models were constructed for coho, sockeye, ocean-type and stream-type Chinook salmon. In all cases transition values were determined from literature data on survival and reproductive characteristics of each species.

A life history model was constructed for coho salmon (*O. kisutch*) with a maximum age of 3. Spawning occurs in late fall and early winter with emergence from March to May. Fry spend 14-18 months in freshwater, smolt and spend 16-20 months in the saltwater before returning to spawn (Pess et al. 2002). Survival numbers were summarized in Knudsen et al. (2002) as follows. The average fecundity of each female is 4,500 with a standard deviation of 500. The observed number of males:females was 1:1. Survival from spawning to emergence is 0.3 (0.07). Survival from emergence to smolt is 0.0296 (0.00029) and marine survival is 0.05 (0.01). All parameters followed a normal distribution (Knudson et al. 2002). The calculated values used in the matrix are listed in Table 4. The growth period for first year coho was set at 180 days to represent the time from mid-spring to mid-fall when the temperatures and resources drop and somatic growth slows (Knudson et al. 2002).

Life history models for sockeye salmon (*O. nerka*) were based upon the lake wintering populations of Lake Washington, Washington, USA. These female sockeye salmon spend one winter in freshwater, then migrate to the ocean to spend three to four winters before returning to spawn at ages 4 or 5. Jacks return at age 2 after only one winter in the ocean. The age proportion of returning adults is 0.03, 0.82, and 0.15 for ages 3, 4 and 5, respectively (Gustafson et al.1997). All age 3 returning adults are males. Hatch rate and first-year survival were calculated from brood year data on escapement, resulting presmolts and returning adults (Pauley et al. 1989), and fecundity (McGurk 2000). Fecundity values for age 4 females were 3,374 (473) and for age 5 females were 4,058 (557) (McGurk 2000). First-year survival rates were 0.737/month (Gustafson et al. 1997). Ocean survival rates were calculated based upon brood data and the findings that 90% of ocean mortality occurs during the first 4 months of ocean residence (Pauley et al. 1989). Matrix values used in the sockeye baseline model are listed in Table 4. The 168 day growth period represents the time from lake entry to early fall when the temperature drops and somatic growth slows (Gustafson et al. 1997).

A life history model was constructed for ocean-type Chinook salmon (*O. tshawytscha*) with a maximum female age of 5 and reproductive maturity at ages 3, 4, or 5. Ocean-type Chinook salmon migrate from their natal stream within a couple months of hatching and spend several months rearing in estuary and nearshore habitats before continuing on to the open ocean. Transition values were determined from literature data on survival and reproductive characteristics from several ocean-type Chinook salmon populations in the Columbia River system (Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004). The sex ratio of spawners was approximately 1:1. Estimated size-based fecundity of 4,511(65), 5,184(89), and 5,812(102) was calculated based on data from Howell et al., 1985, using length-fecundity relationships from Healy and Heard (1984). Control matrix values for the Chinook model are listed in Table 4. The growth period of 140 days encompasses the time the fish rear in freshwater prior to entering the estuary and open ocean. The first three months of estuary/ocean survival are the size-dependent stage. Size data for determining sub-yearling Chinook salmon condition indices came from data collected in the lower Columbia River and estuary (Johnson et al. 2007).

An age-structured life history matrix model for stream-type Chinook salmon with a maximum age of 5 was defined based upon literature data on Yakima River spring Chinook from Knudsen et al. (2006) and Fast et al. (1988), with sex ratios of 0.035, 0.62 and 0.62 for females spawning at ages 3, 4, and 5, respectively. Length data from Fast et al. (1988) was used to calculate fecundity from the length-fecundity relationships in Healy and Heard (1984). The 184-day growth period produces control fish with a mean size of 96 mm, within the observed range documented in the fall prior to the first winter (Beckman et al. 2000). The size-dependent survival encompasses the 4 early winter months, up until the fish are 12 months old.

Acute Toxicity Model

In order to estimate the population-level responses of exposure to lethal pesticide concentrations, acute mortality models were constructed based upon the control life history matrices described above. The acute responses are modeled as direct reduction in the first-year survival rate (S1). All sub-yearling salmon are assumed to be exposed in each scenario. Exposures are assumed to result in a cumulative reduction in survival as defined by the concentration and the dose-response curve as defined by the LC50 and slope for each pesticide. A sigmoid dose-response relationship is used to accurately handle responses well away from LC50 and to be consistent with other does-response relationships. The model inputs for each scenario are the exposure concentration and acute fish LC50, as well as the sigmoid slope for the LC50. For a given concentration a pesticide survival rate (1-mortality) is calculated and is multiplied by the control first-year survival rate, producing an exposed scenario first-year survival for the life history matrix. Variability is incorporated as described above using mean and standard deviation of normally distributed survival and reproductive rates and model output consists of the percent change in lambda from unexposed control populations derived from the mean of 1,000 calculations of both the unexposed control population and the pesticide exposed population. The percent change in lambda is considered different from control when the difference is greater than the percent of one standard deviation from the control lambda.

Results

Sensitivity Analysis

A sensitivity analysis conducted on the organismal model revealed that changes in the control somatic growth rate had the greatest influence on the final weights (Table 1). While this parameter value was experimentally derived for another species (sockeye salmon; Brett et al. 1969), this value was adapted for each model species and is within the variability reported in the literature for other salmonids (reviewed in Weatherley and Gill 1995). Other parameters related to the daily growth rate calculation, including the growth to ration slope (Mgr) and the control ration produced strong sensitivity values. Initial weight, the prey recovery rate and the prey floor also strongly influenced the final weight values (Table 1). Large changes (0.5 to 2X) in the other key parameters produced proportionate changes in final weight.

The sensitivity analysis of all four of the control population matrices predicted the greatest changes in population growth rate (λ) result from changes in first-year survival. Parameter values and their corresponding sensitivity values are listed in Table 4. The elasticity values for the transition matrices also corresponded to the driving influence of first-year survival, with contributions to lambda of 0.33 for coho, 0.29 for ocean-type Chinook salmon, 0.25 for stream-type Chinook salmon, and 0.24 for sockeye.

Model Output

Organismal and population model outputs for all scenarios are shown in Tables 74-78 and were summarized in Figures 44-47 in the main text of the Opinion. As expected, greater changes in population growth resulted from longer exposures to the pesticides. The factors driving the level of change in lambda were the Prey Drift and relative AChE Activity parameters determined by the toxicity values for each pesticide (Table 3). The low Prey Abundance EC_{50} values drive the effects for all three compounds at most concentrations investigated since they have higher AChE EC50 values (Table 3). Output from the acute toxicity models was presented in the Risk Characterization section of the main text. Increases in direct mortality during the first year of life produced large impacts on the population growth rates for all the life history strategies.

While strong trends in effects were seen for each pesticide across all four life history strategies modeled, some slight differences were apparent. The similarity in patterns likely stems from using the same toxicity values for all four models, while the differences are consequences of distinctions between the life history matrices. The stream-type Chinook salmon and sockeye models produced very similar results as measured as the percent change in population growth rate. The ocean-type Chinook salmon and coho models output produced the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output. Combining these factors into the transition matrix for each life history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. In addition, the elasticity analysis can be used to predict relative contribution to lambda from changes in first-year survival on a per unit basis. As detailed by the elasticity values reported above, the same change in first-year survival will produce a slightly greater change in the population growth rate for coho and ocean-type Chinook salmon than for stream-type Chinook salmon and sockeye. While some life history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

Figure 1: Life-History Graphs and Transition Matrix for coho (A), sockeye (B) and Chinook salmon (C). The life history graph for a population labeled by age, with each transition element labeled according to the matrix position, a_{ij}, i row and j column. Dashed lines represent reproductive contribution and solid lines represent survival transitions. D) The transition matrix for the life history graph depicted in C.

Figure 2: Relationships used to link anticholinesterase exposure to the organism's ability to acquire food (potential ration). See text for details. Relationships in B, C, and D use empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either a single compound or mixtures). B) Sigmoidal relationship between exposure concentration and steady-state AChE activity showing a dose-dependent reduction defined by control activity (horizontal line, Ac), sigmoidal (*i.e.*, hille) slope (AChE slope), and the concentration producing 50% inhibition (vertical line, EC50). C) Timecourse of AChE inhibition based on modeling the time-to-effect and time-to-recovery as single exponential curves with different time-constants. At the start of the exposure AChE activity will be at control and then decline toward the inhibited activity (Ai) based on Panel B. D) Linear model relating AChE activity to feeding behavior using a line that passes through the feeding (Fc) and activity (Ac) control conditions with a slope of Mfa. E) The relationship between feeding behavior and the potential ratio an organism could acquire (if not food limited) used a line passing through the control conditions (Fc as in Panel D and the control ration, Rc) and through the origin producing a slope (Mrf) equal to Rc/Fc. F) Timecourse for effect of exposure to anticholinesterase on potential ration produced by combining C and E.

Figure 3: Relationships used to link anticholinesterase exposure to the availability of prey. See text for details. Relationships in B and C utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either single compound or mixtures). B) Sigmoidal relationship between exposure concentration and relative prey abundance showing a dose-dependent reduction defined by control abundance (horizontal line at 1, Pc), sigmoid (*i.e.*, hille) slope (prey slope), the concentration producing a 50% reduction in prey (vertical line,

EC_{50}), and a minimum abundance always present (horizontal line denoted as floor, Pf). C) Timecourse of prey abundance including a 1-day spike in prey drift relative to the available prey and the level of toxicity followed by a drop to the level of impact or the floor whichever is greater. During extended exposures at low toxicity recovery can begin at the constant prey influx rate multiplied by the current level of toxicity. After exposure recovery to control prey drift is at the constant rate of influx from upstream habitats.

Figure 4: Relationships used to link anticholinesterase exposure to growth rate relating to long-term weight gain of each fish. See text for details. Relationships in A, B, and C utilize empirical data. Closed circles represent control conditions. Open circles (*e.g.*, Ai) represent the exposed (inhibited) condition. A&B) Relationships describing the timecourse of the effects of anticholinesterase exposure on the organisms ability to capture food (Panel A, potential ration) and the availability of food to capture (Panel B, relative prey abundance). The figures are the same as those in Figures 2F and 3C, respectively. For a given exposure concentration and time, multiplying potential ration by relative prey abundance yields the final ration acquired by the organism. C) A linear model was used to relate final ration to growth rate using a line passing through the control conditions and through the maintenance condition with a slope denoted by Mgr. D) Timecourse for effect of exposure to anticholinesterase on growth rate produced by combining A, B, & C.

Table 1. List of values used for control parameters to model organismal growth and the model sensitivity to changes in the parameter.

Parameter	Value[1]	Error[2]	Sensitivity[3]
acetylcholinesterase activity (Ac)	1.0[4,5]	0.06[5]	-0.167
feeding (Fc)	1.0[4,5]	0.05[5]	0.088
ration (Rc)	5% weight/day[6]	0.05[7]	-0.547
feeding vs. activity slope (Mfa)	1.0[5]	0.1[5]	-0.047
ration vs. feeding slope (Mrf)	5 (Rc/Fc)	-	-
growth vs. ration slope (Mgr)	0.35[6]	0.02[6]	-0.547
growth vs. activity slope (Mga)	1.75 (Mfa*Mrf*Mgr)	-	-
initial weight	1 gram[8]	0.1[8]	1.00
control prey drift	1.0[4]	0.05[11]	0.116
AChE impact time-to-effect ($t_{1/2}$)	0.5 day[9]	n/a	0.005
AChE time-to-recovery ($t_{1/2}$)	0.25 days[10]	n/a	-0.0001
prey floor	0.20[11]	n/a	0.178
prey recovery rate	0.01[12]	n/a	0.323
somatic growth rate (Gc)	1.3[13]	0.06[6]	2.531

[1] mean value of a normal distribution used in the model or constant value when no corresponding error is listed

[2] standard deviation of the normal distribution used in the model

[3] mean sensitivity when baseline parameter is changed over range of 0.5 to 2-fold

[4] other values relative to control

[5] derived from Sandahl et al., (2005)

[6] derived from Brett et al., (1969)

[7] data from Brett et al., (1969) has no variability (ration was the independent variable) so a variability of 1% was selected to introduce some variability

[8] consistent with field-collected data for juvenile Chinook salmon (Nelson et al., 2004)

[9] estimated from Ferrari et al., 2004

[10] consistent with Labenia et al., 2007

[11] estimated from Van den Brink et al., 1996

[12] derived from Ward et al., 1995, Van den Brink et al., 1996, Colville et al., 2008

[13] derived from Brett and adapted for ocean-type Chinook salmon, used for sensitivity analysis

Table 2. Species specific control parameters to model organismal growth and survival rates. Growth period and survival rate are determined from the literature data listed for each species. Gc and α were calculated to make the basic model produce the appropriate size and survival values from the literature.

	Chinook Stream-type[1]	Chinook Ocean-type[2]	Coho[3]	Sockeye[4]
days to run organismal growth model	184	140	184	168
growth rate % body wt/day (Gc)	1.28	1.30	0.90	1.183
α from equation S	-0.33	-1.99	-0.802	-0.871
Control Survival ϕ	0.418	0.169	0.310	0.295

[1] Values from data in Healy and Heard 1984, Fast et al., 1988, Beckman et al., 2000, Knudsen et al., 2006
[2] Values from data in Healey and Heard 1984, Howell et al., 1985, Roni and Quinn 1995, Ratner et al.,1997, PSCCTC 2002, Green and Beechie, 2004, Johnson et al., 2007
[3] Values from data in Pess et al., 2002, Knudsen et al., 2002
[4] Values from data in Pauley et al., 1989, Gustafson et al., 1997, McGurk 2000

Table 3. Effects values (μg/L) and slopes for AChE activity, acute fish lethality, and prey abundance dose-response curves.

compound	AChE Activity EC_{50}[1] g/L	AChE Activity slope	Fish lethality LC_{50}[2] g/L	Fish lethality slope[3]	Prey Abundance EC_{50}[4] g/L	Prey Abundance Slope[5]
carbaryl	145.8	0.95	250	3.63	4.33	5.5
carbofuran	58.4	0.95	164	3.63	1.22	5.5
methomyl	213	0.95	560	3.63	20.74	5.5

[1] Values from Laetz et. al 2009
[2] Values from EPA BEs
[3] sigmoidal slope that produces responses with a probit slope of Peterson, see text.
[4] Values from analysis of global search of reported LC50 and EC50s reported in EPA's Ecotox database. See text.
[5] Values from Peterson et al., 2001

Table 4. Matrix transition element and sensitivity (S) and elasticity (E) values for each model species. These control values are listed by the transition element taken from the life history graphs as depicted in Figure 1 and the literature data described in the method text. Blank cells indicate elements that are not in the transition matrix for a particular species. The influence of each matrix element on λ was assessed by calculating the sensitivity (S) and elasticity (E) values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to the transition element, defined by δλ/ δa. The elasticity of transition element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. Elasticity values allow comparison of the influence of individual transition elements and comparison across matrices.

Transition Element	Chinook Stream-type			Chinook Ocean-type			Coho			Sockeye		
	Value[1]	S	E	Value[2]	S	E	Value[3]	S	E	Value[4]	S	E
S1	0.0643	3.844	0.247	0.0056	57.13	0.292	0.0296	11.59	0.333	0.0257	9.441	0.239
S2	0.1160	2.132	0.247	0.48	0.670	0.292	0.0505	6.809	0.333	0.183	1.326	0.239
S3	0.17005	1.448	0.246	0.246	0.476	0.106				0.499	0.486	0.239
S4	0.04	0.319	0.0127	0.136	0.136	0.0168				0.1377	0.322	0.0437
R3	0.5807	0.00184	0.0011	313.8	0.0006	0.186	732.8	0.000469	0.333			
R4	746.73	0.000313	0.233	677.1	0.000146	0.0896				379.57	0.000537	0.195
R5	1020.36	1.25E-05	0.0127	1028	1.80E-05	0.0168				608.7	7.28E-05	0.0437

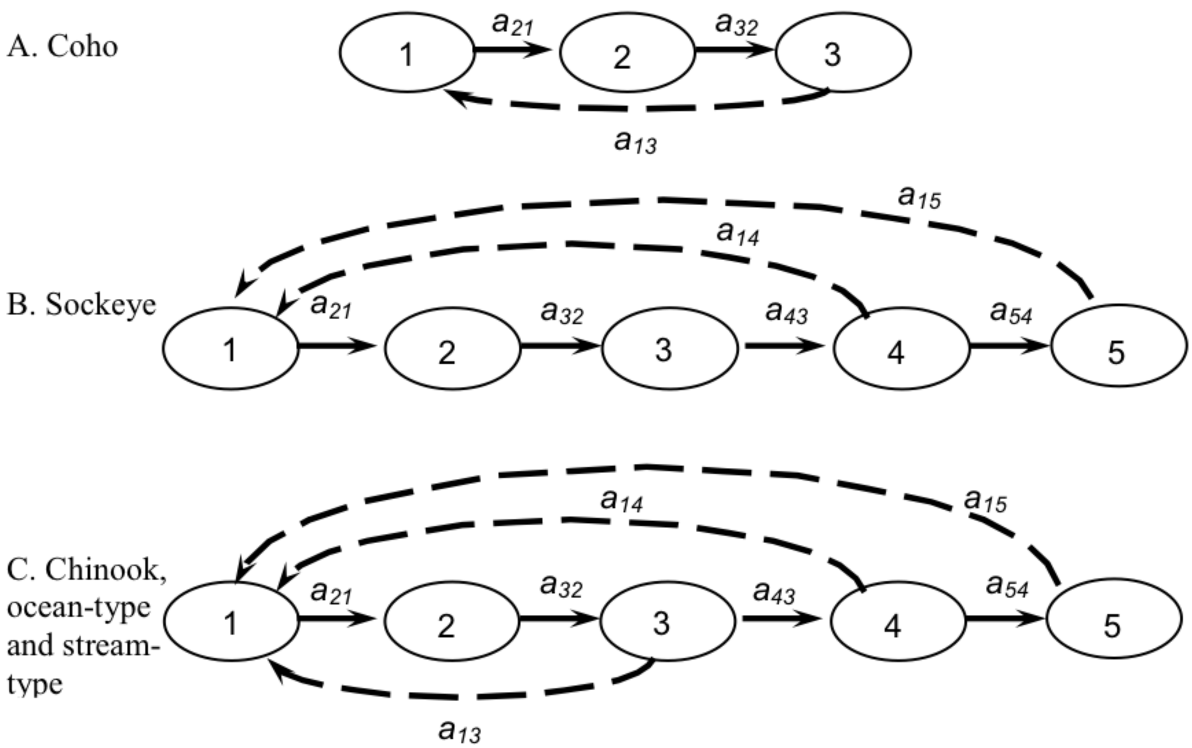
[1] Value calculated from data in Healy and Heard 1984, Fast et al., 1988, Beckman et al., 2000, Knudsen et al., 2006

[2] Value calculated from data in Healey and Heard 1984, Howell et al., 1985, Roni and Quinn 1995, Ratner et al., 1997, PSCCTC 2002, Green and Beechie, 2004, Johnson et al., 2007

[3] Value calculated from data in Pess et al., 2002, Knudsen et al., 2002

[4] Value calculated from data in Pauley et al., 1989, Gustafson et al., 1997, McGurk 2000

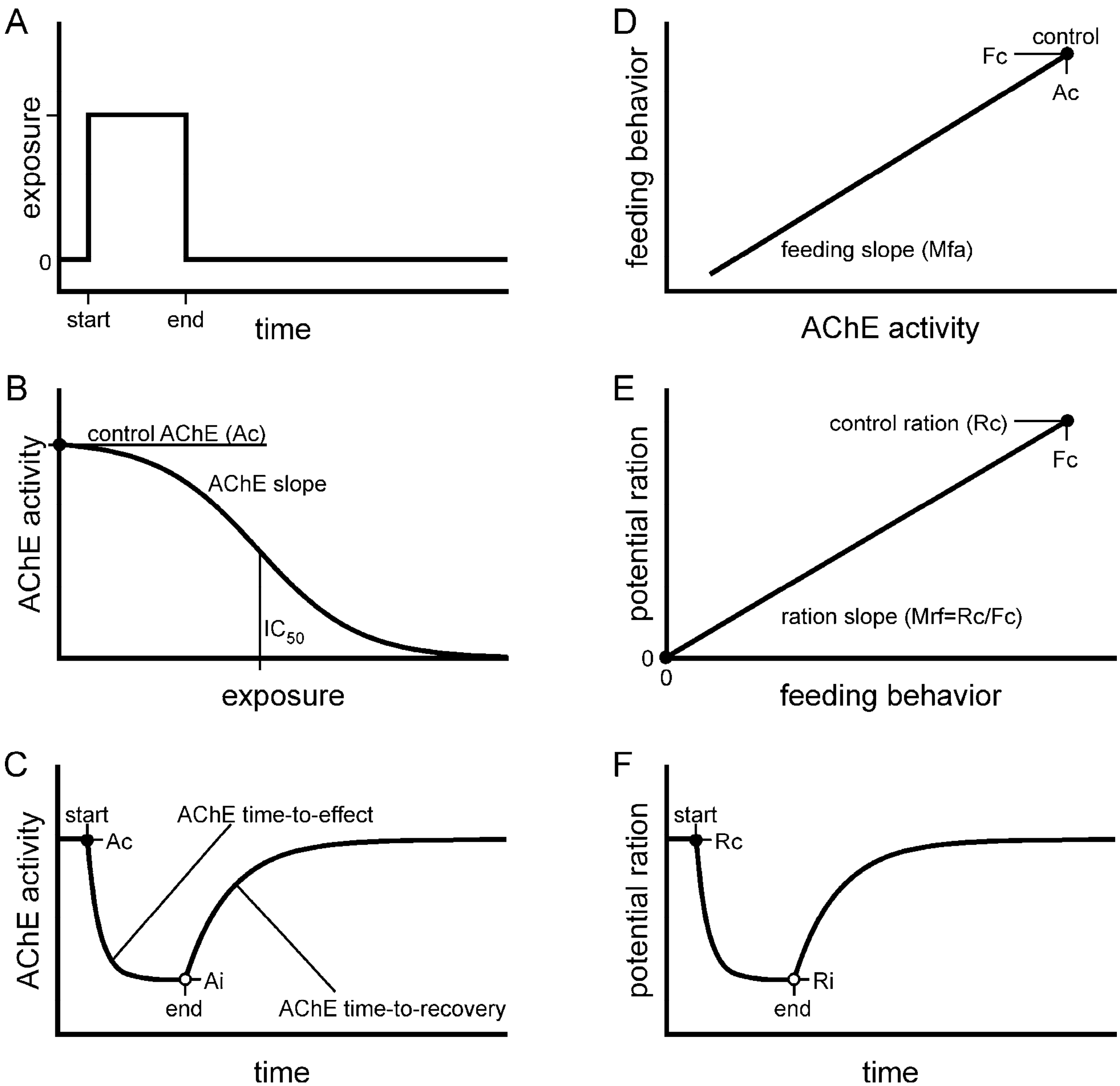
Figure 1.

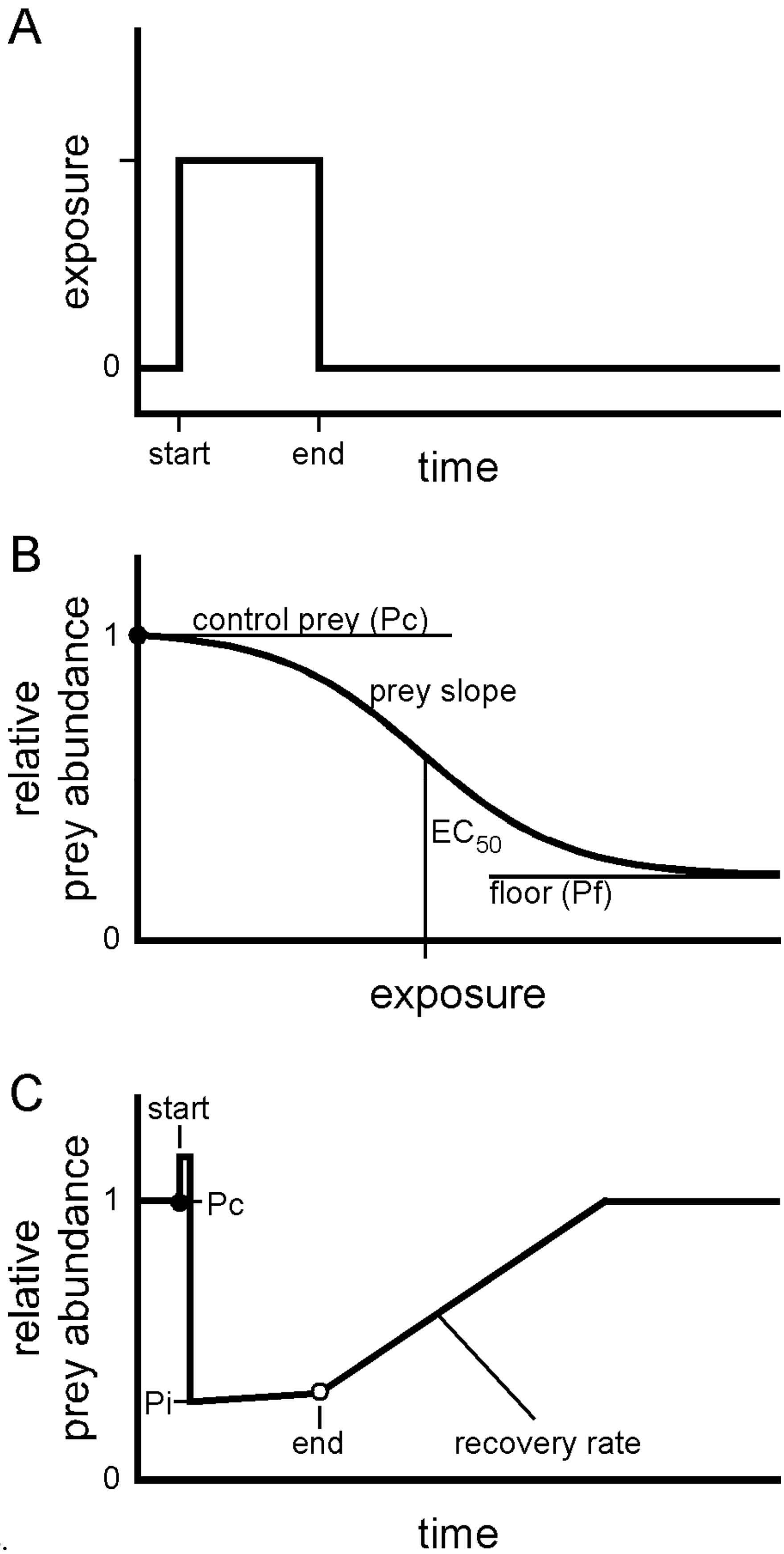


D. Transition Matrix

$$A=\begin{bmatrix} 0 & 0 & R3=a_{13} & R4=a_{14} & R5=a_{15} \\ S1=a_{21} & 0 & 0 & 0 & 0 \\ 0 & S2=a_{32} & 0 & 0 & 0 \\ 0 & 0 & S3=a_{43} & 0 & 0 \\ 0 & 0 & 0 & S4=a_{54} & 0 \end{bmatrix}$$

Figure 2.


A
exposure
0
start
end
time
B
AChE activity
control AChE (Ac)
AChE slope
IC50
exposure
C
AChE activity
start
Ac
AChE time-to-effect
Ai
end
AChE time-to-recovery
time
D
feeding behavior
control
Fc
Ac
feeding slope (Mfa)
AChE activity
E
potential ration
control ration (Rc)
Fc
ration slope (Mrf=Rc/Fc)
0
0
feeding behavior
F
potential ration
start
Rc
Ri
end
time



A
exposure
0
start
end
time
B
relative
prey abundance
1
0
control prey (Pc)
prey slope
EC50
floor (Pf)
exposure
C
start
relative
prey abundance
1
Pc
Pi
end
recovery rate
0
time

Figure 3.

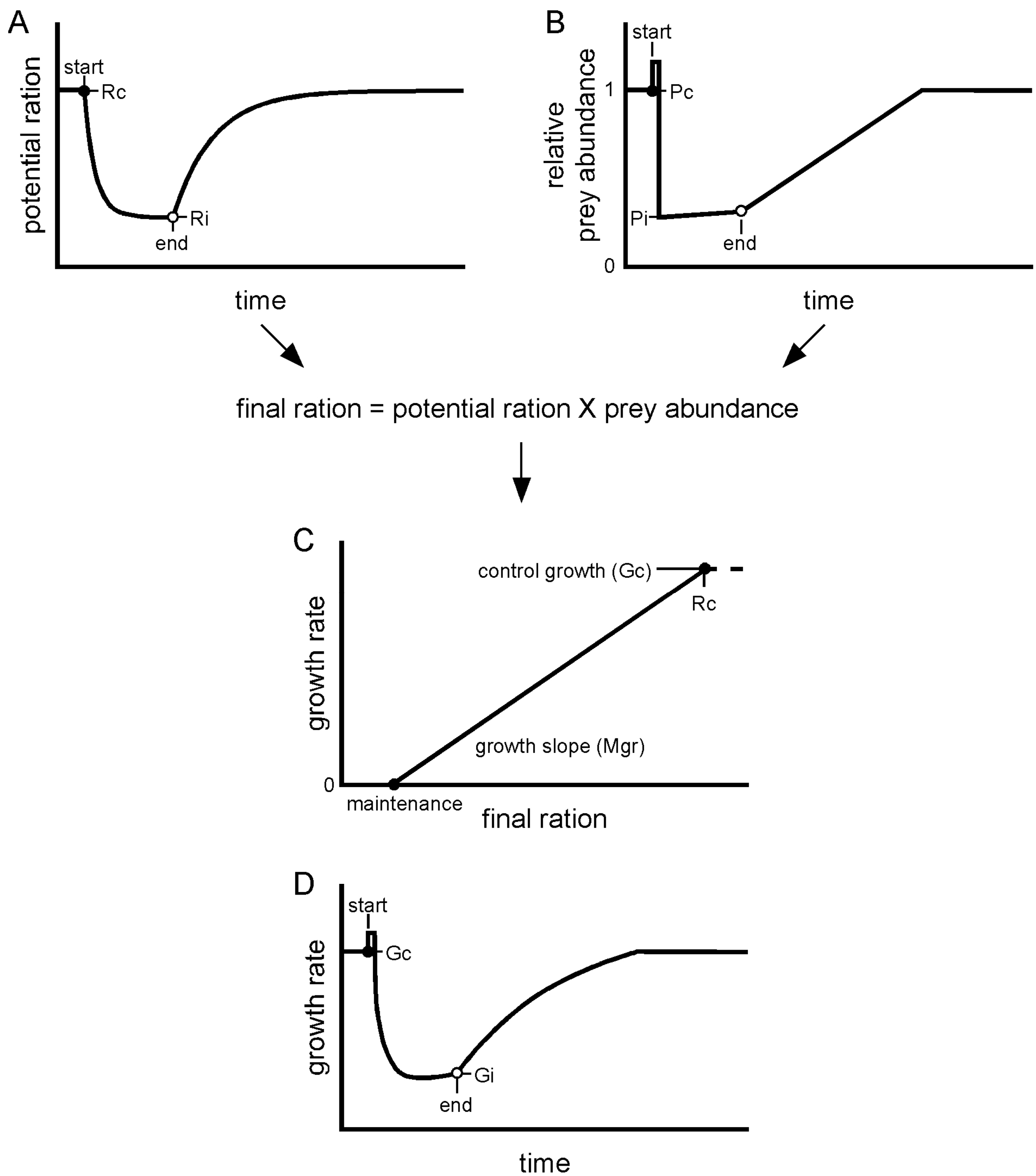

A
potential ration
start
Rc
Ri
end
time
B
relative
prey abundance
start
1
Pc
Pi
0
end
time
final ration = potential ration X prey abundance
C
growth rate
control growth (Gc)
Rc
growth slope (Mgr)
0
maintenance
final ration
D
growth rate
start
Gc
Gi
end
time

Figure 4.

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Appendix 2. Species and Population Annual Rates of Growth

Chinook Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
California Coastal	Eel River	N/A	N/A	N/A
	Redwood Creek	N/A	N/A	N/A
	Mad River	N/A	N/A	N/A
	Humboldt Bay tributaries	N/A	N/A	N/A
	Bear River	N/A	N/A	N/A
	Mattole River	N/A	N/A	N/A
	Tenmile to Gualala	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
Central Valley Spring - Run (Good et al., 2005 - 90% CI)	Butte Creek - spring run	1.300	1.060	1.600
	Deer Creek - spring run	1.170	1.040	1.350
	Mill Creek - spring run	1.190	1.000	1.470
Lower Columbia River (Good et al., 2005) (# = McElhany et al., 2007)	Youngs Bay	N/A	N/A	N/A
	Grays River - fall run	0.944	0.739	1.204
	Big Creek	N/A	N/A	N/A
	Elochoman River - fall run	1.037	0.813	1.323
	Clatskanie River #	0.990	0.824	1.189
	Mill, Abernathy, Germany Creeks - fall run	0.981	0.769	1.252
	Scappose Creek	N/A	N/A	N/A
	Coweeman River - fall run	1.092	0.855	1.393
	Lower Cowlitz River - fall run	0.998	0.776	1.282
	Upper Cowlitz River - fall run	N/A	N/A	N/A
	Toutle River - fall run	N/A	N/A	N/A
	Kalamaha River - fall run	0.937	0.763	1.242
	Salmon Creek / Lewis River - fall run	0.984	0.771	1.256
	Clackamas River - fall run	N/A	N/A	N/A
	Washougal River - fall run	1.025	0.803	1.308
	Sandy River - fall run	N/A	N/A	N/A
	Lower Gorge tributaries	N/A	N/A	N/A
	Upper Gorge tributaries - fall run	0.959	0.751	1.224
	Hood River - fall run	N/A	N/A	N/A
	Big White Salmon River - fall run	0.963	0.755	1.229
	Sandy River - late fall run	0.943	0.715	1.243
	North Fork Lewis River - late fall run	0.968	0.756	1.204
	Upper Cowlitz River - spring run	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Toutle River - spring run	N/A	N/A	N/A
	Kalamaha River - spring run	N/A	N/A	N/A
	Lewis River - spring run	N/A	N/A	N/A
	Sandy River - spring run #	0.961	0.853	1.083
	Big White Salmon River - spring run	N/A	N/A	N/A
	Hood River - spring run	N/A	N/A	N/A

Chinook Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Upper Columbia River Spring - Run (FCRPS)	Methow River	1.100	N/A	N/A
	Twisp River	N/A	N/A	N/A
	Chewuch River	N/A	N/A	N/A
	Lost / Early River	N/A	N/A	N/A
	Entiat River	0.990	N/A	N/A
	Wenatchee River	1.010	N/A	N/A
	Chiawawa River	N/A	N/A	N/A
	Nason River	N/A	N/A	N/A
	Upper Wenatchee River	N/A	N/A	N/A
	White River	N/A	N/A	N/A
	Little Wenatchee River	N/A	N/A	N/A
Puget Sound (only have λ where hatchery fish = native fish), (Good et al., 2005)	Nooksack - North Fork	0.750	0.680	0.820
	Nooksack - South Fork	0.940	0.880	0.990
	Lower Skagit	1.050	0.960	1.140
	Upper Skagit	1.050	0.990	1.110
	Upper Cascade	1.060	1.010	1.110
	Lower Sauk	1.010	0.890	1.130
	Upper Sauk	0.960	0.900	1.020
	Suiattle	0.990	0.930	1.050
	Stillaguamish - North Fork	0.920	0.880	0.960
	Stillaguamish - South Fork	0.990	0.970	1.010
	Skykomish	0.870	0.840	0.900
	Snoqualmie	1.000	0.960	1.040
	North Lake Washington	1.070	1.000	1.140
	Cedar	0.990	0.920	1.060
	Green	0.670	0.610	0.730
	White	1.160	1.100	1.220
	Puyallup	0.950	0.890	1.010
	Nisqually	1.040	0.970	1.110
	Skokomish	1.040	1.000	1.080
	Dosewallips	1.170	1.070	1.270
	Duckabush	N/A	N/A	N/A
	Hamma Hamma	N/A	N/A	N/A
	Mid Hood Canal	N/A	N/A	N/A
	Dungeness	1.090	0.980	1.200
	Elwha	0.950	0.840	1.060
Sacramento River Winter - Run (Good, 2005 - 90% CI))	Sacramento River - winter run	0.970	0.870	1.090

Chinook Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Snake River Fall - Run (Good, 2005)	Lower Snake River	1.024	N/A	N/A
Snake River Spring/Summer - Run (FCRPS)	Tucannon River	1.000	N/A	N/A
	Wenaha River	1.100	N/A	N/A
	Wallowa River	N/A	N/A	N/A
	Lostine River	1.050	N/A	N/A
	Minam River	1.050	N/A	N/A
	Catherine Creek	0.970	N/A	N/A
	Upper Grande Ronde River	N/A	N/A	N/A
	South Fork Salmon River	1.110	N/A	N/A
	Secesh River	1.070	N/A	N/A
	Johnson Creek	N/A	N/A	N/A
	Big Creek Spring Run	1.090	N/A	N/A
	Big Creek Summer Run	1.090	N/A	N/A
	Loon Creek	N/A	N/A	N/A
	Marsh Creek	1.080	N/A	N/A
	Bear Valley / Elk Creek	1.100	N/A	N/A
	North Fork Salmon River	N/A	N/A	N/A
	Lemhi River	1.020	N/A	N/A
	Pahsimeroi River	1.080	N/A	N/A
	East Fork Salmon Spring Run	1.040	N/A	N/A
	East Fork Salmon Summer Run	1.040	N/A	N/A
	Yankee Fork Spring Run	N/A	N/A	N/A
	Yankee Fork Summer Run	N/A	N/A	N/A
	Valley Creek Spring Run	N/A	N/A	N/A
	Valley Creek Summer Run	N/A	N/A	N/A
	Upper Salmon Spring Run	1.060	N/A	N/A
	Upper Salmon Summer Run	1.060	N/A	N/A
	Alturas Lake Creek	N/A	N/A	N/A
	Imnaha River	1.050	N/A	N/A
	Big Sheep Creek	N/A	N/A	N/A
	Lick Creek	N/A	N/A	N/A
Upper Williamette River (McElhany et al., 2007)	Clackamas River	0.967	0.849	1.102
	Molalla River	N/A	N/A	N/A
	North Santiam River	N/A	N/A	N/A
	South Santiam River	N/A	N/A	N/A
	Calapooia River	N/A	N/A	N/A
	McKenzie River	0.927	0.761	1.129
	Middle Fork Williamette River	N/A	N/A	N/A
	Upper Fork Williamette River	N/A	N/A	N/A

Chum Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Columbia River	Youngs Bay	N/A	N/A	N/A
	Grays River	0.954	0.855	1.064
	Big Creek	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy and German Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cowlitz River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	Lewis River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Clackamus River	N/A	N/A	N/A
	Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Gorge tributaries	0.984	0.883	1.096
	Upper Gorge tributaries	N/A	N/A	N/A
Hood Canal Summer - Run (only have λ where hatchery fish reproductive potential = native fish; Good et. al., 2005)	Jimmycomelately Creek	0.850	0.690	1.010
	Salmon / Snow Creeks	1.230	1.130	1.330
	Big / Little Quilcene rivers	1.390	1.170	1.610
	Lilliwaup Creek	1.190	0.750	1.630
	Hamma Hamma River	1.300	1.110	1.490
	Duckabush River	1.100	0.930	1.270
	Dosewallips River	1.170	0.930	1.410
	Union River	1.150	1.050	1.250
	Chimacum Creek	N/A	N/A	N/A
	Big Beef Creek	N/A	N/A	N/A
	Dewetto Creek	N/A	N/A	N/A

Coho Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Central California Coast	Ten Mile River	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Other Mendacino County Rivers	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
	Other Sonoma County Rivers	N/A	N/A	N/A
	Martin County	N/A	N/A	N/A
	San Mateo County	N/A	N/A	N/A
	Santa Cruz County	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
Lower Columbia River (Good et al., 2005)	Youngs Bay	N/A	N/A	N/A
	Grays River	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy, Germany Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	North Fork Toutle River	N/A	N/A	N/A
	South Fork Toutle River	N/A	N/A	N/A
	Coweeman River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	North Fork Lewis River	N/A	N/A	N/A
	East Fork Lewis River	N/A	N/A	N/A
	Upper Clackamas River	1.028	0.898	1.177
	Lower Clackamas River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Upper Sandy River	1.102	0.874	1.172
	Lower Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Columbia River gorge tributaries	N/A	N/A	N/A
	White Salmon	N/A	N/A	N/A
	Upper Columbia River gorge tributaries	N/A	N/A	N/A
	Hood River	N/A	N/A	N/A

Coho Salmon (continued)

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Southern Oregon and Northern California Coast	Southern Oregon and Northern California Coast	N/A	N/A	N/A
Oregon Coast	Necanicum	N/A	N/A	N/A
	Nehalem	N/A	N/A	N/A
	Tillamook	N/A	N/A	N/A
	Nestucca	N/A	N/A	N/A
	Siletz	N/A	N/A	N/A
	Yaquima	N/A	N/A	N/A
	Alsea	N/A	N/A	N/A
	Siuslaw	N/A	N/A	N/A
	Umpqua	N/A	N/A	N/A
	Coos	N/A	N/A	N/A
	Coquille	N/A	N/A	N/A

Sockeye Salmon

ESU	Population	λ - H=0	95% CI -lower	95% CI - upper
Ozette Lake	Ozette Lake	N/A	N/A	N/A
Snake River	Snake River	N/A	N/A	N/A

Steelhead

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Central California Coast (Good et al., 2005)	Russain River	N/A	N/A	N/A
	Lagunitas	N/A	N/A	N/A
	San Gregorio	N/A	N/A	N/A
	Waddell Creek	N/A	N/A	N/A
	Scott Creek	N/A	N/A	N/A
	San Vincente Creek	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
	Soquel Creek	N/A	N/A	N/A
	Aptos Creek	N/A	N/A	N/A
California Central Valley (Good et al., 2005)	Sacramento River	0.950	0.900	1.020
Lower Columbia River (Good et al., 2005)	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	Coweeman River	0.908	0.792	1.041
	South Fork Toutle River	0.938	0.830	1.059
	North Fork Toutle River	1.062	0.915	1.233
	Kalama River - winter run	1.010	9.130	1.117
	Kalama River - summer run	0.981	0.889	1.083
	North Fork Lewis River - winter run	N/A	N/A	N/A
	North Fork Lewis River - summer run	N/A	N/A	N/A
	East Fork Lewis River - winter run	N/A	N/A	N/A
	East Fork Lewis River - summer run	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Washougal River - winter run	N/A	N/A	N/A
	Washougal River - summer run	1.003	0.884	1.138
	Clackamas River	0.971	0.901	1.047
	Sandy River	0.945	0.850	1.051
	Lower Columbia gorge tributaries	N/A	N/A	N/A
	Upper Columbia gorge tributaries	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Middle Columbia River (Good et al., 2005)	Klickitat River	N/A	N/A	N/A
	Yakima River	1.009	N/A	N/A
	Fifteenmile Creek	0.981	N/A	N/A
	Deschutes River	1.022	N/A	N/A
	John Day - upper main stream	0.975	N/A	N/A
	John Day - lower main stream	0.981	N/A	N/A
	John Day - upper north fork	1.011	N/A	N/A
	John Day - lower north fork	1.013	N/A	N/A
	John Day - middle fork	0.966	N/A	N/A
	John Day - south fork	0.967	N/A	N/A
	Umatilla River	1.007	N/A	N/A
	Touchet River	0.961	N/A	N/A
Northern California (Good et al., 2005)	Redwood Creek	N/A	N/A	N/A
	Mad River - winter run	1.000	0.930	1.050
	Eel River - summer run	0.980	0.930	1.040
	Mattole River	N/A	N/A	N/A
	Ten Mile river	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Other Humboldt County streams	N/A	N/A	N/A
	Other Mendocino County streams	N/A	N/A	N/A
Puget Sound*	Puget Sound	N/A	N/A	N/A
Snake River (Good et al., 2005)	Tucannon River	0.886	N/A	N/A
	Lower Granite run	0.994	N/A	N/A
	Snake A run	0.998	N/A	N/A
	Snake B run	0.927	N/A	N/A
	Asotin Creek	N/A	N/A	N/A
	Upper Grande Ronde River	0.967	N/A	N/A
	Joseph Creek	1.069	N/A	N/A
	Imnaha River	1.045	N/A	N/A
	Camp Creek	1.077	N/A	N/A
South-Central California Coast	South-Central California Coast	N/A	N/A	N/A
Southern California	Santa Ynez River	N/A	N/A	N/A
	Ventura River	N/A	N/A	N/A
	Matilija River	N/A	N/A	N/A
	Creek River	N/A	N/A	N/A
	Santa Clara River	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	λ - H=0	95% CI -lower	95% CI - upper
Upper Columbia River (Good et al., 2005)	Wenatchee / Entiat Rivers	1.067	N/A	N/A
	Methow / Okanogan Rivers	1.086	N/A	N/A
Upper Williamette River (McElhany et al., 2007)	Molalla River	0.988	0.790	1.235
	North Santiam River	0.983	0.789	1.231
	South Santiam River	0.976	0.855	1.114
	Calapooia River	1.023	0.743	1.409

Appendix 3: Abbreviations

7-DADMax	7-day average of the daily maximum
ACA	Alternative Conservation Agreement
AChE	acetylcholinesterase
a.i.	active ingredient
APEs	alkylphenol ethoxylates
APHIS	U.S. Department of Agriculture Animal Plant and Health Inspection Service
BE	Biological Evaluation
BEAD	Biological and Economic Analysis Divsion
BLM	Bureau of Land Management
BMP	Best Management Practices
BOR	Bureau of Reclaimation
BOR	Bureau of Reclamation
BPA	Bonneville Power Administration
BRT	Biological Review Team (NOAA Fisheries)
BY	Brood Years
CAISMP	California Aquatic Invasive Species Management Plan
CALFED	CALFED Bay-Delta Program (California Resource Agency)
CBFWA	Columbia Basin Fish and Wildlife Authority
CBI	Confidential Business Information
CC	California Coastal
CCC	Central California Coast
CCV	Central California Valley
CDPR	California Department of Pesticide Regulation
CHART	Critical Habitat Assessment Review Team
CIDMP	Comprehensive Irrigation District Management Plan
CFR	Code of Federal Regulations
cfs	cubic feet per second
CDFG	California Department of Fish and Game
Corps	U.S. Department of the Army Corps of Engineers

CSOs	combined sewer/stormwater overflows
CSWP	California State Water Project
CURES	Coalition for Urban/Rural Environmental Stewardship
CVP	Central Valley Projects
CVRWQCB	Central Valley Regional Water Quality Control Board
CWA	Clean Water Act
d	day
DCI	Date Call-Ins
DDD	Dichloro Diphenyl Dichloroethane
DDE	Diphenyl Dichlorethylene
DDT	Dichloro Diphenyl Trichloroethane
DER	Data Evaluation Review
DEQ	Oregon Department of Environmental Quality
DIP	Demographically Independent Population
DOE	Washington State Department of Ecology
DPS	Distinct Population Segment
EC	Emulsifiable Concentrate Pesticide Formulation
EC_{50}	Median Effect Concentration
EEC	Estimated Environmental Concentration
EFED	Environmental Fate and Effects Division
EIM	Environmental Information Management
EPA	U.S. Environmental Protection Agency
ESPP	Endangered Species Protection Program
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
EU	European Union
EXAMS	Tier II Surface Water Computer Model
FERC	Federal Energy Regulatory Commission
FCRPS	Federal Columbia River Power System
FFDCA	Federal Food and Drug Cosmetic Act
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act

FQPA	Food Quality Protection Act
ft	feet
GENEEC	Generic Estimated Exposure Concentration
h	hour
HCP	Habitat Conservation Plan
HSRG	Hatchery Scientific Review Group
HUC	Hydrological Unit Code
IBI	Indices of Biological Integrity
ICTRT	Interior Columbia Technical Recovery Team
ILWP	Irrigated Lands Waiver Program
IPCC	Intergovernmental Panel on Climate Change
IRED	Interim Re-registration Decision
LCFRB	Lower Columbia Fish Recovery Board
ISG	Independent Science Group
ITS	Incidental Take Statement
km	kilometer
Lbs	Pounds
LC_{50}	Median Lethal Concentration.
LCR	Lower Columbia River
LOAEC	Lowest Observed Adverse Effect Concentration.
LOEL	Lowest Observed Adverse Effect level
LOC	Level of Concern
LOEC	Lowest Observed Effect Concentration
LOQ	Limit of Quantification
LWD	Large Woody Debris
m	meter
MCR	Middle Columbia River
mg/L	milligrams per liter
MOA	Memorandum of Agreement
MPG	Major Population Group
MRID	Master Record Identification Number

MTBE	Methyl tert-butyl ether
NASA	National Aeronautics and Space Administration
NAWQA	U.S. Geological Survey National Water-Quality Assessment
NC	Northern California
NEPA	National Environmental Protection Agency
NLCD	Natural Land Cover Data
NP	Nonylphenol
NPDES	National Pollutant Discharge Elimination System
NPS	National Parks Services
NRCS	Natural Resources Conservation Service
NWS	National Weather Service
NEPA	National Environmental Policy Act
NMA	National Mining Association
NMC	*N*-methyl carbamates
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEC	No Observed Adverse Effect Concentration
NPDES	National Pollution Discharge Eliminating System
NPIRS	National Pesticide Information Retrieval System
NRC	National Research Council
OC	Oregon Coast
ODFW	Oregon Division of Fish and Wildlife
OP	Organophosphates
Opinion	Biological Opinion
OPP	EPA Office of Pesticide Program
PAH	polyaromatic hydrocarbons
PBDEs	polybrominated diphenyl ethers
PCBs	polychlorinated biphenyls
PCEs	primary constituent elements
POP	Persistent Organic Pollutants
ppb	Parts Per Billion

PPE	Personal Protection Equipment
PSP	Pesticide Stewardship Partnerships
PSAMP	Puget Sound Assessment and Monitoring Program
PSAT	Puget Sound Action Team
PRIA	Pesticide Registration Improvement Act
PRZM	Pesticide Root Zone Model
PUR	Pesticide Use Reporting
QA/QC	Quality Assurance/Quality Control
RED	Re-registration Eligibility Decision
REI	Restricted Entry Interval
RPA	Reasonable and Prudent Alternatives
RPM	reasonable and prudent measures
RQ	Risk Quotient
SAP	Scientific Advisory Panel
SAR	smolt-to-adult return rate
SASSI	Salmon and Steelhead Stock Inventory
SC	Southern California
S-CCC	South-Central California Coast
SONCC	Southern Oregon Northern California Coast
SLN	Special Local Need (Registrations under Section 24(c) of FIFRA)
SR	Snake River
TCE	Trichloroethylene
TCP	3,5,6-trichloro-2-pyridinal
TGAI	Technical Grade Active Ingredient
TIE	Toxicity Identification Evaluation
TMDL	Total Maximum Daily Load
TRT	Technical Recovery Team
UCR	Upper Columbia River
USFS	United States Forest Service
USC	United States Code
USFWS	United States Fish and Wildlife Service

USGS	United States Geological Survey
UWR	Upper Willamette River
VOC	Volatile Organic Compounds
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
WLCRTRT	Willamette/Lower Columbia River Technical Review Team
WQS	Water Quality Standards
WWTIT	Western Washington Treaty Indian Tribes
WWTP	Wastewater Treatment Plant

Appendix 4: Glossary

303(d) waters Section 303 of the federal Clean Water Act requires states to prepare a list of all surface waters in the state for which beneficial uses – such as drinking, recreation, aquatic habitat, and industrial use - are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that do not meet the state's surface water quality standards and are not expected to improve within the next two years. After water bodies are put on the 303(d) list they enter into a Total Maximum Daily Load Clean Up Plan.

Active ingredient	The component(s) that kills or otherwise affects the pest. A.i.s are always listed on the label (FIFRA 2(a)).
Adulticide	A compound that kills the adult life stage of the pest insect.
Anadromous Fish	Species that are hatched in freshwater migrate to and mature in salt water and return to freshwater to spawn.
Adjuvant	A compound that aides the operation or improves the effectiveness of a pesticide.
Alevin	Life history stage of a salmonid immediately after hatching and before the yolk-sac is absorbed. Alevins usually remain buried in the gravel in or near the egg nest (redd) until their yolk sac is absorbed when they swim up and enter the water column.
Anadromy	The life history pattern that features egg incubation and early juvenile development in freshwater migration to sea water for adult development, and a return to freshwater for spawning.
Assessment Endpoint	Explicit expression of the actual ecological value that is to be protected (*e.g.*, growth of juvenile salmonids).

Bioaccumulation	Accumulation through the food chain (*i.e.*, consumption of food, water/sediment) or direct water and/or sediment exposure.
Bioconcentration	Uptake of a chemical across membranes, generally used in reference to waterborne exposures.
Biomagnification	Transfer of chemicals via the food chain through two or more trophic levels as a result of bioconcentration and bioaccumulation.
Degradates	New compounds formed by the transformation of a pesticide by chemical or biological reactions.
Distinct Population Segment	A listable entity under the ESA that meets tests of discreteness and significance according to USFWS and NMFS policy. A population is considered distinct (and hence a "species" for purposes of conservation under the ESA) if it is discrete fro an significant to the remainder of its species based n factors such as physical, behavioral, or genetic characteristics, it occupies an unusual or unique ecological setting, or its loss would represent a significant gap in the species' range.
Escapement	The number of fish that survive to reach the spawning grounds or hatcheries. The escapement plus the number of fish removed by harvest form the total run size.
Evolutionarily Significant Unit	A group of Pacific salmon or steelhead trout that is (1) substantially reproductively isolated from other conspecific units and (2) represent an important component of the evolutionary legacy of the species.
Fall Chinook	This salmon stock returns from the ocean in late summer and early

Salmon	fall to head upriver to its spawning grounds, distinguishing it from other stocks which migrate in different seasons.
Fate	Dispersal of a material in various environmental compartments (sediment, water air, biota) as a result of transport, transformation, and degradation.
Flowable	A pesticide formulation that can be mixed with water to form a suspension in a spray tank.
Fry	Stage in salmonid life history when the juvenile has absorbed its yolk sac and leaves the gravel of the redd to swim up into the water column. The fry stage follows the alevin stage and in most salmonid species is followed by the parr, fingerling, and smolt stages. However, chum salmon juveniles share characteristics of both the fry and smolt stages and can enter sea water almost immediately after becoming fry.
Half-pounder	A life history trait of steelhead exhibited in the Rogue, Klamath, Mad, and Eel Rivers of southern Oregon and northern California. Following smoltification, half-pounders spend only 2-4 months in the ocean, then return to fresh water. They overwinter in fresh water and emigrate to salt water again the following spring. This is often termed a false spawning migration, as few half-pounders are sexually mature.
Hatchery	Salmon hatcheries use artificial procedures to spawn adults and raise the resulting progeny in fresh water for release into the natural environment, either directly from the hatchery or by transfer into another area. In some cases, fertilized eggs are outplanted (usually in “hatch-boxes”), but it is more common to release fry or smolts.
Inert ingredients	“an ingredient which is not active” (FIFRA 2(m)). It may be toxic or enhance the toxicity of the active ingredient.

Iteroparous	Capable of spawning more than once before death
Jacks	Male salmon that return from the ocean to spawn one or more years before full-sized adults return. For coho salmon in California, Oregon, Washington, and southern British Columbia, jacks are 2 years old, having spent only 6 months in the ocean, in contrast to adults, which are 3 years old after spending 1 ½ years in the ocean.
Jills	Female salmon that return from the ocean to spawn one or more years before full-sized adult returns. For sockeye salmon in Oregon, Washington, and southern British Columbia, jills are 3 years old (age 1.1), having spent only one winter in the ocean in contrast to more typical sockeye salmon that are age 1.2, 1.32.2, or 2.3 on return.
Kokanee	The self-perpetuating, non-anadromous form of *O. nerka* that occurs in balanced sex ration populations and whose parents, for several generations back, have spent their whole lives in freshwater.
Lambda	Also known as Population growth rate, or the rate at which the abundance of fish in a population increases or decreases.
LRL	Laboratory Reporting Level (USGS NAWQA data)- Generally equal to twice the yearly determined LT-MDL. The LRL controls false negative error. The probability of falsely reporting a non-detection for a sample that contained an analyte at a concentration equal to or greater that the LRL is predicted to be less than or equal to 1 percent.
Major Population Group (MPG)	A group of salmonid populations that are geographically and genetically cohesive. The MPG is a level of organization between demographically independent populations and the ESU.

Main channel	The stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel.
Metabolite	A transformation product resulting from metabolism.
Mode of Action	A series of key processes that begins with the interaction of a pesticide with a receptor site and proceeds through operational and anatomical changes in an organisms that result in sublethal or lethal effects.
Natural fish	A fish that is produced by parents spawning in a stream or lake bed, as opposed to a controlled environment such as a hatchery.
Nonylphenols	A type of APE and is an example of an adjuvant that may be present as an ingredient of a formulated product or added to a tank mix prior to application.
Off-channel habitat	Water bodies and/or inundated areas that are connected (accessible to salmonid juveniles) seasonally or annually to the main channel of a stream including but not limited to features such as side channels, alcoves, ox bows, ditches, and floodplains.
Parr	The stage in anadromous salmonid development between absorption of the yolk sac and transformation to smolt before migration seaward.
Persistence	The tendency of a compound to remain in its original chemical form in the environment.
Pesticide	Any substance or mixture of substances intended for preventing, destroying, repelling or mitigating any pest.

Reasonable and Prudent Alternative (RPA)	Recommended alternative actins identified during formal consultation that can be implemented in a manner consistent with the scope of the Federal agency's legal authority an jurisdiction, that are economically an technologically feasible, an that the Services believes would avoid the likelihood of jeopardizing the continued existence of the listed species or the destruction or adverse modification of designated critical habitat.
Redd	A nest constructed by female salmonids in streambed gravels where eggs are deposited and fertilization occurs.
Riparian area	Area with distinctive soils an vegetation between a stream or other body of water and the adjacent upland. It includes wetlands and those portions of flood plains an valley bottoms that support riparian vegetation.
Risk	The probability of harm from actual or predicted concentrations of a chemical in the aquatic environment – a scientific judgement.
Salmonid	Fish of the family *Salmonidae*, including salmon, trout, chars, grayling, and whitefish. In general usage, the term usually refers to salmon, trout, and chars.
SASSI	A cooperative program by WDFW and WWTIT to inventory and evaluate the status of Pacific salmonids in Washington State. The SASSI report is a series of publications from this program.
Semelparous	The condition in an individual organism of reproducing only once in a lifetime.

Smolt	A juvenile salmon or steelhead migrating to the ocean and undergoing physiological changes to adapt from freshwater to a saltwater environment.
Sublethal	Below the concentration that directly causes death. Exposure to sublethal concentrations of a material may produce less obvious effect on behavior, biochemical, and/or physiological function of the organism often leading to indirect death.
Surfactant	A substance that reduces the interfacial or surface tension of a system or a surface-active substance.
Synergism	A phenomenon in which the toxicity of a mixture of chemicals is greater than that which would be expected from a simple summation of the toxicities of the individual chemicals present in the mixture.
Technical Grade Active Ingredient (TGAI)	Pure or almost pure active ingredient. Available to formulators. Most toxicology data are developed with the TGAI. The percent AI is listed on all labels.
Technical Recovery Teams (TRT)	Teams convened by NOAA Fisheries to develop technical products related to recovery planning. TRTs are complemented by planning forums unique to specific states, tribes, or reigns, which use TRT and other technical products to identify recovery actions.
Teratogenic	Effects produced during gestation that evidence themselves as altered structural or functional processes in offspring.
Total Maximum	defines how much of a pollutant a water body can tolerate (absorb)

Daily Load (TMDL)	daily and remain compliant with applicable water quality standards. All pollutant sources in the watershed combined, including non-point sources, are limited to discharging no more than the TMDL.
Unique Mixture	A specific combination of 2 or more compounds, regardless of the presence of other compounds.
Viable Salmonid Population	An independent population of Pacific salmon or steelhead trout that has a negligible risk of extinction over a 100-year time frame. Viability at the independent population scale is evaluated based on the parameters of abundance, productivity, spatial structure, and diversity.
VSP Parameters	Abundance, productivity, spatial structure, and diversity. These describe characteristics of salmonid populations that are useful in evaluating population viability. See NOAA Technical Memorandum NMFS-NWFSC-, "Viable salmonid populations and the recovery of evolutionarily significant units," McElhany et al., June 2000.
WDFW	Washington Department of Fish and Wildlife is a co-manager of salmonids and salmonid fisheries in Washington State with WWTIT and other fisheries groups. The agency was formed in the early 1990s by the combination of the Washington Department of Fisheries and the Washington Department of Wildlife.
WWTIT	Western Washington Treaty Indian Tribes is an organization of Native American tribes with treaty fishing rights recognized by the U.S. government. WWTIT is a co-manager of salmonids and salmonid fisheries in western Washington in cooperation with the WDFW and other fisheries groups.

WQS	“A water quality standard defines the water quality goals of a waterbody, or portion thereof, by designating the use or uses to be made of the water and by setting criteria necessary to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water Act.” Each state is responsible for maintaining water quality standards.

DRAFT
Endangered Species Act Section 7 Consultation
Biological Opinion & Conference Opinion
On the
U.S. Environmental Protection Agency's
Approval of State or Tribal, or Federal Numeric Water Quality Standards for Cyanide
Based on EPA's Recommended 304(a) Aquatic Life Criteria

National Marine Fisheries Service
Office of Protected Resources
Silver Spring, MD 20910

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National Marine Fisheries Service
Endangered Species Act Section 7 Consultation
Biological Opinion & Conference Opinion

Agency: U.S. Environmental Protection Agency

Activities Considered: Approval of State or Tribal, or Federal Numeric Water Quality Standards for Cyanide Based on EPA's Recommended 304(a) Aquatic Life Criteria

Consultation Conducted by: Endangered Species Division of the Office of Protected Resources, National Marine Fisheries Service

Approved by: ____

Date: ____

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1539(a)(2)) requires each federal agency to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a federal agency's action "may affect" a protected species, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (FWS; together, the Services), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR 402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS or the FWS concur with that conclusion (50 CFR 402.14(b)).

The U.S. Environmental Protection Agency (EPA) initiated formal consultation with NMFS and the FWS on their recommended 304(a) criteria and the approval of state and tribal water quality standards, or federal water quality standards promulgated by EPA that are identical to or more stringent than the section 304(a) aquatic life criteria published pursuant to the Clean Water Act (CWA; 33 U.S.C. 1251 *et seq*.), for the protection of aquatic life from harmful effects of cyanide (CN). This document represents NMFS' biological and conference opinion (Opinion) on EPA's approval of numeric standards for cyanide in fresh and salt waters of the U.S and its effects on threatened and endangered species, their designated critical habitat, and species proposed for listing as threatened or endangered, and critical habitat proposed for designation. This consultation does not address the effects of specific modifications of these criteria that are undertaken by states and tribes or the permits issued by particular states or tribes. This Opinion contains a detailed explanation of the particular circumstances warranting subsequent consultation (tiered consultations) with NMFS' Regional Offices in the section titled *Application* *of this Consultation to Other EPA Actions*.

This Opinion is based on our review of the EPA's *Biological Evaluation of Aquatic Life Criteria- Cyanide*, status reviews, listing documents, and recovery plans for the threatened and endangered species under NMFS' jurisdiction, reports on the status and trends of water quality in the United States that have been prepared by the U.S. Geological Survey, EPA, and others, past and current research and population dynamics modeling efforts, and published and unpublished scientific information on the biology and ecology of threatened and endangered sea turtles, marine mammals, salmon, sturgeon, sawfish, abalone, and seagrasses in the action area, and other sources of information which are discussed in greater detail in the *Approach to the Assessment* section of this Opinion.. This Opinion has been prepared in accordance with section 7 of the ESA and associated implementing regulations.

Consultation History

On January 18, 2001, the Services and EPA signed a Final Memorandum of Agreement (MOA) on the enhanced coordination under the ESA and the CWA. The final MOA published in the *Federal Register* on February 22, 2001 (66 FR 36) and described, among other things, a plan for assisting EPA in meeting it's section 7 responsibilities on two CWA programs: water quality standards and the National Pollutant Discharge Elimination System (NPDES) permits program.

In January 2004, the Services and EPA decided to proceed with a data call for the first batch of pollutants that would be reviewed in consultation, while continuing to work on the *Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria--Methods Manual.*

On May 14, 2004, the Services and EPA issued data calls to the regional staff and science center staff requesting information and data on cyanide, ammonia, chromium III and chromium VI. The data call requested regions and science centers send relevant studies to Headquarters by June 30, 2004.

On November 12, 2004, EPA provided the Services a revised *Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria--Methods Manual* (dated October 29, 2004, on the document). This version represented a methodology developed collaboratively, and which had been peer reviewed by subject experts outside of the Federal government.

In December 2004, NMFS and EPA exchanged comments on recommended revisions to the November draft methodology. EPA also informed the Services that they had received a draft BE for cyanide from their contractor and were reviewing the document to ensure the contractor had followed the BE methodology accurately.

On January 24, 2005, EPA emailed NMFS a partial draft of their CN BE.

On May 3, 2005, the Services jointly provided comments to EPA on their January 19, 2005, draft biological evaluation for cyanide criteria.

On January 26, 2006, EPA provided NMFS with a draft CN BE and requested a review of the

BE’s “completeness” in fulfilling the information requirements for section 7 consultation. On April 21, 2006, NMFS provided comments to EPA on the “completeness” of the draft BE.

In a June 29, 2006, letter, EPA requested NMFS’ concurrence with their determination that proposed approval of cyanide criteria was not likely to adversely affect all listed species and critical habitat under NMFS’ jurisdiction.

On November 11, 2006, the FWS sent NMFS a copy of EPA’s revised *Draft Framework for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual*, which EPA revised and submitted to FWS in July 31, 2006 and which EPA used to support their effects determinations.

On November 15, 2006, NMFS sent EPA a letter with a detailed explanation as to why NMFS could not concur with EPA’s determinations that the recommended water quality standards for cyanide “may affect, but are not likely to adversely affect” threatened and endangered species and designated critical habitat.

On March 23, 2007, EPA requested formal consultation supported by their March 23, 2007, *Biological Evaluation of Aquatic Life Criteria—Cyanide*, which concluded their action was “not likely to jeopardize the continued existence of any federally listed species or result in the destruction or adverse modification of designated critical habitat [*sic*].”

On June 21, 2007, NMFS sent a letter to EPA acknowledging the initiation of formal consultation. NMFS’ letter acknowledged that the scope and complexity of the national consultation on the aquatic life criteria for cyanide may require more time than usual to complete the biological opinion.

On May 5-9, 2008, the Services met with EPA to conduct a “Kaizen” “lean event.” The purpose of the meetings was to analyze the cyanide consultation process from the development of a biological assessment through the anticipated completion of formal consultation in an effort to find efficiencies in the process. The Services and EPA also discussed coordination and communication with respect to the national consultation on cyanide and local consultation on EPA promulgation of Oregon water quality standards.

On June 12, 2008, the Services and EPA met to follow up on the Kaizen lean event. Subsequent follow up meetings were cancelled until the Services completed draft biological opinions.

BIOLOGICAL & CONFERENCE OPINION

Description of the Proposed Action

The action considered in this Opinion, and beginning a series of national water quality consultations with EPA, is EPA's continuing approval of state or tribal water quality standards, or federal water quality standards promulgated by EPA, that are identical to or more stringent than EPA's recommended 304(a) aquatic life criteria for cyanide. These water quality standards define water column concentrations of cyanide that should protect against adverse ecological effects to aquatic life in fresh and salt water. The 304(a) aquatic life criteria recommendations, which are the foundation for many approved 303(c) standards, are designed to protect aquatic organisms from unacceptable toxicity during acute (short) and chronic (long) exposures in the water column. The intent is to define a level in the waterbody of a pollutant that will be fully protective of the designated use and which a regulatory authority may use in adopting regulatory water quality standards and thereby control, reduce, or eliminate discharges of that pollutant (BE page 11).

Section 304(a)(1) of the CWA directs EPA to publish criteria for water quality accurately reflecting the latest scientific knowledge on a number of factors including "… the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, esthetics, and recreation which may be expected from the presence of pollutants in any body of water, including ground water; on the concentration and dispersal of pollutants, or their byproducts, through biological, physical and chemical processes; and on the effects of pollutants on biological community diversity, productivity, and stability including information on the factors affecting rates of eutrophication and rates of organic and inorganic sedimentation for varying types of receiving waters." The water quality standards program is authorized under section 303(c) of the CWA (33 U.S.C. 1313(c)) and directs states to adopt numeric criteria for specific toxic pollutants that appear on a priority pollutant list[1] and for which EPA published 304(a) criteria recommendations. States can, pursuant to section 303(c) of the CWA, adopt water quality standards that differ from EPA's 304(a) criteria values whenever adequately justified, but states and tribes generally choose to adopt EPA's 304(a) criteria verbatim. Once adopted into state water quality standards, criteria form the legal basis for implementing the CWA programs to control pollution and achieve the goals and requirements of the CWA.

The purpose of these national consultations is to assess the effect of the EPA's 304(a) criteria recommendation and the subsequent approval of state and tribal water quality standards, or federal water quality standards promulgated by EPA that are identical to or more stringent than the section 304(a) aquatic life criteria on threatened and endangered species and their designated critical habitat (together, listed resources), and species and critical habitat that are proposed for

[1] Section 307(a) of the CWA, which defines priority pollutants as compounds and families that are among the most persistent, prevalent and toxic chemicals.

listing or designation (together, proposed resources). In particular, this Opinion analyzes whether EPA's approval of state standards that rely on the national criteria for cyanide are not likely to jeopardize the continued existence of threatened and endangered species (including species proposed for listing as threatened or endangered), or result in the destruction or adverse modification of designated critical habitat (see the BE, page 1).

In 1985 EPA published two values for cyanide pollution in each fresh and salt "waters of the United States," the criterion maximum concentration (CMC) and the criterion continuous concentration (CCC). EPA's ambient water quality criteria for cyanide are expressed as free cyanide (Table 1). The CMC represents EPA's estimate of the highest concentration of cyanide in fresh or salt water to which an aquatic community's brief exposure (acute limit) would not result in an unacceptable effect. The CMC is derived from a set of LC50 values for a variety of aquatic species. The LC50 value is the lethal concentration of a chemical that causes 50% mortality, immobilization, or loss of equilibrium in the test organism in 48 to 96-hour laboratory tests. The CMC is then set to one-half of the fifth percentile of the genus mean acute value (GMAV) for the various species tested to provide a level of protection that is better than 50% mortality. EPA recommends that the one-hour average exposure concentrations should not exceed the CMC more than once every three years on the average, making such exceedances a relatively rare event (EPA 1991).

Table 1. Cyanide 304(a) Aquatic Life Criteria (in μg/L of free cyanide [EPA 1985])

Medium	Criterion Maximum Concentration	Criterion Continuous Concentration
Fresh water	22.36	5.221
Saltwater	1.015	1.015

The CCC represents EPA's estimate of the highest concentration of cyanide in either fresh or salt water, to which an aquatic community's prolonged exposure (chronic limit) would not result in an unacceptable effect. The CCC is derived from a set of chronic values, which are the geometric mean of the highest no observed effect concentrations (NOECs) and lowest observed effect concentrations (LOECs) for survival, growth, or reproduction in tests that range from seven days to several months. EPA sets the CCC to the estimated fifth percentile of the chronic values either by direct calculation or by using the acute-to-chronic ratios (ACRs). For the CCC, EPA recommends that the four-day average exposure concentrations should not exceed the CCC more frequently than once every three years on average (EPA 1991).

Approach to the Assessment

Section 7(a)(2) of the ESA of 1973, as amended (16 U.S.C. 1536(a)(2)), requires federal agencies, in consultation with and with the assistance of the Services, to ensure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of endangered species or threatened species or result in the destruction or adverse modification of designated critical habitat. When NMFS consults with federal agencies to help them comply with this requirement, we first assess the direct and indirect effects of the proposed federal action to determine whether the proposal is likely to (a) appreciably increase a species' extinction probability (or reduce their probability of being conserved or recovered) or (b) appreciably reduce the conservation value of critical habitat that has been designated for one or more of those species. If we conclude that one of these outcomes is likely, we work with the federal agency, applicant, or both, to develop alternatives that avoid this likelihood.

NMFS approaches its section 7 analyses through a series of steps. The first step identifies those aspects of proposed actions that are likely to have individual, interactive, or cumulative direct and indirect effects on the environment (the potential stressors of an action). As part of this step, we identify the spatial extent of these stressors, including changes in their spatial extent over time. The spatial extent of these stressors represents the *Action Area* for consultation.

To begin the second step of our analyses, we determine whether endangered species, threatened species or designated critical habitat are likely to occur in the same space and the same time as the potential stressors. These species then become the focus of our *Exposure Analysis*. As our point of reference for evaluating the risk posed by their exposure, we rely on our understanding of the condition of the species and the conservation value of critical habitat, and any biological and ecological information on the species and their critical habitat that is relevant to our effects analysis (this information is represented in the *Status of the Species and Critical Habitat*). In the status of the species section of our Opinion, we review the species' legal status, trends, and the threats that led to this status as well as those that may be impeding the species' chances of recovery. Our assessment is also informed by the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat, and ecosystem. This information is presented in the *Environmental Baseline*. By regulation, the environmental baseline for an action includes the past and present impacts of all federal, state, or private actions and other human activities in an action area, and the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions that are contemporaneous with the consultation in process. The environmental baseline is designed to assess the condition of the habitat and the species within the action area.

Often, NMFS will combine the status of the species and the environmental baseline where the status encompasses the entire range of a species. In this Opinion, we address the two separately, focusing the environmental baseline on the current condition of the nation's fresh water and marine aquatic habitats. In some cases we address watersheds that may not contain listed species under NMFS' jurisdiction because the watershed influences coastal conditions where listed marine and anadromous species occur. Our summary of the environmental baseline complements the information provided in the status of the species section of this Opinion, and provides information on the past and present ecological conditions of the action area that is necessary to further understand the species' current risk of extinction.

Our effects analyses, summarized in the *Effects of the Action* section of this Opinion, identify the nature of the listed species and critical habitat co-occurrence with the effects of the action over space and time (their exposure). Our exposure analyses identify the physical or biological features of critical habitat that would be exposed to the action, including any listed primary constituent elements that require special management consideration or protection such as sites for breeding and rearing, food, water, space for growth and normal behavior, and cover and shelter; and we identify the number, age or life stage, and gender of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent. Once we identify the individuals and populations, or constituent laments that are likely to be exposed to an action's effects and the nature of that exposure, we examine the scientific and commercial data available to determine whether and how those listed species and their critical habitat (collectively termed listed resources) are likely to respond given their exposure (these represent our *response analyses*). The final steps of our analyses—establishing the risks those responses pose to listed resources—are different for listed species and designated critical habitat (these represent our *risk analyses*).

Our jeopardy determinations must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been listed, which can include the biological species, subspecies, or distinct population segments of vertebrate species. Because the continued existence of listed species depends on the fate of the populations that comprise them, the viability (probability of extinction or probability of persistence) of listed species depends on the viability of the populations that comprise them. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them; populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so). Our risk analyses reflect the relationships between the listed species and the populations that comprise them, and the individuals that comprise those populations. Our risk analyses begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individuals' risks to identify consequences to the populations they represent and next we determine the consequences of population-level effects on the species as listed.

We measure risks to listed individuals using the individual's "fitness," which are changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable responses to an action's effect on the environment (which we identify during our response analyses) are likely to have consequences for the individual's fitness. When individual listed plants or animals are expected to experience reductions in fitness, we would expect those reductions to also reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of the populations those individuals represent (see Stearns 1992). A reduction in one or more of these variables (or one of the variables we derive from them) is a *necessary* condition for reductions in a population's viability, which itself is a *necessary* condition for reductions in a species' viability. On the other hand, when listed plants or animals exposed to an action's effects are not expected to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise (for example, see Anderson 2000, Mills and Beatty 1979, Stearns 1992). If we conclude that listed plants or animals are not likely to experience reductions in their fitness we would conclude our assessment.

If, however, we conclude that listed plants or animals are likely to experience reductions in their fitness, our assessment examines if those reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the population's abundance, reproduction, spatial structure and connectivity, growth rates, genetic health, or variance in these measures to make inferences about the population's extinction risks). In this step of our analyses, we use the population's base condition (established in the *Environmental Baseline* and *Status of Listed Resources* sections of this Opinion) as our point of reference.

Our assessment framework assumes—an assumption that is supported by published evidence—that the health and fitness of individual plants or animals will integrate the effects of the physical, chemical, and biological phenomena they are exposed to during their lifetimes and at specific developmental stages of their lives. That is, our assessment framework assumes that the total effects of exposing an animal to a suite of stressors, for example, coho salmon to a combination of toxic chemicals and an altered hydrograph from various flow controls will appear as a reduction in the fitness (reductions in annual or lifetime reproductive success) of individual coho salmon thus exposed. If exposing endangered or threatened marine and anadromous animals to chemical pollutants interacts with their exposure to other anthropogenic stressors, such as construction noise or disturbance or other toxic chemicals, and produces consequences that would not occur without that interaction, the consequence would appear as a reduction in performance of the individual animals.

Thus our assessment of the impact of the proposed action begins by considering the impact of the environmental baseline on the fitness of the individuals in the action area. As part of this assessment, we must consider how listed individuals are likely to respond to any interactions and synergisms between the proposed action and its stressors, pre-existing stressors and experience (represented by the *Status of the Species* and *Environmental Baseline*, as well as those stressors that are reasonably certain to occur in the action area for the future life of the action (represented by *Cumulative Effects*). If we conclude that listed individuals are likely to experience reductions in their annual or lifetime reproductive success, we then ask if those reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the population's abundance, reproduction, spatial structure and connectivity, genetic health, growth rates, or variance in these measures to make inferences about the population's probability of becoming extinct). Finally, if we conclude that the viability of one or more populations of a listed species is likely to be reduced, we determine whether that reduction is likely to be sufficient to reduce the viability of the species those populations comprise (here, a species' "viability" is its probability of becoming extinct or of being "recovered" to the point at which the protections of the ESA are no longer necessary or warranted). In this step of our analyses, we use the species' status as our point of reference.

For designated critical habitat, our destruction or adverse modification determinations must be based on an action's effects on the conservation value of habitat that has been designated as critical.[2] If an area encompassed in a critical habitat designation is likely to be exposed to the direct or indirect consequences of the proposed action on the natural environment, we ask if

[2] Several courts have ruled the definition of destruction or adverse modification that appears in the section 7 regulations at 50 CFR 402.02 as invalid. Consequently, we do not rely on that definition for the determinations we make in this Opinion. Instead, we use the conservation value of critical habitat for our determinations which focuses on the designated area's ability to contribute to the conservation of the species for which the area was designated.

constituent elements included in the designation (if there are any) or physical, chemical, or biotic phenomena that give the designated area value for the conservation of the species, are likely to respond to that exposure. If those constituent elements (or phenomena) are likely to respond, we ask if those responses are likely to be sufficient to reduce the quantity, quality, or availability of those constituent elements or physical, chemical, or biotic phenomena. If the conservation value is reduced, we then ask if those reductions are likely to be sufficient to reduce the conservation value of the entire critical habitat designation.

National Programmatic Consultations

Our national programmatic consultations typically analyze the general environmental consequences of a broad scope of actions or policy alternatives under consideration by a federal agency. In these types of consultations we focus on the general patterns associated with an agency's decision to authorize a particular national or programmatic action. Subsequent consultations that "tier" off of these national consultations, when warranted, would analyze the project and site specific effects typical of most consultations. Any subsequent section 7 consultations conducted by NMFS personnel would be designed to determine whether and to what degree the specific action under review fits within the general pattern identified in the national consultations, and would determine whether the specific action, is or is not likely to jeopardize the continued existence of endangered and threatened species or result in the destruction or adverse modification of designated critical habitat.

Thus, our national programmatic consultations focus on the evidence available to determine whether and to what degree the agency's action is likely to prevent exposure, or mitigate the responses or risks any responses would pose to listed species or their designated critical habitat. An agency can generally satisfy this requirement when the action contains features that: (1) prevent listed resources from being exposed to subsequent actions or their direct or indirect effects; (2) mitigate how listed resources respond to that exposure, when listed resources are exposed to the actions and their effect; or (3) mitigate the risks any responses pose to listed individuals, populations, species, or designated critical habitat when listed resources are likely to be exposed and respond to that exposure.

In examining an agency's program, we would examine the general activities the agency would authorize, fund or carry out. The steps of the national-level assessment remain much the same as described for our site-specific consultation, as outlined earlier in this section. National broad scale assessments and programmatic assessments, however, are necessarily focused on whether and to what degree a federal action can ensure that actions taken under the program are not likely to individually or cumulatively, jeopardize the continued existence of endangered or threatened species and are not likely to result in the destruction or adverse modification of designated critical habitat. Our description of the probable responses of the listed resources to the national action and the risks the national action poses to those listed resources is at the core of our evaluation, and is informed by the general patterns we observed through prior experience with an agency's actions or classes of activities.

The conceptual model NMFS uses for national consultations focuses on four main elements of action agency's national action: (1) the decision-making process an action agency uses to authorize, fund, or carry out national actions; (2) the national action, and any subsequent actions or activities the agency would authorize, fund or carry out in accordance with the national action; (3) the intended and unintended consequences that are likely to result from authorized activities; and (4) the mechanisms that improve the agency's action(s) over time. We begin our national consultations by recognizing that an agency's program normally represents the agency's decision to authorize fund, or carry out a suite or class of activities (or recommend actions) that may (or may not) require specific actions undergo subsequent review and decision-making.

An agency's decision-making process will normally identify certain standards that an action must satisfy before an agency would authorize, fund or carry them out. Generally, decision-making involves hard or formal procedures (such as agency regulatory procedures and public noticing requirements), soft or flexible information standards (e.g., agency "guidelines", and the best professional judgments personnel make when considering conflicting information and making recommendation in the face of uncertainty). These procedures outline how the agency would decide whether or not to authorize, fund or carry out specific actions. Typically an agency's decision making process is shaped to respond to:

- the statutory and regulatory standards an action must satisfy before the agency would authorize, fund, or carry them out;
- any data and other information the agency must gather and evaluate to satisfy their statutory and regulatory requirements, as well as requirements of the Administrative Procedure Act, Information Quality and related administrative statutes, like the Paperwork Reduction Act, Regulatory Flexibility Act, and so on.
- the agency's obligation to review and analyze the relevant information within the context of applicable standards to ensure that specific actions satisfy all applicable statutory and regulatory requirements;
- the results of the agency's efforts to monitor specific actions the agency has authorized, funded or carried out, and the consequences of those decisions;
- and any feedback mechanism an agency has created to ensure that a program satisfies its statutory mandates, regulatory requirements, and applicable goals, and minimizes unintended consequences from the agency action.

If an agency proposes to satisfy its section 7(a)(2) obligations using a decision-making process that insures that listed resources are not exposed to specific actions without undergoing a tiered section 7 consultation on a specific action, we examine the evidence available to determine whether and to what degree the agency's decision-making process is likely to produce that outcome. If the agency's decision-making process is designed to mitigate the consequences of exposing listed resources to specific actions, we examine the evidence available to determine whether and to what degree the agency's decision-making process produces that outcome. When we consult on a pre-existing program, the program's general pattern of performance over its history becomes our primary evidence.

After we examine an agency's decision-making process, we then examine the classes of actions the program would authorize, fund, or carry out. This step of our assessment is designed to determine whether and to what degree listed resources are likely to be exposed to different classes of activities that would be authorized, funded, or carried out under a program. During this step of our assessment, we consider the geographic distribution, timing, and constraints of the different classes of activities that would be authorized, funded, or carried out by a program (the geographic distribution of the activities' effects defines the action area of programmatic consultations). These analyses represent the "exposure analyses" of our programmatic consultations in which we try to identify the populations or subpopulations, ages (or life stages), and gender of the individuals that are likely to be exposed to an action's effects.

Then we use the best scientific and commercial data available to identify the classes of intended and unintended consequences that are likely to result from the different classes of activities. These analyses identify the probable direct and indirect consequences of exposing listed resources to those classes of activities for listed individuals, populations, and species, and designated critical habitat; these analyses represent the "response analyses" and "risk analyses" of our programmatic consultations. Our "response analyses" review the scientific and commercial data available to determine whether, how, and to what degree listed resources are likely to respond given their exposure to the intended and unintended consequences of classes of activities. Our "risk analyses" begin by identifying the probable consequences of those responses for the "performance" of listed individuals, and then they identify the consequences of changes in individual performance on the viability of the populations those individual represent. Our "risk analyses" conclude by determining the consequences of changing the viability of the populations, and the species those populations comprise. As stated earlier, our assessment is based on the general patterns that we observe through our prior experiences with a program or class of activities.

Evidence Available for the Consultation

To conduct our analyses, we considered lines of evidence available through published and unpublished sources that represent evidence of adverse consequence or the absence of such consequences. In particular, we considered information contained in *EPA's Biological Evaluation for Cyanide*, and published information used in deriving the 304(a) aquatic life criteria for cyanide. We supplemented this information by conducting electronic searches of literature published in English or with English abstracts using research platforms in the *Online Computer Library Center's* (OCLC) *First Search*, *CSA Illumina*, *Toxline*, *Science Direct*, *Water Resources Abstracts*, *Oceanic Abstracts*, *BioOne Abstracts and Indexes*, *Conference Papers Index*, *Lexis-Nexis*, *Google Scholar*, and *ISI Web of Science*. These platforms allowed us to cross search multiple databases for journals, open access resources, books, proceedings, web sites, doctoral dissertations and master's theses for literature on the biological, ecological, and medical sciences. Particular databases we searched for this consultation included *Basic Biosis*, *Dissertations*, *ArticleFirst*, *Proceedings*, *Aquatic Sciences and Fisheries Abstracts* and ECO databases, which index the major journals dealing with ecological risk, biology and ecology of particular species, and the toxicology of cyanide in freshwater, estuarine, and marine ecosystems (e.g., journals such as *Environmental Toxicology and Chemistry*, *Human and Ecological Risk Assessment*, *Journal of Mammalogy*, *Canadian Journal of Zoology*, *Transactions of the American Fisheries Society*, *Conservation Biology*, and others).

For our literature searches, we used paired combinations of the keywords cyanide, salmon, marine mammals, sea turtles, sturgeon, coral, sawfish, seagrass, and many others to search these electronic databases. Electronic searches have important limitations, however. First, often they only contain articles from a limited time span (e.g., First Search only provides access to master's theses and doctoral dissertations completed since 1980 and Aquatic Sciences and Fisheries Abstracts only provide access to articles published since 1964). Second, electronic databases commonly do not include articles published in small or obscure journals or magazines. Third electronic databases do not include unpublished reports from government agencies, consulting firms, and non-governmental organizations. To overcome these limitations, we supplemented our electronic searches by searching the literature cited sections and bibliographies of references we retrieved to identify additional papers that had not been captured in our electronic searches. We acquired references that, based on a reading of their titles and abstracts, appeared to comply with our keywords. If a references' title did not allow us to eliminate it as irrelevant to this inquiry, we acquired the reference.

Additionally, we separately searched the websites of the U.S. Geological Survey, EPA, states, U.S. Department of Health and Human Services, and the International Union for the Conservation of Nature (IUCN) for documents and data that identified potential effects of cyanide on marine, estuarine, and freshwater ecosystems and the individuals that inhabit these ecosystems. We conducted searches of EPA's Toxics Release Inventory (TRI) and Storage and Retrieval (STORET) databases for water quality data to identify areas where discharges are monitored for cyanide, and to characterize the general patterns of known occurrence and reported values over time and space.

From these documents we extracted the following: when the information for the study or report was collected, the study design, which species the study gathered information on, the sample size, the form of cyanide associated with the study, whether the study was conducted in a controlled laboratory environment or *in situ* (in the field or natural environment), whether other stressors were associated with study, study objectives, and study results. There is some concern that the exposure concentration and response observed in some studies on cyanide may not be accurate or reliable given differences between the analytical methods used, and forms of cyanide studied. Therefore, we followed a similar classification scheme as developed by Gensemer et al. (2007) to make comparisons among the type of cyanide exposure measurements performed in the studies. We classified studies according to whether they measured: (1) free cyanide using a reliable test method (e.g., ASTM 4282-95); (2) measured free cyanide but the test method accuracy is unknown; (3) measured weak acid dissociable cyanide; (4) measured total cyanide, and provided an estimate of free cyanide; (5) measured total cyanide, but did not estimate free cyanide; (6) did not provide an analytical verification of the cyanide concentration. Within each class of studies, we ranked each of the studies based on the quality of their study design, sample sizes, level of scrutiny before and during publication, and study results. We ranked carefully designed experiments (for example, experiments that control potentially confounding variables) higher than experiments that were not designed to control potentially confounding variables. We ranked carefully designed experiments higher than computer simulations, and we ranked studies on the response of listed species higher than studies on other, non-listed species. We also ranked studies that produced large sample sizes with small variances higher than studies with small sample sizes or large variances. Articles that did not rely on evidence produced by controlled experiment, uncontrolled field experiments, opportunistic observations of animal behavior or computer simulation received the lowest rating, but we considered the arguments and conclusions within these articles within our analyses.

Application of this Approach in this Consultation

The EPA proposes to continue approving state and tribal water quality standards for cyanide, which are based on their recommended 304(a) aquatic life criteria that were developed and published in the 1980s under the authority of the CWA. Section 304(a) of the CWA, the goals and purposes of the CWA, the implementing regulations for water quality standards (40 CFR 130-131), and the *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (later referred to as “the *Guidelines*”; Stephan et al. 1985) form the foundation, or the standards that the cyanide criteria were designed to meet. This Opinion represents NMFS’ evaluation of whether EPA’s approval of state or tribal water quality standards, or federal water quality standards promulgated by EPA, that are identical to or more stringent than the section 304(a) aquatic life criteria for cyanide satisfies EPA’s obligations pursuant to section 7(a)(2) of the ESA of 1973, as amended.

NMFS’ evaluation proceeds by asking if the approval of cyanide consistent with (or more stringent than) the 304(a) aquatic life criteria for cyanide proposed by EPA is likely to prevent the exposure of endangered species, threatened species, and designated critical habitat from aqueous cyanide concentrations that are toxic, given the approach it uses to approve water quality standards? If listed resources are not likely to be exposed to the direct and indirect effects of cyanide from activities the water quality standards would authorize, both individually and cumulatively, given the approach EPA uses to approve a water quality standards, we would conclude that EPA’s proposal to continue recommending the 304(a) aquatic life criteria for cyanide is not likely to jeopardize the continued existence of endangered species, threatened species, or result in the destruction or adverse modification of designated critical habitat under NMFS’ jurisdiction. If, however, listed resource are likely to be exposed to the direct and indirect effects of cyanide from activities the water quality standards would authorize, both individually and cumulatively, we would ask whether and to what degree listed species are likely to respond to their exposure, given the approach EPA uses to approve a water quality standards. As part of this analysis, we would examine whether and to what degree EPA has identified chemical, physical and biological scenarios that influence cyanide toxicity and presence in the environment inhabited by listed species and their critical habitat, the nature of any in situ effects, and the consequences of those effects for listed resources under NMFS’ jurisdiction, to determine if EPA can insure that the approval of state and tribal water quality standards that they are proposing is not likely to jeopardize the continued existence of endangered species or threatened species, or result in the destruction or adverse modification of critical habitat that has been designated for these species.

Understanding the Water Quality Program

EPA has asked that the Services consult on their approval of water quality standards where states and tribes adopt the standards that are consistent with or more stringent than the nationally recommended 304(a) aquatic life criteria. Since our analysis must consider the direct and indirect effects of the action together with the direct and indirect effects of any interdependent and interrelated actions[3], a critical first step to any consultation is determining whether and to what extent there are actions interrelated and interdependent with the action under consultation.

While EPA's BE does not examine interrelated and interdependent actions, it did provide us partial insight into the issue of what EPA considers interrelated and interdependent actions, inasmuch as EPA highlighted the general protective measures that states may adopt as part of their water quality programs as further evidence that listed resources would rarely, if ever, be exposed to cyanide at the recommended criteria values. Since the action as EPA has described it in its BE and subsequent documents, is the approval of water quality standards that states and tribes implement as enforceable standards for cyanide then it follows that the direct and indirect effects of any actions that are interrelated or interdependent with that approval must be considered in this consultation.

We developed a simple conceptual model to illustrate our understanding of the overall water quality program, and to assist us in determining whether there are actions that are interrelated or interdependent to the EPA's recommended 304(a) aquatic life criteria and subsequent approval of cyanide standards when states and tribes adopt the recommended numeric values. In part, we were interested in exploring the relationships among program components and EPA's decision to approve a particular standard and, specifically, whether the protective measures described in the BE and imposed by states and tribes should be considered in this consultation as interrelated and interdendent with the action to approve.

Our model depicts the relationship between EPA's 304(a) aquatic life criteria and other components of EPA's water quality-based approach to pollution control (Figure 1). Figure 1 also illustrates those relationships between "any action authorized, funded or carried out by" EPA under the composite program and section 7(a)(2) of the ESA. The model is based on the discussion of the water quality-based approach to pollution control, and the interrelated parts of executing the CWA as it was described by EPA in the *Water Quality Standards Handbook* (EPA 1994), information on the program characteristics that were provided by EPA in the cyanide BE, and is also based on our prior experiences with state water quality standards and NPDES permits issued by states and EPA. Our model, as with any descriptive model, represents a simplified map of the characteristics of the larger water-quality based pollution control program.

The goals and policies of the CWA establish the foundation for EPA's pollution control program. Pollution control begins, in part, with the identification of a target or priority pollutant and EPA's decision to "develop and publish" … (and from time to time thereafter revise) 304(a) criteria for water quality for that particular pollutant. EPA derives 304(a) aquatic life criteria through an established decision-making process outlined by the *Guidelines* (Stephan et al. 1985), which we depict at the top of Figure 1. Upon deriving a numeric value for a pollutant, EPA recommends (publishes) the numeric value for adoption and implementation. Publication typically involves a draft stage and a final stage in between which EPA solicits public comments.

The national aquatic life criteria provide the foundation for a wide variety of programs aimed at

[3] Interdependent actions are those actions that have no independent utility apart from the action under consideration. Interrelated actions are those actions that are part of a larger action and depend upon the larger action for their justification (50 CFR 402.02).

addressing pollution control under the CWA. EPA's 304(a) aquatic life criteria serve as guidelines or recommendations to states and tribes for defining water column concentrations of cyanide that EPA expects would protect against adverse ecological effects to aquatic life in fresh and salt water. The 304(a) aquatic life criteria recommendations are calculated to protect aquatic organisms from unacceptable toxicity during acute (short) and chronic (long) exposures in the water column. The intent is to define a level in the waterbody of a pollutant that will be fully protective of the designated uses of a water body and that a state or tribe may use in adopting its regulatory water quality standards and achieve the goals of their waterbodies (BE page 11, 40 CFR 131.2). States and tribes may use the 304(a) aquatic life criteria as a basis for developing enforceable water quality standards. The CWA requires all states to adopt water quality standards to restore and maintain the physical, chemical, and biological integrity of the Nation's waters. The CWA allows that states with an approved water quality program may adopt the 304(a) criteria as an enforceable standard (in combination with other relevant program elements), or they may modify the recommended criteria to reflect site-specific conditions, or create unique water quality standards (40 CFR 131.11(b)).

The focus of our national consultation with EPA, are those instances where a state or tribe "adopts" a water quality standard that is consistent with the recommended aquatic life criteria. In Figure 1, the consultation on this national approval is depicted by the yellow box, "National Section 7 Consultation".

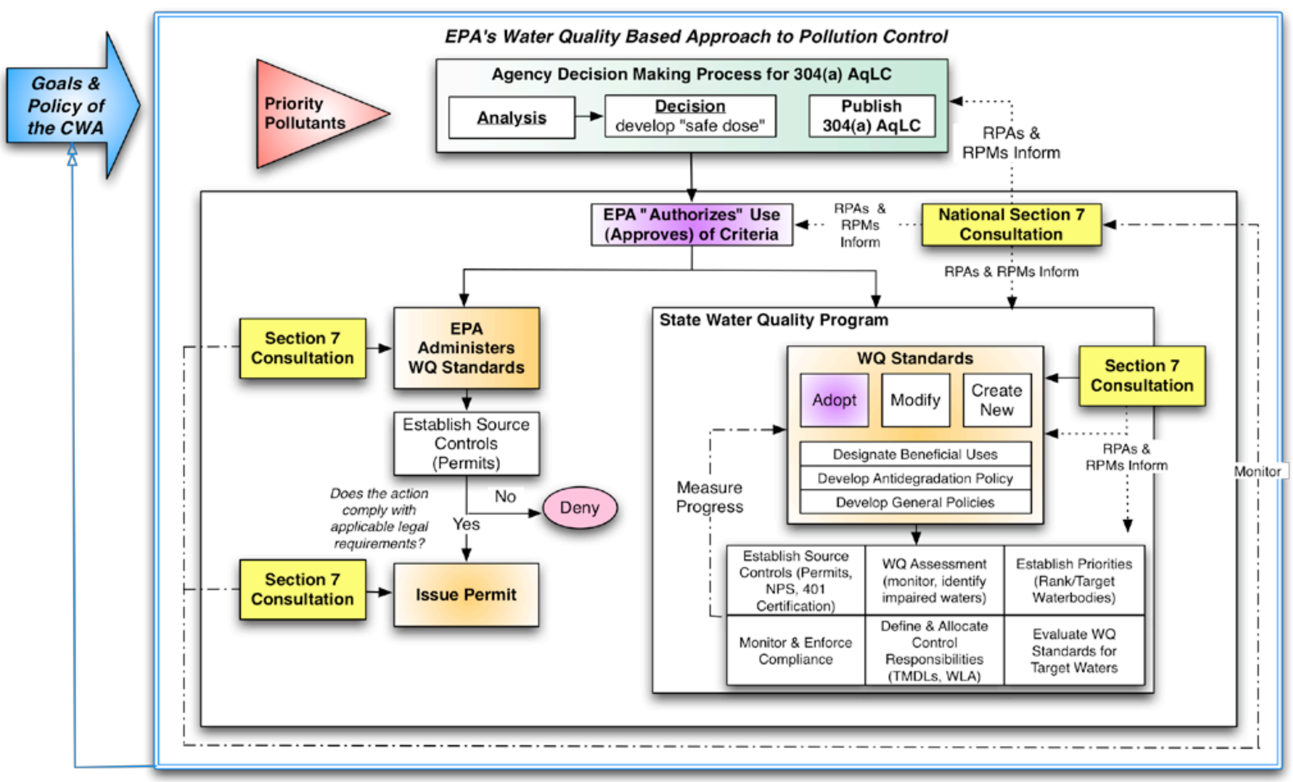

Figure 1. EPA's 304(a) aquatic life criteria and its relationship to the water quality-based pollution control program and section 7.

An approved standard, however, is more than just a numeric value for pollutants. Rather "a water quality standard defines the water quality goals of a water body, or portion thereof, by designating the use or uses to be made of the water and by setting the criteria necessary to protect the uses. States adopt water quality standards to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water Act….. Such standards serve the dual purposes of establishing the water quality goals for a specific water body and serve as the regulatory basis for the establishment of water-quality based treatment controls and strategies….. (40 CFR 131.2)." A state's water quality program contains eight general parts with specific regulatory requirements and guidance. We included the eight general parts of a state's water quality program on the right side of Figure 1. The eight parts are described by EPA (1994) as follows:

Establish protection levels. EPA's approach to pollution control begins with the identification of problem water bodies, and the water quality standards establish the assessment goals, and the water body uses intended for protection. Standards are not simply a numeric pollutant threshold level, but standards consist of three main elements (1) designated beneficial uses of a waterbody or segment of a waterbody (e.g., protection of aquatic life, recreation), (2) water quality criteria necessary to protect the use or uses of that particular waterbody (expressed in either numeric or narrative form[4]), and (3) an antidegradation policy. Additionally, states, at their discretion, may adopt general policies in their standards affecting the application and implementation of standards (e.g., mixing zone policies, variance policies, critical flow policies for permit based-limits).

Water quality assessments. Once water quality standards are adopted, states conduct water quality monitoring to identify those waters that are "water quality limited" or not meeting standards. Monitoring is important to evaluating whether designated uses are attained, determining whether Total Maximum Daily Limits (TMDL) are needed, and assessing compliance with permits and so on. Under section 305(b) of the CWA states are required to prepare a water quality inventory every two years to document the status of assessed water bodies. At this point the state may make a determination that the water body is not impaired but that the condition is due to natural conditions.

Establish priority waterbodies. When waters are identified that don't meet standards or are water quality limited, a state is expected to prioritize (rank) waterbodies for TMDL development.

Evaluate water quality standards for target waters. At this point in the water quality management process, States have targeted priority water quality-limited water bodies. EPA recommends that States re-evaluate the appropriateness of the water quality standards for the targeted waters if: 1) States have not conducted in-depth analyses of appropriate uses and criteria; 2) changes in the uses of the water body may require changes in the standard; 3) more recent water quality monitoring show the standard is being met; and, 4) site-specific criteria may be appropriate because of specific local environmental conditions or the presence of species more or less sensitive than those included in the national criteria data set.

[4] States must adopt numeric standards for toxic pollutants listed pursuant to section 307(a)(1) of the CWA and for which criteria have been published under 304(a).

1. *Define and allocate control responsibilities.* For water quality limited waters, States must establish a total maximum daily load (TMDL) that quantifies pollutant sources, and a margin of safety, and allocates allowable loads to the contributing point and non-point source discharges so that the water quality standards are attained. EPA recommends States develop TMDLs on a watershed basis.

2. *Establish source controls.* Source loads of pollutants are controlled through the TMDL, waste load allocations (WLA), best management practices (BMPs), and through the technology-based and water quality-based controls implemented through the NPDES permitting process. Although, many states and territories have authority to implement at least a portion of the NPDES program in their jurisdiction, EPA retains full or partial authority in many states and territories. In the case of nonpoint sources, both State and local laws may authorize the implementation of nonpoint source controls, such as best management practices (BMPs) or other management measures.

Monitor and enforce compliance. Monitoring is critical to the water quality-based decision making, and includes assessing compliance with TMDLs, permits, as well as in water loading (necessary to also capture nonpoint source pollution loads) and attainment of water quality goals. Point source dischargers are required to provide reports on compliance with NPDES permit limits. A monitoring requirement can be put into the permit as a special condition as long as the information is collected for the purposes of writing a permit limit. Effective monitoring programs are also required for evaluating nonpoint source control measures and EPA provides guidance in implementing and evaluating nonpoint source control measures. EPA and States are authorized to bring civil or criminal action against facilities that violate their NPDES permits. State nonpoint source programs are enforced under State law and to the extent provided by State law.

Measure progress. Arguably, one of the most important elements of the overall program are the efforts by the states (and EPA) to assess the effectiveness of the controls and standards, to determining water quality standards need to be revised, or more stringent controls are necessary (e.g., through permits or WLA and TMDLs). This is particularly important in determining whether a water body on the 303(d) list of impaired waters achieves water quality standards and can be removed from the state's 303(d) list, or to determine if WLA must be modified. This element is depicted as a feedback arrow between the general program elements and the foundation of state programs, the numeric standards and the policies that govern the program execution.

The left side of Figure 1 depicts those aspects of the water quality-based approach to pollution control that are approved and carried out directly by EPA. Criteria developed, published and approved by EPA are the foundation for many actions administered by EPA, including the promulgation of national water quality standards, and the issuance of NPDES permits.

Figure 1 also illustrates a general need by EPA to consult on actions that EPA "approves, funds, and carries out" under the program, which includes nationally approved criteria, as well as the approval of new state standards and the triennial review of those standards, and EPA's issuance of NPDES permits. The scope and details of such consultations depend upon EPA's discretionary control or authority to insure that its decisions on these actions comply with the CWA, its implementing regulations and policies. The yellow boxes in Figure 1 generally depict those areas where EPA would consult with the Services on their actions.

The consultation boxes in Figure 1 are linked to the national consultation to illustrate that NMFS will use the evidence obtained in regional and site-specific consultations to determine whether a particular consultation produced the expected results or produced results that were not consistent with the assumptions and conditions of the national consultation. That is, this first national consultation establishes a feedback framework to assist NMFS in assessing (a) the reliability, validity, or relevance of any evidence it relied upon in its national consultation; (b) whether the national consultation produced the anticipated results or produced results that were not consistent with subsequent consultations, (c) assessing the current status of any reasonable and prudent alternatives, reasonable and prudent measures, terms and conditions, and reporting requirements that EPA must comply with under the national consultation; and (e) the current and projected trends of listed resources, and the altered environmental baseline. The arrows in connecting these consultations in Figure 1 are broken because this is a newly developed feedback framework and has not previously been implemented by NMFS in its water quality consultations with EPA.

Interrelated and Interdependent Actions

The effects of EPA's 304(a) criteria recommendation must be understood in the larger context of the CWA. This larger context is framed by Congress' stated objective, goals, and policies of the CWA, and the programs and activities authorized by the CWA and implemented by EPA, and states and tribes to achieve these objectives, goals and policies. It is the CWA requirement that all states adopt water quality standards to restore and maintain the physical, chemical, and biological integrity of the Nation's waters that places the standards at the core of the overall strategy for water-quality based pollution control. As described previously, standards serve as the regulatory basis for the water quality-based approach to pollution control and are used to identify water quality problems caused by various land uses, such as improperly treated wastewater discharges, runoff or discharges from active or abandoned mining sites, sediment, and so on.

As a practical matter most states and tribes adopt EPA's recommended 304(a) criteria for most pollutants as part of their water quality standards even though they can develop unique criteria for their waters (EPA 1999). According to a review of state water quality criteria for cyanide, we found that more than 80% of the states and territories adopted EPA's acute and chronic freshwater criteria for cyanide or criteria that were more stringent[5] (Appendix A). Eleven states (Arizona, Arkansas, California, Iowa, Louisiana, Nebraska, Ohio, Oklahoma, Texas, Washington, and Wisconsin) adopted higher values in their standards, some significantly so. Some of these states adopted different values for cold waters versus warm waters (e.g., Arizona) or specified particular areas subject to these different values (e.g., Washington, California). States that set significantly higher standards than EPA's nationally recommended 304(a) criteria included Iowa, Louisiana, Ohio and Texas. No states set lower salt water values than EPA

[5] We interpreted "more stringent" to be a lower value that would lead to less cyanide in the water. Most states and territories that had set lower standards for cyanide were only a few tenths to hundredths lower than the value recommended by EPA.

recommended, but a few established higher values. California established levels as high as 10.0 μg/L CN for the saltwater instantaneous maximum and Texas set their chronic and acute saltwater criteria at 5.6 μg/L CN. Local exemptions in some state waters are much higher than these broader state limits. For instance, Illinois allows for 100 μg/L for acute exposure and 1,000,000 μg/L in some waterways in Cook County (home to Chicago). Although several states adopted new standards that differ from EPA's recommended values, the fact that most states follow EPA recommendations for cyanide verbatim illustrates the influence that EPA's guidance has on state standards. We suspect that EPA's action to develop and publish (recommend) 304(a) aquatic life criteria likely is sufficient to dissuade many states from investing the resources to develop unique water quality standards, particularly in times of economic hardship and reduced state budgets.

That the CWA creates an independent statutory requirement that states adopt enforceable water quality standards is sufficient reasoning to support the argument that state standards have "independent utility" and would not generally be considered interdependent with EPA's 304(a) criteria. However, since the vast majority of states adopt the 304(a) criteria as developed and published by EPA, and EPA has requested that this consultation, programmatically, address their need to consult on their approval of the water quality standards that are consistent with, or more stringent than the 304(a) recommended criteria the argument of independent utility is moot. That is, it is EPA's expectation that this national consultation address their general action to approve any state or tribal water quality standards for cyanide that are consistent with, or more stringent than the numeric value they recommend, and by doing so EPA hopes to eliminate subsequent regional consultations on water quality standards. .

As we described earlier, the level of protection afforded to a water body under the CWA is defined by the sum of the designated uses, criteria, antidegradation policy[6], and general policies. While all are required in a state submission, the designated uses and criteria are particularly inseparable components of a water quality standard as evidenced by EPA's language on approving a submission. That is, to approve a proposed water quality standard EPA must find that a state has adopted uses that are consistent with the requirements of the CWA and that adopted criteria protect those designated uses. EPA cannot approve a numeric value for a particular pollutant, like cyanide, if that numeric value does not support the uses the state has designated for a particular water body. The designated uses are integral to the approval and have no independent utility apart from the approval of a water quality standard, but are one of the most important parts of a water quality standard. More so, a water quality standard, by definition, is not complete without a finding that a particular criterion meets the designated uses. Therefore, designated uses are also interrelated with a particular criterion value because they are integral parts of the standard (part of the larger action), and depend upon the larger action for their justification.

[6] In a January 27, 2005, memorandum to it Regional Offices, EPA concluded that ESA section 7 consultation does not apply to EPA's approvals of state antidegradation policies because EPA's approval action does not meet the "Applicability" standard defined in the regulations implementing section 7 of the ESA (EPA 2005; 50 CFR 402.03). Section 402.03 of the consultation regulations (50 CFR part 402) states that section 7 and the requirements of 50 CFR part 402 apply to all actions in which there is discretionary Federal involvement or control. EPA concluded that they are compelled to approve State antidegradation policies if State submissions meet all applicable requirements of the *Water Quality Standards Regulation* (40 CFR part 131) and lack discretion to implement measures that would benefit listed species. As a result, EPA determined that consultation is not warranted on antidegradation policies because the Agency does not possess the regulatory authority to require more than the minimum required elements of the regulations.

When we embarked on this evaluation, however, we noted we were particularly interested in determining whether the protective measures described in the BE and imposed by states and tribes should be considered in this evaluation. EPA stated that states and tribes may, in addition to adopting numeric criteria, adopt: narrative criteria, biological criteria, or site-specific criteria for cyanide. EPA also noted that during implementation of their water quality standards, several other assumptions are made when allocating pollutants, for permitting purposes, among point source discharges to protection of species. As part of the TMDL and NPDES permit development, according to EPA most states and tribes use the following protective assumptions in the development of their TMDLs and water quality based effluent limitations: (1) assume that all dischargers are discharging the contaminant at the maximum permitted levels, (2) provide for an unallocated "margin of safety" when developing TMDLs, (3) assume the maximum permitted discharge volume, (4) assume the maximum concentration of loading of pollutants, (5) assume no environmental degradation of pollutants, (6) assume all discharged pollutants remain biologically available, (7) assume receiving stream flows are very low, (8) assume that acute toxicity limits apply at the "end of the pipe", (9) assume that only a portion of the design flow is available for mixing for controls on chronic toxicity, (10) assume that aquatic species live continuously at the "edge of the mixing zone", (11) assume no internal dilution of process wastewater, (12) assume conservative values for upstream concentrations of pollutants, (13) permit conditions should not be relaxed in subsequent permit reissuance (antibacksliding), (14) antidegradation requirements protect existing uses, (15) assume low threshold for "reasonable potential" if few data are available. While we cannot disagree that these components of a state's water quality program warrant further examination, and may even qualify as interrelated and interdependent actions that demonstrate the success (or failures) of various specific programs and the success of the overall water quality program, the BE provided no evidence of the general patterns of the implementation of these measures, nor an evaluation of the success or failures of these protective mechanisms across the national landscape. We further acknowledge that each TMDL, WLA, and NPDES could in fact be considered actions interdependent to EPA's approval of a state's water quality standards because the standards and goals for a water body "serve as the regulatory basis for the establishment of water-quality based treatment controls and strategies (40 CFR 131.2)."

Perhaps the most compelling reason that the above mentioned actions and other general program operations have independent utility, however, is the fact Congress intentionally divided many of these state and tribal actions into different sections of the CWA. In fact much of the statute directs the actions of state and tribes, not EPA's, supporting state autonomy for the protection of their waters. That the sections were designed to work together to achieve the goals set forth by Congress should not be a surprise, and in of itself should not be reason to consider all programs that rely on the water quality standards as interrelated or interdependent to the approval of water quality standards. Thus we default to the statutory construct, and the distinctive sections of the Act that instruct states and tribes on the execution and operation of their overall water quality program, as providing the strongest argument for independent utility.

Moveover, we note that the inclusion into this consultation of the myriad of such actions as dictated by the different programs that rely on water quality standards would easily make this national consultation untenable in short order. Thus, unless we can establish evidence of the general pattern in which the protective measures EPA noted in their BE are implemented across states and tribes (information which was not contained within the BE) then these assertions served little relevance to our analysis on the national scale. We further submit that individual NPDES permits, TMDLs, WLA, and other management aspects of a state's water quality program, while emanating from EPA's approved water quality standards, merit evaluation in subsequent consultations, where appropriate. Where EPA does not retain discretion, and such actions may affect listed resources, then states and tribes ought to seek a permit from the Services pursuant to section 10(a)(1)(b) of the ESA. We therefore propose that while each of the actions that are part of the overall approach to protecting aquatic life in waters of the United States are targeted to assessing compliance with standards and instituting change to achieve compliance through modification to allowable discharges or to the standards themselves, they have independent and significant roles in achieving the goals of the CWA. Consequently, they merit separate reviews as appropriate under the ESA. Such separate reviews can be linked through our conceptual model feedback links (see Figure 1), to assist NMFS and EPA in conducting holistic review of the effectiveness of the programs for protecting listed resources.

What we cannot separate on the basis of independent utility, however, as they are intimately a part of the action as EPA has proposed it, are the elements of a state or tribal water standard that must be included in each submittal to EPA for review and in order for EPA to approve said standard (see EPA 1994). As established in the foregoing discussion, these include: designated uses, criteria, antidegradation policy, and general policies. Hence, we address these other components as they are an essential part of any standard EPA approves, as interrelated and interdependent actions to EPA's approval of a numeric pollutant value in a state or tribal standard. These interrelated and interdependent actions are discussed in the *Effects of the Action* section of this Opinion.

Water Quality Standards

Water quality standards, as mentioned previously, are the mechanism by which protection levels for a water body are established. The water quality standards establish the assessment goals (e.g., numeric or narrative criteria), and the water body uses intended for protection. Whenever a state revises or adopts a new water quality standard such revised or new standard must be submitted to EPA for review. The water quality standard must include designated uses consistent with the provisions of section 101(a)(20 and 303(c)(2) of the CWA, the methods used and analyses conducted to support water quality standards revisions, water quality criteria sufficient to protect the designated uses, an antidegradation policy, certification that the water quality standards were duly adopted pursuant to state law, and general information that will aid the EPA in determining the adequacy of the scientific basis of the standards (40 CFR 131.6).

According to the CWA, the standards shall protect the public health or welfare, enhance the quality of water and serve the purposes of the Act, and shall be established taking into consideration their use and value for public water supplies, propagation of fish and wildlife, recreational purposes, and agricultural, industrial, and other purposes, and also taking into consideration their use and value for navigation. The phrase to "serve the purposes of the Act" as defined in 303(c) of the CWA, means that the water quality standards should meet the objectives of the Act "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." In order to achieve this objective Congress declared that---

(1) It is the national goal that the discharge of pollutants into the navigable waters be eliminated by 1985;

(2) It is the national goal that where ever attainable, an interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water be achieved by July 1, 1983;

(3) It is the national policy that the discharge of toxic pollutants in toxics amounts be prohibited….”

These three goals, which are commonly referred to as the “zero discharge” goal, “the fishable/swimmable” goal, and the “no toxics in toxic amounts” goal, are accompanied in the statute by a number of subsidiary goals and policies (Adler et al. 1993). Water quality standards for aquatic life are primarily designed to meet the fishable/swimmable goal of the CWA.

Water quality standards (in particular, the numeric criteria coupled with a water body’s designated uses) are the core mechanism for meeting the goal of the CWA, and “getting water quality standards right starts with getting designated uses right (EPA 2008a).” When a state submits a water quality standard, EPA must review and approve (or disapprove) a standard based upon whether a state has: (a) adopted uses that are consistent with the requirements of the CWA, (b) adopted criteria that protect the designated uses, (c) followed legal procedures for adopting standards, (d) whether the submission meets the regulatory requirements (40 CFR 131.5). In specifying appropriate water uses, each state must take into consideration the protection and propagation of fish, shellfish and wildlife, and recreation in and on the water (the “fishable/swimmable” goal among other things; 40 CFR 131.10(a)), whether or not a use is currently being attained.

Designated Uses

Designated uses are statements of management objectives and expectations for water bodies under state or tribal jurisdiction. As defined in 40 CFR 131.3, designated uses are specified in the water quality standards for each water body or water body segment regardless of whether or not they are being attained. Designated uses include, but are not limited to: water supply (domestic, industrial and agricultural); stock watering; fish and shellfish uses (salmonid migration, rearing, spawning, and harvesting; other fish migration, rearing, spawning, and harvesting); wildlife habitat; ceremonial and religious water use; recreation (primary contact recreation; sport fishing; boating and aesthetic enjoyment); and commerce and navigation.

The water quality standards regulation requires that states and tribes specify which water uses are to be achieved and protected. These uses are determined by considering the value and suitability of water bodies based on their physical, chemical, and biological characteristics as well as their geographical settings, aesthetic qualities and economic attributes. Each water body does not necessarily require a unique set of uses. Rather, water bodies sharing characteristics necessary to support a use can be grouped together. If water quality standards specify designated uses of a lower standard than those that are actually being attained, the State or Tribe is required to revise its standards to reflect these uses.

Antidegradation

Antidegradation implementation procedures identify the steps and questions that must be addressed when proposed activities may affect water quality. The water quality standards regulation requires that states and tribes establish a three-tiered antidegradation program. The specific steps to be followed depend upon which tier or tiers apply. These tiers are listed below:

- Tier 1: These requirements are applicable to all surface waters. They protect existing uses and water quality conditions necessary to support such uses. These uses can be established if they can be demonstrated to have actually occurred since November 28, 1975, or if water quality can be demonstrated to be suitable for such uses. If an existing use is established, it must be protected even if it is not listed in the water quality standards as a designated use.

- Tier 2: These requirements maintain and protect "high quality" water bodies where existing conditions are better than those necessary to support CWA § 101(a)(2) "fishable/swimmable" uses. Although the water quality in these water bodies can be lowered, states and tribes must identify procedures that must be followed and questions that must be answered before a reduction in water quality can be allowed. The water quality of these water bodies cannot be lowered to a level that would interfere with existing or designated uses.

- Tier 3: These requirements maintain and protect water quality in outstanding national resource waters (ONRWs) and generally include the highest quality waters of the United States. ONRW classification also offers special protection for waters of exceptional ecological significance. Except for certain temporary changes, water quality cannot be lowered in these waters. states and tribes decide which water bodies qualify as ONRWs.

In a January 27, 2005, memorandum to it Regional Offices, EPA concluded that ESA section 7 consultation does not apply to EPA's approvals of state antidegradation policies because EPA's approval action does not meet the "Applicability" standard defined in the regulations implementing section 7 of the ESA (EPA 2005; 50 CFR 402.03). Section 402.03 of the consultation regulations (50 CFR Part 402) states that section 7 and the requirements of 50 CFR part 402 apply to all actions in which there is discretionary Federal involvement or control.

EPA concluded that they are compelled to approve State antidegradation policies if State submissions meet all applicable requirements of the *Water Quality Standards Regulation* (40 CFR part 131) and lack discretion to implement measures that would benefit listed species. As a result, EPA determined that consultation is not warranted on antidegradation policies because the Agency does not possess the regulatory authority to require more than the minimum required elements of the regulations. For these reasons, antidegradation will not be a part of this consultation.

General Policies

States and tribes may adopt general policies and provisions regarding the implementation of water quality standards. These policies and provisions are subject to EPA review and approval. General policies must relate to designated use criteria or antidegradation. These policies and provisions include:

1. Mixing Zones: A mixing zone is a defined area surrounding or downstream from a point source discharge where the effluent is diluted by the receiving water and criteria otherwise applicable to the water body may be exceeded. At their discretion, states and tribes may allow mixing zones for point source discharges. Mixing zone procedures describe the methodology for determining the location, size, shape, and quality of mixing zones.

2. Variances: Variances temporarily relax a water quality standard. They are subject to public review every three years and may be extended. A variance may specify interim water quality criteria applicable for the duration of the variance. States or tribes may wish to include a variance as part of a water quality standard as an alternative to removing a designated use. Variances are intended to help assure that further progress toward improving water quality is achieved.

3. Low Flows: State and tribal water quality standards may identify policies and procedures to determine critical low flow conditions. For example, such procedures are applied when calculating discharge requirements to be included in National Pollutant Discharge Elimination System (NPDES) permits.

Evaluating Exposure at the National Level

The next step in our analysis involved evaluating the contaminant, cyanide (the stressor), in the environment in which the listed resources occur. Although we searched, we simply could not find sufficient data to conduct a quantitative assessment of the likelihood of exposure, or the likelihood of exposure at a particular numeric value. Therefore, our analysis focuses largely on the consequences of an exposure at criterion value. However, to examine a species' (and their critical habitat's) risk of exposure, we searched for evidence that would help us describe the (1) the transport, fate, and persistence of cyanide in the environment, (2) the distribution of uses and occurrences of cyanide across the U.S., and (3) temporal and spatial changes, where we could find evidence of these changes, across the U.S.

We began by constructing a simple conceptual model for evaluating the effects of contaminants on listed species and critical habitat. This model depicts the release of a contaminant into the environment, its transport through the environment and its contact with the listed species (Figure 2). The fate of pollutant, and whether it reaches habitats containing listed species, depends upon a wide number of variables including chemical form and structure, volume dispersed and the manner in which it is dispersed, distance of travel, and processes of sorption, degradation, and dilution, to name a few.

In describing the basic properties of cyanide, we also looked at chemical, biological and physical attributes in the environment that might act as “filters” or “magnifiers” that influence the relationship between cyanide and the induction of effects on listed species. For instance, Cloern (2001) used tidal energy to illustrate the importance of filters in eliciting certain responses within an ecosystem—tidal energy influences turbulent kinetic energy and mixing in shallow waters, and ultimately the expression of eutrophication. Differences in tidal amplitude are one mechanism by which different estuaries will respond dissimilarly to equally high loads of nutrients, and in turn the filters acting within different ecosystems would dictate potentially very different pollutant concentrations to which listed species would be exposed.

Some of the particular features of an ecosystem or site that can act as filters, influencing the nature, magnitude, and spatial and temporal distribution of pollutants to which listed species may be exposed include: water hardness, pH, precipitation, wind, light, bathymetry, stratigraphy, topography, trace gas absorption, mineral weathering, elemental storage ability, soil chemical processes, microbial transformation, and so on. For site-specific assessments, as much as possible, the site's features should be described and used to evaluate associations between the listed species and their critical habitat, and the particular pollutant under evaluation. At the national scale, however, we look for evidence of the types of filters that generally would be expected to interact with cyanide along its general transport pathway, and that would influence its availability, toxicity and severity.

Our simple transport model, illustrated in Figure 2, serves as a map for our analysis. That is, it illustrates the main pathways— the physical course cyanide generally takes from the source to the receptor organism or communities of interest (Suter et al. 2002). For section 7 evaluations of pollutants, the receptor organism is the listed species or designated critical habitat. An exposure pathway is complete when the chemical(s) under evaluation reach the receptor organism. A pathway is incomplete when the stressor does not reach the organism under evaluation. Simply, in the latter case when the pathway is incomplete, the chemical does not co-occur with the listed species or its designated critical habitat.

Our conceptual transport model emphasizes the exposure route through surface waters because the primary route of exposure to chemical contaminants for most of NOAA's trust resources will often be through water-borne exposures. As with any conceptual model, this visual depiction of exposure pathways is a simplified representation of what can be expected in the natural environment. For instance, not only would some species be exposed to surface water contaminants, animals that live portions of their life cycle out of water like many marine mammals (aquatic-dependent species) may be exposed to contaminants on land. Even wholly aquatic species, like salmon may be exposed to contaminants in terrestrial vegetation—through leaf litter and insects (allochthonous stream input)—and contaminated soils that enter the aquatic environment.

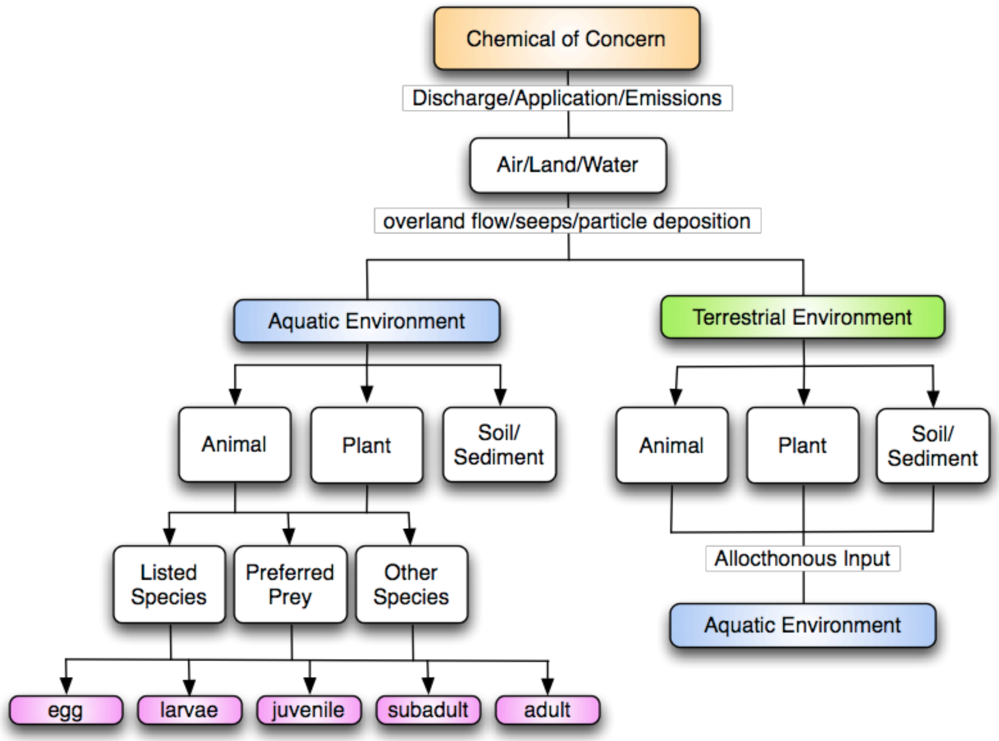

Figure 2. Simple transport model depicting pollutant pathways to aquatic habitats and aquatic species.

We would consider an exposure pathway complete when the chemical under evaluation would generally be expected to reach the listed species and incomplete when the stressor does not reach the listed species. Often the more difficult aspect of a section 7 evaluation is identifying the indirect pathways by which a listed species or their critical habitat is affected by a chemical stressor, which requires an examination of relationship of the listed species to the communities of which it is a part, and the environment in which it resides, depends upon, and is adapted. To capture indirect exposure pathways we look at the relationship of the listed species to the community and environment in which it lives. This means, that not only do we look for effects of cyanide directly acting on the listed species, we examine the effect that cyanide has on the biological community and environment in which the listed species lives (Figure 3). We do this to determine if cyanide would induce community and environmental changes that would likely affect the listed species, such as changes in prey availability or health.

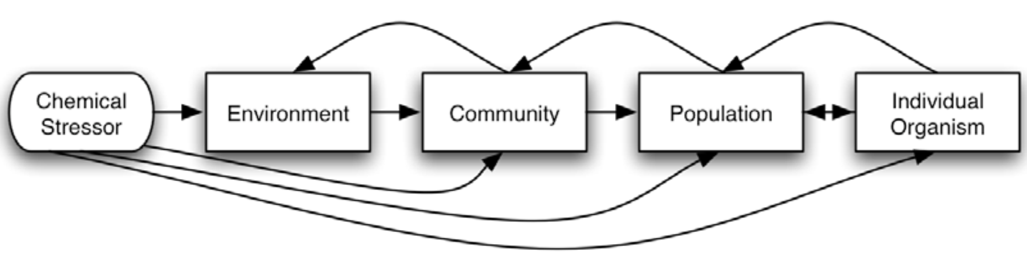

Figure 3. A chemical stressor and its potential relationships with organisms in the wild

Our challenge in this step is to identify: what populations, life history forms or stages are exposed to the proposed action; the number of individuals that are exposed; the pathways of their exposure; the timing and duration of their exposure; the frequency and magnitude of the concentrations of the exposure; and how exposure might vary depending upon the characteristic of the environment and individual behavior. Typically, in this step of our analysis we would identify how many individuals are likely to be exposed, which populations the individuals represent, where and when the exposure would occur, how long the exposure would occur, the frequency of the exposure, and any other particular details that help characterize the exposure. To do this we require knowledge of a species' population structure and distribution, migratory behaviors, life history strategy, and abundance.

All of the species under NMFS' jurisdiction are "aquatic" or "aquatic-dependent", meaning that at least one or more life stages are aquatic and could potentially be exposed to aqueous pollutants. Therefore, since EPA has asked that this consultation cover their national approval of standards that are consistent with their recommended aquatic life criteria, we began our assessment with the basic assumption that all of the listed species and critical habitat under NMFS' jurisdiction, as well as any species proposed for listing and critical habitat proposed for listing, would potentially be exposed to cyanide at the recommended criteria values at some time during their life cycle. NMFS assumes the recommended criteria value is an appropriate starting assumption for exposure in particular because the recommended value is assumed to represent a "safe dose" of cyanide.

Using this assumption, we asked whether and to what degree would animals that are exposed at the recommended level be protected if exposed at that value (this is part of our response analysis). Next, we asked whether and to what degree the proposed action and any interrelated and interdependent actions would mitigate, minimize or avoid allowing cyanide discharges to reach (or exceed) the recommended criteria. Because this examination is done at the national level, we looked for general patterns of cyanide where that information was available to us. We used such information as general patterns of the distribution of uses, manufacturing, and incidental occurrences of cyanide in the environment, and we looked for temporal and spatial changes in these uses to characterize the past 20 years of cyanide in the environment, and as a basis for predicting the future of cyanide in the environment across our action area. Our evaluation is explained in detail in our effects analysis.

Action Area

EPA has defined the action area for the cyanide consultation, and for the 304(a) aquatic life criteria consultations in general as all "waters of the United States" including "territorial seas" (see the BE, pages 8 and 9, and the *Methods Manual* page 6). The CWA (33 USC 1362) defines territorial seas as "the belt of the seas measured from the line of ordinary low water along that portion of the coast which is in direct contact with the open sea and the line marking the seaward limit of inland waters, and extending seaward a distance of three miles." This action area includes such waters within and surrounding Indian Country, the 50 States, and all United States territories. The terms "waters of the United States" is defined under 40 CFR Section 122.2 and reiterated in EPA's cyanide BE.

As early as 1789, the United States territorial sea was established at three nautical miles. On 27 December 1988, however, President Regan issued a proclamation that extended the United States territorial sea to 12 nautical miles from the baselines of the United States. Although, nothing in the proclamation extended or otherwise altered existing federal or State law subsequent to the 1988 proclamation, several federal laws adopted the terms of the Proclamation to define the United States territorial sea for purposes of that particular statute (e.g., the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990, the Antiterrorism and Effective Death Penalty Act of 1996). However, others, including the Federal Water Pollution Control Act (aka. the CWA) continue to use the three mile limit in its definition of the United States territorial sea.

The action area for the purposes of consultation, however, is not limited to the area of an agency's jurisdiction. Rather, in consultation the action area is defined as all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR 402.02). Many federal actions that NMFS consults on occur in the United States territorial sea, the contiguous zone, exclusive economic zone, and on the high seas. The issue of jurisdiction is relegated to the point in the Opinion at which NMFS prescribes management actions (Reasonable and Prudent Alternatives and Reasonable and Prudent Measures) for the purpose of exempting the taking of threatened and endangered species from the prohibitions of section 4(d) and 9 of the ESA. (See the section of this Opinion titled *Reasonable and Prudent Alternatives*). Consequently, the action area for EPA's 304(a) aquatic life criteria consultations includes the minimal area, as defined by the freshwater, estuarine and ocean water bodies of the United States and its territories (delineated by the CWA) and any areas the particular pollutant under consultation (in this case cyanide) is transported beyond these limits by such biotic and abiotic factors as river runoff, tidal energy, topography, stratigraphy, biota [trapping/assimilation), that may influence chemical transport processes beyond original areas of dispersion. We expect, based on the chemical processes (sources, transport, and fate) of cyanide, which are described later in this Opinion, that most of the action area for this consultation on cyanide is contained by the jurisdictional waters as outlined by the CWA. However, in certain localities we expect that conveyance systems may extend to the outer edge of this action area, or that the discharge plume may extend beyond three nautical miles. Unfortunately, we cannot identify the specific areas or conveyance systems where this may occur, and thus recognize that our action area is generally delineated according to three nautical miles extending from the United States coastline.

Since NMFS has jurisdiction over marine and anadromous threatened and endangered species, and their critical habitat, this Opinion addresses the potential effects of EPA's aquatic life criteria in a portion of the action area defined for 304(a) aquatic life criteria. Specifically, this Opinion focuses on the direct and indirect effects of the recommended criteria along the coastal regions of the United States and its territories, where listed resources under NMFS' jurisdiction occur. As such, although interior fresh waters (e.g., landlocked lakes and ponds of the midwest United States) constitute a portion of the action area for this consultation, listed resources under NMFS' jurisdiction do not occur in these areas and these portions of the action area are not considered further in this Opinion.

Status of the Species and Critical Habitat

In this section of this Opinion we describe the threatened and endangered species and their designated critical habitat that occur in the action area and may be exposed to EPA's approved aquatic life criteria for cyanide. All listed species within NMFS' jurisdiction are "aquatic" or "aquatic dependent" and may occur within portions of the action area for the recommended aquatic life criteria. NMFS has determined that the following species and critical habitat may occur within the action area for EPA's 304(a) aquatic life criteria for cyanide (Table 2).

Table 2. Species Listed as Threatened and Endangered and Proposed for listing, and their designated Critical Habitat (denoted by asterisk) in the Action Area. Double asterisks denote Proposed Critical Habitat.

Common Name (Distinct Population Segment or Evolutionarily Significant Unit)	Scientific Name	Status
Cetaceans		
Beluga whale** (Cook Inlet)	*Delphinapterus leucas*	Endangered
Blue whale	*Balaenoptera musculus*	Endangered
Bowhead whale	*Balaena mysticetus*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
Killer Whale (Southern Resident*)	*Orcinus orca*	Endangered
North Atlantic right whale*	*Eubalaena glacialis*	Endangered
North Pacific right whale*	*Eubalaena japonica*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Pinnipeds		
Hawaiian monk seal*	*Monachus schauinslandi*	Endangered
Steller sea lion (Eastern*)	*Eumetopias jubatus*	Threatened
Steller sea lion (Western*)		Endangered
Marine Turtles		
Green sea turtle (Florida & Mexico's Pacific coast colonies)*	*Chelonia mydas*	Endangered
Green sea turtle (All other areas)*		Threatened
Hawksbill sea turtle*	*Eretmochelys imbricate*	Endangered
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered
Leatherback sea turtle* (also **)	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle	*Caretta caretta*	Threatened
Olive ridley sea turtle (Mexico's Pacific coast breeding colonies)	*Lepidochelys olivacea*	Endangered
Olive ridley sea turtle (All other areas)		Threatened
Anadromous Fish		
Atlantic salmon*	*Salmo salar*	Endangered
Chinook salmon (California coastal*)	*Oncorhynchus tschawytscha*	Threatened
Chinook salmon (Central Valley spring-run*)		Threatened
Chinook salmon (Lower Columbia River*)		Threatened
Chinook salmon (Upper Columbia River spring-run*)		Endangered
Chinook salmon (Puget Sound*)		Threatened
Chinook salmon (Sacramento River winter-run*)		Endangered
Chinook salmon (Snake River fall-run*)		Threatened
Chinook salmon (Snake River spring/summer-run*)		Threatened

Chinook salmon (Upper Willamette River*)		Threatened
Chum salmon (Columbia River*)	*Oncorhynchus keta*	Threatened
Chum salmon (Hood Canal summer-run*)		Threatened
Coho salmon (Central California coast*)	*Oncorhynchus kisutch*	Endangered
Coho salmon (Lower Columbia River)		Threatened
Coho salmon (Southern Oregon & Northern California coast*)		Threatened
Coho salmon (Oregon coast*)		Threatened
Green sturgeon (Southern*)	*Acipenser medirostris*	Threatened
Gulf sturgeon*	*Acipenser oxyrinchus desotoi*	Threatened
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered
Smalltooth sawfish*	*Pristis pectinata*	Endangered
Sockeye salmon (Ozette Lake*)	*Oncorhynchus nerka*	Threatened
Sockeye salmon (Snake River*)		Endangered
Steelhead (Central California coast*)	*Oncorhynchus mykiss*	Threatened
Steelhead (California Central Valley*)		Threatened
Steelhead (Lower Columbia River*)		Threatened
Steelhead (Middle Columbia River*)		Threatened
Steelhead (Northern California*)		Threatened
Steelhead (Puget Sound)		Threatened
Steelhead (Snake River*)		Threatened
Steelhead (South-Central California Coast*)		Threatened
Steelhead (Southern California*)		Endangered
Steelhead (Upper Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened
Marine Invertebrates		
Black abalone	*Haliotis cracherodii*	Endangered
Elkhorn coral*	*Acropora palmata*	Threatened
Staghorn coral*	*Acropora cervicornis*	Threatened
White abalone	*Haliotis sorenseni*	Endangered
Marine Plants		
Johnson's seagrass*	*Halophilia johnsonii*	Threatened
Proposed for Listing		
Bocaccio	*Sebastes paucispinis*	Proposed Endangered
Canary rockfish	*Sebastes pinniger*	Proposed Threatened
Pacific eulachon/smelt	*Thaleichthys Pacificus*	Proposed Threatened
Spotted seal	*Phoca largha*	Proposed Threatened
Yelloweye rockfish	*Sebastes ruberrimus*	Proposed Threatened

The species' narratives that follow focus on attributes of a species' life history and distribution that influence the manner and likelihood that a particular species may be exposed to the proposed action, as well as the species potential response and risk when exposure occurs. Consequent narratives summarize a larger body of information on worldwide distribution, as well as localized movements within fresh water, estuarine, intertidal, and ocean waters, population structure, feeding, diving, and social behaviors. We also provide a brief summary of the species status and trends as a point of reference for our jeopardy determinations, which we make later in this Opinion. That is, we rely on a species' status and trend to determine whether or not an action's direct or indirect effects are likely to increase the species' probability of becoming extinct. Similarly, each species narrative is followed by a description of its critical habitat with particular emphasis on any essential features of the habitat that may be exposed to the proposed action and may warrant special attention. Because this is a national consultation that does not consider site-specific data, we only summarize information on the geographic distribution of the species, their ecological relationship with waters of the United States, their status, and the principal threats to their survival and recovery.

Species Not Considered Further in This Opinion

Species and Critical Habitat under Joint Jurisdiction

The Services share joint jurisdiction for the management of sea turtles, gulf sturgeon, Atlantic salmon. For sea turtles, NMFS is responsible for their in-water conservation while FWS is responsible for their conservation on land. This Opinion discusses the effects of the proposed action on listed marine sea turtles and their designated critical habitat in the following section.

The Services have divided the consultation responsibilities for Atlantic salmon according to whether the federal action occurs in fresh water or estuarine or marine waters (74 FR 29344). When a federal action traverses marine and fresh waters, then the Services decide which agency will assume the lead role for consultation. For the purposes of this consultation, the FWS' Opinion addresses the effects of the action on Atlantic salmon pursuant to section 7. However, because Atlantic salmon are one of the few species for which direct exposure data are available on the effects of cyanide, this Opinion contains numerous references to this data and its utility in evaluating the effects of cyanide on other species. The full evaluation as to how the federal action affects Atlantic salmon, and whether the action is likely to jeopardize the continued existence of Atlantic salmon is addressed in the FWS' Biological Opinion on cyanide. Similarly, NMFS and FWS share jurisdiction over Gulf sturgeon and generally divide consultations according to whether the federal activity occurs within marine or fresh water. The critical habitat listing for gulf sturgeon clarifies, however, that the FWS will consult with EPA on water quality issues (68 FR 13370). Therefore, the FWS' Biological Opinion on cyanide addresses whether the federal action is likely to jeopardize the continued existence of gulf sturgeon, and the likelihood that the designated critical habitat would be destroyed or adversely modified.

Species and Critical Habitat Not Likely to be Adversely Affected by the Proposed Action

Based upon our analysis, we established that we can concur with EPA's effect determination that a number species are not likely to be adversely affected when exposed to cyanide at criterion values. Specifically, we would not expect the following threatened or endangered species to respond physically, physiologically, or behaviorally to cyanide at the CMC or the CCC: Blue whale, bowhead whale, fin whale, humpback whale, North Atlantic right whale, North Pacific right whale, sei whale, sperm whale, Hawaiian monk seal, Western Steller sea lion, Eastern Steller sea lion, green sea turtle, hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtle, olive ridley sea turtle, smalltooth sawfish, elkhorn coral, staghorn coral, white abalone, black abalone, and Johnson's seagrass. Similarly, we expect the designated critical habitat for the following species is not likely to be adversely affected by cyanide at the CMC or the CCC: North Pacific right whale, Hawaiian monk seal, Western Steller sea lion, Eastern Steller sea lion, green sea turtle, hawksbill sea turtle, leatherback sea turtle, smalltooth sawfish, elkhorn coral, staghorn coral, and Johnson's seagrass. Based upon our analysis, the following proposed species[7] are not likely to be adversely affected when exposed to cyanide at the salt water CMC or the CCC: bocaccio, canary rockfish, spotted seal and yelloweye rockfish. The effects of the proposed action on the Pacific eulachon have not been evaluated.

Listed cetaceans, pinnipeds, sea turtles, marine invertebrates and plants, and marine fishes are distributed in coastal areas that may be exposed to aquatic cyanide. Certain species, like the blue whale and sei whale, are likely to have limited exposure to cyanide sources as their migratory patterns are circumglobal with definite seasonal movements to offshore areas outside the likely extent of cyanide discharges. Nonetheless, we could not conclude that exposures would not occasionally occur, and thus evaluated the potential responses of these species when exposed to cyanide levels equivalent to the salt water CCC and CMC.

Unfortunately, data to evaluate the potential responses of listed marine species or for suitable surrogate species when exposed to cyanide at the recommended aquatic life values is severely lacking. It is for these reasons that Gensemer et al. (2007) declined to evaluate the protectiveness of the saltwater cyanide criteria for marine threatened and endangered species. Pursuant to Section 7 of the ESA, however, we are not proffered the opportunity to withhold judgment. To evaluate the effects of cyanide, particularly on marine species, the lack of data is disconcerting and warrants studies to evaluate response thresholds for more marine species.

In the interim, until further investigations that establish threshold responses are available, current information suggests that the effects of cyanide at the salt water CMC and CCC values of 1.015 µg CN/L on listed marine species and their designated critical habitat, and proposed marine species are extremely unlikely to occur and thus discountable. Our conclusion is based on available data on the responses of marine species relative to the saltwater aquatic life criteria thresholds. The recommended saltwater CMC and CCC are set at very low levels, 1.015 µg CN/L. The CMC value for cyanide was driven by data on the eastern rock crab, *Cancer irroratus*. The species mean acute value for eastern rock crab is 4.893 µg CN/L making the crab six times more sensitive than the next most sensitive marine species, the calanoid copepod, *Acartia tonsa* (EPA 1985). Data were available on the chronic effects of cyanide to only two marine species when EPA established the recommended aquatic life criteria, the mysid, *Mysidopsis bahia*, and the sheepshead minnow, *Cyprinodon variegatus*. Recognizing that these species are relatively resistant to cyanide, EPA set the CCC equal to the CMC because doing so was probably more indicative of the chronic sensitivity of the rock crab than obtained using chronic response data from other species and using other derivation methods (ACR). We found no data to suggest that listed marine species would respond to cyanide exposures at or below 1.015 µg CN/L.

Marine Mammals & Turtles

According to the *Methods Manual*, marine mammals and sea turtles are part of a broad category of "aquatic-dependent" species that whose respiratory oxygen is gained from surface air, not from oxygen dissolved in the water column (like "aquatic species"). For these species, the

7 Proposed species were listed after the completion of EPA's BE. Little data exists to discern adverse effects at levels below the saltwater CCC or CMC. Unlike the other proposed species, the Pacific Eulachon has a freshwater and saltwater life stage. Salt water exposure to cyanide at the CCC and CMC is not likely to result in adverse effects; however, Pacific eulachon still to be evaluated consistent with the approach used to evaluate the effects of the action on other freshwater fishes.

analysis would focus primarily on dietary exposure because this route is generally considered the important route of exposure. The *Methods Manual* expressly discounts dermal or other routes of exposure as areas that are “not explicitly sought in the literature search” when EPA develops the biological evaluations for pollutants but notes that in the event information is uncovered during a literature search that would suggest otherwise, it would be considered in EPA’s effects analysis. Otherwise, the assessment of toxicity on aquatic-dependent listed species, which accounts for all listed marine mammals, sea turtles, and pinnipeds, is based on the estimated dietary effects concentration (dietary EC). The dietary effect would be evaluated by producing estimates of bioconcentration factors (BCFs) and bioaccumulation factors (BAFs). However, there is no published evidence to suggest that cyanide bioaccumulates in fresh- or saltwater aquatic animals. As such exposure to cyanide via the dietary or sediment pathways may not be particularly important.

High doses of cyanide that are ingested can be rapidly lethal (doses exceeding the saltwater CCC), and low doses of cyanide are rapidly metabolized and excreted. Eisler (1991) suggested that repeated sublethal dietary doses may be tolerated by many species for extended periods. The acute oral toxicity of cyanide was calculated on a small set of surrogate species and based on the wet weight of the oral dose. Species used for this analysis ranged from a variety of birds to small and large mammals such as rats, and cows. The minimum acute dietary LD_{50} for birds is 1.4 mg/kg body mass and for mammals is 2.2 mg/kg body mass. Based on these values, marine mammals, sea turtles, and pinnipeds would have to consume cyanide well in excess of the saltwater CMC to experience a lethal response. The saltwater CMC is also likely set below any potential chronic dietary threshold for marine mammals and turtles.

EPA also evaluated toxicity values for a wide range of food items, grouping them into common categories (e.g., insects, invertebrates, fish, etc). Calculated response values were above the CMC and the CCC for both saltwater and freshwater environments. Although the central tendency of the response value was used for the assessment, and not the 5th percentile conservative estimate as was used for listed species, we expect this approach provides a reasonable estimate of adverse effects to prey species particularly given that most of NMFS’ species are generalist feeders and a minor reduction in a particular food item should generally result in discountable and insignificant effects to listed species. For instance, the fin whale is a baleen whale and eats krill, a tiny crustacean. As mentioned previously, the species most sensitive to cyanide is the eastern rock crab. The threshold values from the eastern rock crab were used to determine the effect that cyanide may have on krill. Similarly, the loggerhead sea turtle feeds on mollusks, sponges and crabs. The food item analysis conducted by EPA for this species, was driven by the EC for mollusks (4.7) but should have been reviewed against the invertebrate EC (2.2), because it eats invertebrates and mollusks the dietary analysis should have been reviewed against the lowest EC possible. Nonetheless, the outcome remains the same in this instance—that is, marine food items should not be adversely affected by cyanide at the saltwater criteria.

Based on the best scientific and commercial data available, as discussed previously, we do not expect that the proposed action would adversely affect the quantity, quality or availability any of the constituent elements of critical habitat, or the physical, chemical, or biotic phenomena that give the designated area value for the conservation of the species when no constituent elements were identified in the designation. Although through the proposed action, we would expect critical habitat for North Pacific right whale, Hawaiian monk seal, Western Steller sea lion, Eastern Steller sea lion, green sea turtle, hawksbill sea turtle, the leatherback sea turtle, and proposed critical habitat for the leatherback sea turtle would be exposed to cyanide, the concentration of cyanide would be sufficiently low that we expect the effects would be discountable. As reviewed in the above summary, there is little evidence to discern the effects of cyanide at levels as low as recommended by EPA in the saltwater aquatic life criteria. That said, the data that is available suggests that 1.015 µg CN/L is not likely to adversely alter water quality that supports growth and development, feeding and food resources, reproduction, areas for nesting and reproduction, or other physical, chemical or biological attributes of critical habitat for these species.

Marine Invertebrates and Plants

No dose-response data is available to derive a lethal threshold for *Acropora* species. Much of the data on corals is largely from studies that have examined the effects of the very destructive practice of cyanide fishing, which tends to employ cyanide concentrations well in excess of the saltwater criteria. At high doses, cyanide kills coral, causes loss of zooxanthellae, impaired photosynthesis, disruption of protein synthesis and altered rates of mitosis (Jones and Steven 1997; Jones and Hoegh-Guldberg 1999; Cato and Brown 2003; Cervino 2003). A few studies have been conducted on the short-term exposure of coral species to sublethal concentrations, but the concentrations have been well above the saltwater criteria. According to Dzombak et al. (2006) some studies have observed no response of coral to cyanide exposures at concentrations as low as 26 µg CN/L. More research is needed to discern the response threshold for listed species. However, given the limited data available at this time, it appears that exposure to cyanide at the low concentrations recommended by the aquatic life criteria that that any effects would likely be discountable and insignificant.

We also have very little data to suggest what the threshold response concentrations would be for marine plants. Evidence suggests that some plants are capable of transforming cyanide through enzymatic activity and can avoid cyanide intoxication by directly degrading the cyanogenic compounds or assimilating them into their metabolism. The effectiveness of this response would depend upon the plant, the balance of activity and the exposure concentration. EPA's best estimate of response thresholds is based on the freshwater blue-green algae, *Microcystis aeruginos*, and the marine red algae. The latter has a NOEC of 11 µg CN/L, well above the saltwater CMC or CCC. Using red algae as a surrogate to predict the response of Johnson's sea grass, we expect the effects of cyanide at the aquatic life criteria would be discountable and insignificant.

There were too few data available to generate a species sensitivity distribution for white or black abalone through the class level. We found only one study on the effect of cyanide on an abalone species, the *Haliotis varia*, the varied ear shell or variable abalone. Given that the varied ear shell abalone is within the same genus, the reported LC_{50} of 1012 µg CN/L is the best estimate of a lethal response for both black abalone and white abalone. Lasut (1999) studied the effects of cyanide and salinity on the mortality of abalone and found that mortality increased within decreased salinity. Abalone subjected to lethal concentrations of potassium cyanide and sodium cyanide experienced a 19% increase risk of mortality when exposed to 25‰ salinity over that observed in 34‰ salinities. Even so, the response occurs well above the saltwater CMC. Therefore, we would not expect the species would be adversely affected when exposed to cyanide at the CMC saltwater value of 1.015 µg CN/L.

Based on the best scientific and commercial data available, as discussed previously, we do not expect that the proposed action would adversely affect the quantity, quality or availability any of the constituent elements of critical habitat, or the physical, chemical, or biotic phenomena that give the designated area value for the conservation of the species when no constituent elements were identified in the designation. Although through the proposed action, we would expect critical habitat for elkhorn coral, staghorn coral, and Johnson's seagrass would be exposed to cyanide, the concentration of cyanide would be sufficiently low that we expect the effects would be discountable. As reviewed in the above summary, there is little evidence to discern the effects of cyanide at levels as low as recommended by EPA in the saltwater aquatic life criteria. That said, the data that is available suggests that 1.015 µg CN/L is not likely to adversely alter water quality that supports growth and development, feeding and food resources, reproduction, areas for nesting and reproduction, or other physical, chemical or biological attributes of critical habitat for these species.

Marine Fishes

Too few data exist to generate a species sensitivity distribution estimate for this smalltooth sawfish, or the recently proposed rockfish species, bocaccio, yelloweye, and canary rockfish, through the class level. In comparison of the mean LC50 and NOEC values for the most closely related marine fishes range from 59.3 to 372 and 5.608 to 35.18 µg CN/L, respectively. Data on most acutely sensitive marine fish, the Atlantic silverside, results in acute and chronic EC_As of 26.12 and 5.608 µg CN/L, in the range of the most acutely sensitive freshwater fish species. Since insufficient data are available to model species sensitivity distributions for marine species, we relied on the calculated EC_As of the most sensitive marine fish for which data was available in making our effects determination. Although not included in EPA's biological evaluation, the three proposed rockfish would be evaluated using the same EC_A values, as not enough data exists to employ other evaluation methods. As such, data on the Atlantic silverside suggests that the saltwater cyanide criteria would likely result in discountable and insignificant effects on bocaccio, yelloweye, and canary rockfish, and smalltooth sawfish.

Based on the best scientific and commercial data available, as discussed previously, we do not expect that the proposed action would adversely affect the quantity, quality or availability any of the essential features of critical habitat. Although through the proposed action, we would expect critical habitat for smalltooth sawfish would be exposed to cyanide, the concentration of cyanide would be sufficiently low that we expect the effects would be discountable. As reviewed in the above summary, there is little evidence to discern the effects of cyanide at levels as low as recommended by EPA in the saltwater aquatic life criteria. That said, the data that is available suggests that 1.015 µg CN/L is not likely to adversely alter water quality that supports growth and development, feeding and food resources, reproduction, areas for nesting and reproduction, or other physical, chemical or biological attributes of critical habitat for these species.

Species and Critical Habitat Likely to be Adversely Affected by the Proposed Action

Anadromous Fishes

Chinook Salmon

Description of the Species

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). In this section, we discuss the distribution, status, and critical habitats of the nine species[8] of endangered and threatened Chinook salmon separately, and summarize their common dependence on waters of the United States. However, because Chinook salmon in the wild are virtually indistinguishable between listed species, and are the same biological species we begin this section describing those characteristics common across ESUs (the listed species).

Of the Pacific salmon species considered herein, Chinook salmon exhibit arguably one of the most diverse and complex life history strategies with multiple races within which there is substantial variation. One form, the “stream-type”, resides in freshwater for a year or more following emergence and the “ocean-type” migrates to the ocean within their first year. The ocean-type typifies populations north of 56ºN (Healy 1991). Within each race, there is often variation in age at seaward migration, age of maturity, timing of spawning migrations, male precocity, and female fecundity.

The general Chinook salmon life cycle spans fresh and marine waters, with one reproductive event per adult (that is, Chinook salmon are semelparous and die after spawning). Spawning migrations generally occur in the spring and fall, although the precise timing of spawning migrations and spawning varies across populations and can vary within populations. Temperature and stream flow can significantly influence the timing of upstream migrations and spawning, and the selection of spawning habitat (Geist et al. 2009; Hatten and Tiffan 2009). However, a general latitudinal cline is apparent across the species’ range with spawning typically occurring earlier in the spring/summer at northern latitudes and later in southern latitudes (Healy 1991).

On the spawning grounds, mate competition is intense with males competing to fertilize eggs and females competing for optimal nest site selection. Once fertilization occurs, female Chinook salmon bury the eggs in nests –termed “redds”- and they guard the nests until their death, which generally occurs a couple days later to a couple weeks after spawning. A female generally deposits eggs in more than one depression within a redd, excavating stream rock as she moves upstream, increasing the size of her redd until all eggs are deposited.

Size and age at maturity is partially under genetic control, but can be influenced by environment

[8] We use the word “species” as it has been defined in section 3 of the ESA, which include “species, subspecies, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature (16 U.S.C 1533).” Pacific salmon that have been listed as endangered or threatened were listed as “evolutionarily significant units (ESU)” which NMFS uses to identify distinct population segments (DPS) of Pacific salmon. Any ESU or DPS is a “species” for the purposes of the ESA.

and migration behavior (Roni and Quinn 1995). Generally, ocean-type salmon are at sea longer than their stream-type counterparts and tend to be larger in size at spawning. Body size can be important in determining reproductive success in terms of nest selection and mating competition (Foote 1990). Chinook salmon age at maturity ranges from 1 to 7 years with most returning to spawn between 3 and 4 years of age.

The time necessary for egg incubation until emergence of alevins in fresh water varies among basins and among years within a basin, and is closely correlated to water temperatures such that low temperatures can prolong incubation. Incubation generally takes a couple of months or more. Alevin (also called "yolk-sac" fry) remain buried until their yolk-sac is absorbed, at which time they become free swimming fry. Egg to fry survival can also vary widely across basins, years, and habitat conditions within a basin. In general, the survival of eggs and alevin, and the fitness of emerging fry are affected by sediment loading, intergravel water flow and dissolved oxygen levels, gravel composition, spawn timing, floods, redd and spawner density, and water temperatures.

Once emerged, fry behavior varies among populations and among individuals within races. Some juvenile Chinook salmon rear in fresh water for a few weeks to a few years, others move immediately downstream coastal waters where they rear in estuaries for a few weeks to months, while others migrate directly to ocean waters. Stream-type Chinook salmon do not migrate to sea until the spring following emergence, and ocean-type Chinook salmon migrate to the ocean within their first year. Generally, most fry move at night probably to reduce detection by predators, although some fish will move downstream during daylight. Not all movement is volitional as stream flows often displace fry to downstream areas after emergence. Density-dependent factors such as space, prey, or stream flows may influence the outmigration behavior of individual juvenile Chinook salmon.

While in fresh water, juvenile Chinook salmon are often found in the lower reaches of a river near its estuary, where they inhabit river margins in areas of shallow water, near woody debris, or other areas of low water velocity. As juveniles grow in size, they tend to move away from the shoreline to deeper waters where the velocity is higher (Healey 1991). Generally, Chinook salmon outmigrants (termed smolts) are about 2 to 5 inches long when they enter saline (often brackish) waters. The process of smoltification is a physiologically demanding process that enables salmon to adapt to sea water and maintain the appropriate osmotic pressure necessary to maintain body fluid concentration and composition, and homeostasis as the fish enters waters of increased salinity. The transformation from the fresh water fry/parr juvenile stage to smolt involves multiple physiological changes including an increase in: body silvering, hypoosmotic regulatory capability, salinity tolerance and preference, growth rate, oxygen consumption, ammonia production, endocrine activity (e.g., activation of thyroid, interregnal and pituitary growth hormone cells), and gill Na^+, K^+-ATPase activity. At the same time, the ratio of weight standardized to length (condition factor) declines and total body lipid content declines (Wedemeyer et al. 1980). Several factors can affect smoltification process, not only at the interface between fresh water and salt water, but higher in the watershed as the process of transformation begins long before fish enter salt waters including: exposure to chemicals such as heavy metals, and elevated water temperatures (Wedemeyer et al. 1980).

Life at sea varies according to population, race, and age-class. Chinook salmon tend to remain at sea between 1 and 6 years, with most fish returning to fresh water after 2 to 4 years at sea. Fishery catches indicate that ocean- and stream-type fish exhibit divergent migratory pathways while in the ocean (Healey 1983, 1991). Ocean-type Chinook salmon tend to be found along the coastline, whereas stream-type Chinook salmon are found in the open ocean far from the coast (Healey 1983, 1991).

Chinook salmon feed on a variety of prey organisms depending upon life stage. Adult oceanic Chinook salmon eat small fish, amphipods, and crab megalops (Healey 1991). Fish, in particular herring, make up the largest portion of an adult Chinook salmon's diet. In estuaries, Chinook salmon smolts tend to feed on chironomid larvae and pupae, *Daphnia*, *Eogammarus*, *Corphium* and *Neomysis*, as well as juvenile herring, sticklebacks and other small fish. In fresh water, Chinook salmon juveniles feed on adult and larval insects including terrestrial and aquatic insects such as dipterans, beetles, stoneflies, chironomids, and plecopterans (Healey 1991).

Threats

Natural Threats. Chinook salmon are exposed to high rates of natural predation during freshwater rearing and migration stages, as well as during ocean migration. In general, Chinook salmon are prey for pelagic fishes, birds, and marine mammals, including harbor seals, sea lions, and killer whales. There have been recent concerns that the increasing size of tern, seal, and sea lion populations in the Pacific Northwest may have reduced the survival of some salmon species.

Anthropogenic Threats. Salmon survive only in aquatic ecosystems and, therefore, depend on the quantity and quality of those ecosystems. Chinook salmon have declined under the combined effects of fishery over-harvest; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile Chinook salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific Northwest (Buhle et al. 2009).

Salmon along the west coast of the United States share many of the same threats. Therefore, anthropogenic threats for all species and populations are summarized here. Population declines have resulted from several human-mediated causes, but the greatest negative influence has likely been the establishment of waterway obstructions such as dams, power plants, and sluiceways for hydropower, agriculture, flood control, and water storage. These structures have blocked salmon migration to spawning habitat or resulted in direct mortality and have eliminated entire salmon runs as a result. While some of these barriers remain, others have been reengineered, renovated, or removed to allow for surviving runs to access former habitat, but success has been limited. These types of barriers alter the natural hydrograph of basins, both upstream and downstream of the structure, and significantly reduce the availability and quality of spawning and rearing habitat (Hatten and Tiffan 2009). Many streams and rivers, particularly in urban or suburban areas, suffer from streamside development, which contributes sediment, chemical pollutants from pesticide applications and automobile or industrial activities, altered stream flows, loss of streamside vegetation and allochthonous materials to name a few. These factors can directly cause mortality, reduce reproductive success, or affect the health and fitness of all salmon life stages.

Artificial propagation of hatchery fish has had profound consequences on the viability of some natural salmon populations, but there are potential benefits to the artificial production of salmon as well. Adverse effects of artificial propagation include: a decline in the natural population from the taking of wild broodstock for artificial propagation, the genetic erosion of populations (introgression, hybridization), an increased incidence of disease in the wild and increased rates of competition with and predation on naturally spawned salmon populations. Potential benefits to artificial propagation include the bolstering of the numbers of naturally spawning fish in the short-term, the conservation of genetic resources, and guarding against the catastrophic loss of naturally spawned populations at critically low abundance levels.

Fishing for salmon has also negatively impacted salmon populations. Fishing reduces the number of individuals within a population and can lead to uneven exploitation of certain populations and size classes (Reinsenbichler 1997; Mundy 1997). Targeted fishing of larger individuals results in excluding the most fecund individuals from spawning (Reinsenbichler 1997). Genetic changes that promote smaller body sizes have occurred in heavily exploited populations in response to size-selective harvest pressures (Reinsenbichler 1997; Mundy 1997; Swain et al. 2007). Fishing pressure can reduce age at maturity in fished populations as the fished populations compensate for the reductions in the numbers of spawning adults (Reinsenbichler 1997).

Pacific salmon species are exposed to a number of contaminants throughout their range and life history cycle. Exposure to pollution is also of significant concern for all life stages, but is likely particularly significant for freshwater life stages. Organic pollutants, particularly PCBs, DDT and its congeners, pesticides, and endocrine disruptors are of particular concern. These chemicals can inhibit smell, disrupt reproductive behavior and physiology, impair immune function, and lead to mortality through impairment of water balance when traveling between fresh and salt water systems (Varanasi et al. 1993). Diffuse and extensive population centers contribute increase contaminant volumes and variety from such sources as wastewater treatment plants and sprawling development. Urban runoff from impervious surfaces and roadways often contains oil, copper, pesticides, PAHs, and other chemical pollutants and flow into surface waters. Point and nonpoint pollution sources entering rivers and their tributaries affect water quality in available spawning and rearing habitat for salmon. Juvenile salmonids that inhabit urban watersheds often carry high contaminant burdens, which is partly attributable to the biological transfer of contaminants through the food web (Brown et al. 1985; Stein et al. 1992; Varanasi et al. 1993).

Climate change poses significant hazards to the survival and recovery of salmonids along the west coast. Paleoecological data (which exclude anthropogenic influences) suggest regional and global climate factors on decadal, centennial, and millennial time scales are tied to abundance patterns of Pacific salmonids (Finney et al. 2009). Increases in global temperatures are likely to have profound effects on salmonids directly and indirectly through altered hydrological regimes. Increases in instream temperatures may decrease habitat available for refugia, increase species interactions and competition, accelerate incubation timing and premature emergence, increase susceptibility to parasites and disease, reduce fry survival, delay migration and spawning, and accelerate loss of energy reserves. Using emission scenarios from the Intergovernmental Panel on Climate Change (IPCC), O'Neal (2002) estimates that direct thermal changes in freshwater temperatures could cause the loss of between 4-20% of existing salmon and trout habitat by the year 2030, 7-34% by 2060, and 14-42% by 2090, depending on the trout or salmon species, IPCC emission scenario considered, and the model used. Projected salmon habitat loss would be most severe in Oregon and Idaho, at losses of 40% or greater of 2007 habitat estimates. While the predicted losses are substantial, the estimates may underestimate the overall effect global climate change will have on salmon and trout abundance since these models do not consider the related effects from changes in seasonal hydrological patterns and water volumes that result from altered weather patterns and precipitation (O'Neal 2002).

Changes in hydrological regimes are closely linked to salmon abundance (Hicks et al. 1991; Clark et al. 2001). From studies that have examined the effects of timber harvest and other changes in land use patterns, we know that changes in hydrology (i.e., increased peak flows, decreased low flows, altered timing discharge events, and rapid fluctuations in flows) can profoundly affect salmon abundance and the amount and availability of quality habitat. Hydrology is strongly correlated to in-redd and young of the year survival, can lead to the displacement of young fish, alter immigration and emigration timing, alter the volume of available habitat by affecting channel structure (e.g., pool to riffle ratios, debris loading, substrate composition, erosion and sediment loading) and the relative abundance of salmon and trout species within a watershed, as well as the relative abundance of age-classes (see Hicks et al. 1991; Gregory and Bisson 1997). Such ecosystem changes are also likely to alter macroinvertebrate communities and habitats, affecting important forage for salmon and trout (McCarthy et al. 2009; Williams et al. 2009).

Upstream changes in riverine habitat can affect downstream estuarine ecosystems through alterations in sediment delivery (timing and volume), and changes in freshwater volumes and timing can influence the volume of the spring/summer salt-wedge (O'Neal 2002). In turn, changes in the trophic dynamics of the estuary may occur. At the same time, physical changes in the ocean associated with warming include increases in temperature, increased water column stratification, and changes in the intensity and timing of coastal upwelling. These changes will alter primary and secondary productivity, the structure of marine communities, and, in turn, the growth, productivity, survival, and migrations of salmonids. Changing ocean temperatures may alter salmon behavior, distribution, and migrations, increasing the distance from home streams to ocean feeding areas. Energetic demands increase at warmer temperatures, requiring increased feeding to maintain growth. This could lead to intensified competition for food and reduction in growth rates, further exacerbating the prey/predator relationship. Increasing concentrations of carbon dioxide in the oceans lowers pH, which reduces the availability of carbonate for shell-forming marine animals. Pteropods are expected to be negatively affected, and they can comprise more than 40% of some salmon diets. If salmon migrate farther to the north and/or food is less available, longer times may be required to reach maturity, delaying return of adult migrations into coastal water and rivers.

California Coastal Chinook Salmon

Distribution and Description of the Listed Species

The California Coastal Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from rivers and streams south of the Klamath River to the Russian River, California. Seven artificial propagation programs are part of this ESU: The Humboldt Fish Action Council (Freshwater Creek), Yager Creek, Redwood Creek, Hollow Tree, Van Arsdale Fish Station, Mattole Salmon Group, and Mad River Hatchery fall-run Chinook hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

California Coastal Chinook salmon are a fall-run, ocean-type fish. A spring-run (river-type) component existed historically, but is now considered extinct (Bjorkstedt et al. 2005). Table 3 identifies populations within the California Coastal Chinook salmon ESU, their abundances, and the relative contribution of artificially propagated fish to the population.

Table 3. California coastal Chinook populations and selected measures of population viability

Population	Historical Abundance[a]	Mean Number of Spawners (Range)[b]	Percent Hatchery Contribution[c]	Long-term Trend[d]
Freshwater Creek		22 (13-22)	30-70	0.137 (-0.405, 0.678)
Eel River	17,000-55,000		~30	
Mainstem Eel River	13,000			
Sprowl Creek		43 (43-497)		-0.096 (-0.157, -0.034)
Tomki Creek		61 (13-2233)		-0.199 (-0.351, -0.046)
Van Duzen River	2,500			
Middle Fork Eel River	13,000			
South Fork Eel River	27,000			
North Fork Eel River				
Upper Eel River				
Redwood Creek	1,000-5,000			
Mad River	1,000-5,000			
Canyon Creek		73 (19-103)		0.0102 (-0.106, 0.127)
Bear River	100			
Mattole River	1,000-5,000		~17	
Russian River	50-500		~0	
Humbolt Bay tributaries	40			
Tenmile to Gualala			0	
Small Humboldt County rivers	1,500		0	
Rivers north of Mattole River	600		0	
Noyo River	50		0	

[a]Historical abundance estimates based on professional opinion and evaluation of habitat conditions (reported in Good et al. 2005).
[b]5-year (1997-2001) geometric mean number of counts of adults (quasi-systematic surveys of spawners – Canyon, Tomki, and Sprowl creeks; returning spawners at Freshwater Creek weir).
[c]Hatchery production in this ESU is at low levels, aimed at supplementing depressed runs. Operational procedures and low production suggest that the ESU may not be at substantial risk of degraded genetic integrity (Good et al. 2005).

[d]Long-term trends were calculated using the entire available data set (see Good et al. 2005). The 90% confidence intervals are noted in parentheses.

Status and Trends

NMFS listed California Coastal Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and they retained their threatened status on June 28, 2005 (70 FR 37160). California Coastal Chinook salmon were listed due to the combined effect of dams that prevent them from reaching spawning habitat, logging, agricultural activities, urbanization, and water withdrawals in the river drainages that support them. Historical estimates of escapement, based on professional opinion and evaluation of habitat conditions, suggest abundance was roughly 73,000 in the early 1960s with the majority of fish spawning in the Eel River (CDFG 1965 *in* Good et al. 2005). The species exists as small populations with highly variable cohort sizes. The Russian River probably contains some natural production, but the origin of those fish is not clear because of a number of introductions of hatchery fish over the last century. The Eel River contains a substantial fraction of the remaining Chinook salmon spawning habitat for this species. Since its original listing and status review, little new data are available or suitable for analyzing trends or estimating changes in this population's growth rate (Good et al. 2005).

Long-term trends in Freshwater Creek are positive, and in Canyon Creek, although only slightly different than zero, the trend is positive (Table 3). Long-term trends in Sprowl and Tomki creeks (tributaries of the Eel River), however, are negative. Good et al. (2005) caution making inferences on the basin-wide status of these populations as they may be weak because the data likely include unquantified variability due to flow-related changes in spawners' use of mainstem and tributary habitats. Unfortunately, none of the available data is suitable for analyzing the long-term trends of the ESU or estimating the population growth rate.

Critical Habitat

NMFS designated critical habitat for California Coastal Chinook salmon on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Redwood Creek, Trinidad, Mad River, Eureka Plain, Eel River, Cape Mendocino, Mendocino Coast, and the Russian River. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, California Coastal Chinook salmon occupy 45 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 1,500 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 45 watershed reviewed in NMFS' assessment of critical habitat for California Coastal Chinook salmon, eight watersheds received a low rating of conservation value, 10 received a medium rating, and 27 received a high rating of conservation value for the species.

Critical habitat in this ESU consists of limited quantity and quality summer and winter rearing habitat, as well as marginal spawning habitat. Compared to historical conditions, there are fewer pools, limited cover, and reduced habitat complexity. The limited instream cover that does exist is provided mainly by large cobble and overhanging vegetation. Instream large woody debris, needed for foraging sites, cover, and velocity refuges is especially lacking in most of the streams throughout the basin. NMFS has determined that these degraded habitat conditions are, in part, the result of many human-induced factors affecting critical habitat including dam construction, agricultural and mining activities, urbanization, stream channelization, water diversion, and logging, among others.

Central Valley Spring-Run Chinook Salmon

Distribution and Description of the Listed Species

The Central Valley spring-run Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California. This ESU includes one artificial propagation program, the Feather River Hatchery spring-run Chinook salmon program. This artificially propagated population is no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Central Valley spring-run Chinook salmon ESU includes Chinook salmon entering the Sacramento River from March to July and spawning from late August through early October, with a peak in September. Spring-run fish in the Sacramento River exhibit an ocean-type life history, emigrating as fry, sub-yearlings, and yearlings. Central Valley spring-run Chinook salmon require cool freshwater while they mature over the summer.

Status and Trends

NMFS originally listed Central Valley spring-run Chinook salmon as threatened on September 16, 1999 (64 FR 50393), a classification this species retained on June 28, 2005 (70 FR 37160). This species was listed because dams isolate them from most of their historic spawning habitat and the habitat remaining to them is degraded. Historically, spring-run Chinook salmon were predominant throughout the Central Valley occupying the upper and middle reaches (1,000 to 6,000 feet) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud and Pit Rivers, with smaller populations in most tributaries with sufficient habitat for over-summering adults (Stone 1874; Rutter 1904; Clark 1929).

Table 4. Central Valley spring-run Chinook salmon populations and selected measures of population viability

Population	Historical Abundance[a]	Mean Number of Spawners (Range)[b]	Percent Hatchery Contribution[c]	Mean Annual Population Growth Rate (λ)[d]
Butte Creek spring-run		4,513 (67-4,513)		1.30 (1.09-1.60)
Deer Creek spring-run		1,076 (243-1,076)		1.17 (1.04-1.35)
Mill Creek spring-run		491 (203-491)		1.19 (1.00-1.47)

[a]Historical abundance for the total ESU, based on gillnet fishery catches, is estimated at about 700,000 (Fisher 1994). Individual river estimates of historical abundance not provided.
[b]Recent geometric mean number of spawners as reported by Good et al. 2005. Note the current geometric mean for Butte, Deer and Mill creeks are also the maximum means.
[c]Between 1967 and 1999 the Feather River Hatchery released between less than 1 million to as much as 5.5 million spring-run Chinook salmon in any given year. Returns ranged from less than 1,000 spawners to about 7,000 in the late 1980s (see Good et al. 2005). No other hatchery data reported.
[d]The λ calculation, provided by Good et al. 2005, is an estimate of the population growth rate. The 90% confidence intervals are noted in parentheses.

The Central Valley drainage as a whole is estimated to have supported spring-run Chinook salmon runs as large as 700,000 fish between the late 1880s and the 1940s (Fisher 1994), although these estimates may reflect an already declining population, in part from the commercial gillnet fishery that occurred in this ESU (Good et al. 2005). Before construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone (Fry 1961). Following the completion of Friant Dam, the native population from the San Joaquin River and its tributaries (i.e., the Stanislaus and Mokelumne Rivers) was extirpated. Spring-run Chinook salmon no longer exist in the American River due to the operation of Folsom Dam. Naturally spawning populations of Central Valley spring-run Chinook salmon currently are restricted to accessible reaches of the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Feather River, Mill Creek, and Yuba River (CDFG 1998). Since 1969, the Central Valley spring-run Chinook salmon ESU (excluding Feather River fish) has displayed broad fluctuations in abundance ranging from 25,890 in 1982 to 1,403 in 1993 (CDFG unpublished data *in* Good et al. 2005).

The average abundance for the ESU was 12,499 for the period of 1969 to 1979, 12,981 for the period of 1980 to 1990, and 6,542 for the period of 1991 to 2001. In 2003 and 2004, total run size for the ESU was 8,775 and 9,872 adults respectively, well above the 1991 to 2001 average. Evaluating the ESU as a whole, however, masks significant changes that are occurring among populations that comprise the ESU (metapopulation). For example, the mainstem Sacramento River population has undergone a significant decline while the abundance of many tributary populations increased. Average abundance of Sacramento River mainstem spring-run Chinook salmon recently declined from a high of 12,107 for the period 1980 to 1990, to a low of 609 for the period 1991 to 2001, while the average abundance of Sacramento River tributary populations increased from a low of 1,227 to a high of 5,925 over the same periods.

Abundance time series data for Mill, Deer, Butte, and Big Chico creeks spring-run Chinook salmon confirm that population increases seen in the 1990s have continued through 2001 (Good et al. 2005). Habitat improvements, including the removal of several small dams and increases in summer flows in the watersheds, reduced ocean fisheries, and a favorable terrestrial and marine climate, have likely contributed to this. All three spring-run Chinook salmon populations in the

Central Valley have long-and short-term positive population growth. Although the populations are small, Central Valley spring-run Chinook salmon have some of the highest population growth rates in the Central Valley.

Critical Habitat

NMFS designated critical habitat for Central Valley spring-run Chinook salmon on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Tehama, Whitmore, Redding, Eastern Tehama, Sacramento Delta, Valley-Putah-Cache, Marysville, Yuba, Valley-American, Colusa Basin, Butte Creek, and Shasta Bally hydrological units. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Central Valley spring-run Chinook salmon occupy 37 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 1,100 miles of stream habitat and about 250 square miles of estuarine habitat in the San Francisco-San Pablo-Suisun Bay complex. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 37 watersheds reviewed in NMFS' assessment of critical habitat for Central Valley spring-run Chinook salmon, seven watersheds received a low rating of conservation value, three received a medium rating, and 27 received a high rating of conservation value for the species.

Factors contributing to the downward trends in this ESU include: reduced access to spawning/rearing habitat behind impassable dams, climatic variation, water management activities, hybridization with fall-run Chinook salmon, predation, and harvest (CDFG 1998). Several actions have been taken to improve and increase the primary constituent elements of critical habitat for spring-run Chinook salmon, including improved management of Central Valley water (e.g., through use of CALFED Environmental Water Account and Central Valley Project Improvement Act (b)(2) water accounts), implementing new and improved screen and ladder designs at major water diversions along the mainstem Sacramento River and tributaries, removal of several small dams on important spring-run Chinook salmon spawning streams, and changes in ocean and inland fishing regulations to minimize harvest. Although protective measures and critical habitat restoration likely have contributed to recent increases in spring-run Chinook salmon abundance, the ESU is still below levels observed from the 1960s through 1990.

Threats from hatchery production (i.e., competition for food between naturally spawned and hatchery fish, and run hybridization and homogenization), climatic variation, reduced stream flow, high water temperatures, predation, and large scale water diversions persist.

Lower Columbia River Chinook Salmon

Distribution and Description of the Listed Species

The Lower Columbia River Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Washington and Oregon, east of the Hood River and the White Salmon River, and includes the Willamette River to Willamette Falls, Oregon, exclusive of spring-run Chinook salmon in the Clackamas River. Seventeen artificial propagation programs are part of this ESU: The Sea Resources Tule, Big Creek Tule, Astoria High School (STEP) Tule, Warrenton High School (STEP) Tule, Elochoman River Tule, Cowlitz Tule, North Fork Toutle Tule, Kalama Tule, Washougal River Tule, Spring Creek National Fish Hatchery Tule, Cowlitz spring (Upper Cowlitz River and Cispus River), Friends of the Cowlitz spring, Kalama River spring, Lewis River spring, Fish First spring, and the Sandy River Hatchery Chinook salmon programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Lower Columbia River Chinook salmon have three life history types, including early fall runs (tules), late fall runs (brights), and spring-runs. Spring and fall runs have been designated as part of a Lower Columbia River Chinook salmon ESU. The Cowlitz, Kalama, Lewis, White Salmon, and Klickitat Rivers are the major river systems on the Washington side, and the lower Willamette and Sandy Rivers are foremost on the Oregon side. The eastern boundary for this species occurs at Celilo Falls, which corresponds to the edge of the drier Columbia Basin Ecosystem and historically may have been a barrier to salmon migration at certain times of the year. The predominant life history type for this species is the fall-run. Fall Chinook salmon typically enter the Columbia River in August through October to spawn in the mainstem of the large rivers (Kostow 1995). Spring Chinook salmon enter freshwater in March through June to spawn in upstream tributaries and generally emigrate from fresh water as yearlings.

Status and Trends

NMFS originally listed Lower Columbia River Chinook salmon as threatened on March 24, 1999 (64 FR 14308); NMFS reaffirmed the threatened status of Lower Columbia River Chinook salmon on June 28, 2005 (70 FR 37160). Historical records of Chinook salmon abundance are sparse, but cannery records suggest a peak run of 4.6 million fish (43 million pounds) in 1883 (Lichatowich 1999). Although fall-run Chinook salmon are still present throughout much of their historical range, they are still subject to large-scale hatchery production, relatively high harvest, and extensive habitat degradation. The Lewis River late-fall-run Chinook salmon population is the healthiest and has a reasonable probability of being self-sustaining. Abundances largely declined during 1998 to 2000 and trend indicators for most populations are negative, especially if hatchery fish are assumed to have a reproductive success equivalent to that of natural-origin fish (see Table 5).

Most populations for which data are available have a long-term declining population trend (Table 5). Currently, the spatial extent of populations in the Coastal and Cascade fall runs are similar to their respective historical conditions. New data include spawner abundance estimates through 2001, new estimates of the fraction of hatchery spawners, and harvest estimates. In addition, estimates of historical abundance have been provided by the Washington Department of Fish and Wildlife. The Willamette/Lower Columbia River Technical Review Team estimated that 8 to 10 historic populations have been extirpated, most of them spring-run populations. Near loss of that important life history type remains an important concern. Although some natural production currently occurs in 20 or so populations, only one exceeds 1,000 spawners. Almost all spring-run Chinook salmon are at very high risk of extinction. High hatchery production continues to pose genetic and ecological risks to natural populations and to mask their performance for Coastal, Cascade, and Gorge fall run populations. Most Lower Columbia River Chinook salmon populations have not seen increases in recent years as pronounced as those that have occurred in many other geographic areas.

Table 5. Lower Columbia River Chinook salmon life histories, populations and selected measures of population viability

Life History	Population	Historical Abundance[a]	Mean Number of Spawners (range)[b]	Percent Hatchery Contribution[c]	Long-term Median Growth Rate (λ)[d]
Fall run	Youngs Bay				
	Grays River	2,477	99	38	0.944, 0.844
	Big Creek				
	Elochoman River		676	68	1.037, 0.800
	Clatskanie River[e]		50 (34-74)		0.99
	Mill, Abernathy, and Germany Creeks		734	47	0.981, 0.829
	Scappoose Creek				
	Coweeman River	4,971	274	0	1.092, 1.091
	Lower Cowlitz River	53,956	1,562	62	0.998, 0.682
	Upper Cowlitz River		5,682		
	Toutle River	25,.392			
	Kalama River	22,455	2,931	67	0.973, 0.818
	Salmon Creek and Lewis River	47,591[f]	256	0	0.984, 0.979
	Clackamas River		40		
	Washougal River	7,518	3,254	58	1.025, 0.815
	Sandy River		183		
	Columbia Gorge-lower tributaries				
	Columbia Gorge-upper tributaries	2,363	136 (Wind River only)	13 (Wind River only	0.959, 0.955
	Hood River		18		
	Big White Salmon River		334	21	0.963, 0.945
Late fall (bright)	Sandy River[e]		3085 (2337-4074)	3	0.997

	North Fork Lewis River		7,841	13	0.968, 0.948
Spring run	Upper Cowlitz River				
	Cispus River		1,787		
	Tilton River				
	Toutle River	2,901			
	Kalama River	4,178	98		
	Lewis River		347		
	Sandy River[e]		297 (202-438)		0.961
	Big White Salmon River				
	Hood River		51		

[a]Historical abundance for various rivers was calculated using the Ecosystem and Diagnosis Treatment (EDT) model, which attempts to predict population performance based on reach-specific habitat attributes. Estimates are provided as a means of comparing the historical abundance of populations relative to current abundance. See Good et al. (2005) for a discussion about the uncertainty associated with these estimates.
[b]Recent geometric mean number of spawners as reported in Good et al. 2005
[c]Recent average hatchery-origin spawners (%) as reported by Good et al. 2005. Natural-origin spawners are those that had parents that spawned in the wild, as opposed to hatchery-origin fish, whose parents were spawned in a hatchery.
[d]The long-term median growth rate (λ) is an estimate of the natural growth rate after accounting for hatchery-origin spawners. The two values are estimates under two hypotheses about the reproductive success of hatchery origin spawners. Hatchery fish are assumed to have zero reproductive success in the first estimate. In the second estimate hatchery fish are assumed to have the same relative reproductive success as natural-origin fish. Growth rates were not calculated for all populations, as adequate data were not available (see Good et al. 2005 for 95% confidence intervals on growth estimates).
[d]Values for these populations are reported in McElhany et al. 2007, and represent estimates based on the total available data series, which varies by population.
[fe]Combined estimate of Lewis River fall run (East Fork only) and Lewis River brights (Good et al. 2005)

Critical Habitat

NMFS designated critical habitat for Lower Columbia River Chinook salmon on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Hood Rivers as well as specific stream reaches in a number of tributary subbasins. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 52 subbasins reviewed in NMFS' assessment of critical habitat for the Lower Columbia River Chinook salmon ESU, 13 subbasins were rated as having a medium conservation value, four were rated as low, and the remaining subbasins (35), were rated as having a high conservation value to Lower Columbia River Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes resulting from hydropower development, loss of tidal marsh and swamp habitat, and degraded freshwater and marine habitat from industrial harbor and port development, and urban development. Limiting factors identified for this species include reduced access to spawning/rearing habitat in tributaries, hatchery impacts, loss of habitat diversity and channel stability in tributaries, excessive fine sediment in spawning gravels, elevated water temperature in tributaries, and harvest impacts.

Upper Columbia River Spring-run Chinook Salmon

Distribution and Description of the Listed Species

The Upper Columbia River spring-run Chinook salmon ESU includes all naturally spawned populations of Chinook salmon in all river reaches accessible to Chinook salmon in Columbia River tributaries upstream of Rock Island Dam and downstream of Chief Joseph Dam in Washington, excluding the Okanogan River. Six artificial propagation programs are part of this ESU: the Twisp River, Chewuch River, Methow Composite, Winthrop National Fish Hatchery, Chiwawa River, and White River spring-run Chinook salmon hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU. Spring-run Chinook salmon currently spawn in only three river basins above Rock Island Dam: the Wenatchee, Entiat, and Methow Rivers. Table 6 identifies the Upper Columbia River Chinook salmon ESU populations, their abundances, and estimates of the proportion of hatchery fish that contribute to the run size.

Upper Columbia River spring-run Chinook salmon begin returning to the Columbia in early spring and enter upper Columbia tributaries from April through July, with a peak in mid-May. After migration, Upper Columbia River spring-run Chinook salmon hold in freshwater tributaries until spawning in late summer, peaking in mid- to late August. Juvenile spring-run Chinook salmon remain in fresh water for a full year before emigrating to salt water in the spring of their second year.

Table 6. Upper Columbia River Chinook salmon populations and selected measures of population viability

Population	Mean Number of Spawners (Range)[a]	Percent Hatchery Contribution[b]	Current Short-term trend (Previous)[c]
Methow River	680 (79-9,904)	59	+2.0 (-15.3)
Methow mainstem	161 redds (17-2,864)	59	+6.5
Twisp River	58 redds (10-369)	54	-9.8 (-27.4)
Chewuch River	58 redds (6-1,105)	41	-2.9 (-28.1)
Lost/Early Winter creeks	12 (3-164)	54	-14.1 (-23.2[d])
Entiat River	111 (53-444)	42	-1.2 (-19.4)
Wenatchee River	470 (119-4,446)	42	-1.5 (-37.4)
Chiwawa River	109 redds (34-1,046)	47	-0.7 (-29.3)
Nason Creek	54 redds (8-374)	39	-1.5 (-26.0)
Upper Wenatchee River	8 redds (0-215)	66	-8.9
White River	9 redds (1-104)	8	-6.6 (-35.9)
Little Wenatchee River	11 redds (3-74)	21	-25.8 (-25.8)

[a]5-year geometric mean number of spawners unless otherwise noted; Includes hatchery fish. Range denoted in parentheses. Means calculated from years 1997 to 2001, except Lost/Early Winter creeks did not include 1998 as no data was available. Data reported in Good et al. 2005.
[b]Percent hatchery-origin from 1987-1996, and reported in Good et al. 2005.
[c]Current trend – percent/year – from years 1997 to 2001. Previous trend, noted in parentheses, from 1987-1996. From Good et al. 2005.
[d]Lost River data only.

Status and Trends

NMFS listed Upper Columbia River spring-run Chinook salmon as endangered on March 24, 1999 (64 FR 14308), and reaffirmed their status as endangered on June 28, 2005 (70 FR 37160), because they had been reduced to small populations in three watersheds. Based on redd count data series, spawning escapements for the Wenatchee, Entiat, and Methow rivers have declined an average of 5.6%, 4.8%, and 6.3% per year, respectively, since 1958. In the most recent 5-year geometric mean (1997 to 2001), spawning escapement for naturally produced fish was 273 for the Wenatchee population, 65 for the Entiat population, and 282 for the Methow population, only 8% to 15% of the minimum abundance thresholds, although escapement increased substantially in 2000 and 2001 in all three river systems. Based on 1980-2004 returns, the average annual growth rate for this ESU is estimated as 0.93 (meaning the population is not replacing itself; Fisher and Hinrichsen 2006). Assuming that population growth rates were to continue at 1980 to 2004 levels, Upper Columbia River spring-run Chinook salmon populations are projected to have very high probabilities of decline within 50 years. Population viability analyses for this species (using the Dennis Model) suggest that these Chinook salmon face a significant risk of extinction: a 75 to 100% probability of extinction within 100 years (given return rates for 1980 to present).

Hatchery influence and genetic diversity are significant issues for the continued survival of Upper Columbia River Chinook salmon. This is a result of reduced genetic diversity from homogenization of populations that occurred under the Grand Coulee Fish Maintenance Project from 1939 to 1943. Stray hatchery fish and a high proportion of hatchery fish during spawning have contributed to the high genetic diversity risk.

Critical Habitat

NMFS designated critical habitat for Upper Columbia River spring-run Chinook salmon on September 2, 2005 (70 FR 52630). The designation includes all Columbia River estuaries and river reaches upstream to Chief Joseph Dam and several tributary subbasins. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The Upper Columbia River spring-run Chinook salmon ESU has 31 watersheds within its range. Five watersheds received a medium rating and 26 received a high rating of conservation value to the ESU. The Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Factors contributing to the downward trends in this ESU include mainstem Columbia River hydropower system mortality, tributary riparian degradation and loss of in-river wood, altered tributary floodplain and channel morphology, reduced tributary stream flow and impaired passage, and harvest impacts.

Puget Sound Chinook Salmon

Distribution and Description of the Listed Species

The Puget Sound Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from rivers and streams flowing into Puget Sound including the Straits of Juan De Fuca from the Elwha River, eastward, including rivers and streams flowing into Hood Canal, South Sound, North Sound and the Strait of Georgia in Washington. Twenty-six artificial propagation programs are part of the ESU: the Kendal Creek Hatchery, Marblemount Hatchery (fall, spring yearlings, spring sub-yearlings, and summer run), Harvey Creek Hatchery, Whitehorse Springs Pond, Wallace River Hatchery (yearlings and sub-yearlings), Tulalip Bay, Issaquah Hatchery, Soos Creek Hatchery, Icy Creek Hatchery, Keta Creek Hatchery, White River Hatchery, White Acclimation Pond, Hupp Springs hatchery, Voights Creek Hatchery, Diru Creek, Clear Creek, Kalama Creek, George Adams Hatchery, Rick's Pond Hatchery, Hamma Hamma Hatchery, Dungeness/Hurd Creek Hatchery, and Elwha Channel Hatchery Chinook salmon hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

The Puget Sound ESU is comprised of 31 historical populations, of which 22 or more are believed to be extant and nine are considered extinct. Table 7 identifies the current populations within the Puget Sound Chinook salmon ESU for which there are data, and their recent abundance and long-term trends.

Chinook salmon in this area generally have an "ocean-type" life history. Puget Sound populations include both early-returning and late-returning Chinook salmon spawners described by Healey (1991). However, within these generalized behavioral forms, significant variation occurs in residence time in fresh water and estuarine environments. For example, Hayman et al. (1996) described three juvenile Chinook salmon life histories with varying residency times in the Skagit River system in northern Puget Sound. Chinook salmon utilize nearshore Puget Sound habitats year-round, although they can be far from their natal river systems (Brennan et al. 2004).

Table 7. Puget Sound Chinook salmon populations and selected measures of population viability

Population	Historical Abundance[a]	Mean Number of Spawners (Natural-origin)[b]	Percent Hatchery Contribution (Range)[c]	λ (+/- SE)[d]
Nooksack-North Fork	26,000	1,538 (125)	91 (88-95)	0.75 (0.07)
Nooksack-South Fork	13,000	338 (197)	40 (24-55)	0.94 (0.05)
Lower Skagit	22,000	2,527 (2,519)	0.2 (0-0.7)	1.05 (0.09)
Upper Skagit	35,000	9,489 (9,281)	2 (2-3)	1.05 (0.06)
Upper Cascade	1,700	274 (274)	0.3	1.06 (0.05)
Lower Sauk	7,800	601 (601)	0	1.01 (0.12)
Upper Sauk	4,200	324 (324)	0	0.96 (0.06)
Suiattle	830	365 (365)	0	0.99 (0.06)
Stillaguamish-North Fork	24,000	1,154 (671)	40 (13-52)	0.92 (0.04)
Stillaguamish-South Fork	20,000	270		0.99 (0.02)*
Skykomish	51,000	4,262 (2,392)	40 (11-66)	0.87 (0.03)
Snoqualmie	33,000	2,067(1,700)	16 (5-72)	1.00 (0.04)
North Lake Washington		331		1.07 (0.07)*
Cedar		327		0.99 (0.07)*
Green		8,884 (1,099)	83 (35-100)	0.67 (0.06)*
White		844		1.16 (0.06)*
Puyallup	33,000	1,653		0.95 (0.06)*
Nisqually	18,000	1,195		1.04 (0.07)*
Skokomish		1,392		1.04 (0.04)*
Dosewallips	4,700	48		1.17 (0.10)*

Duckabush		43		
Hamma Hamma		196		
Mid Hood Canal		311		
Dungeness	8,100	222		1.09 (0.11)*
Elwha		688		0.95 (0.11)*

[a]Estimated total historical abundance for this ESU was about 700,000 fish, but is not meant to reflect a summation of individual river historic estimates. Individual river estimates of historical abundance are based on an EDT analysis as reported in Good et al. 2005.
[b]5-year geometric mean number of spawners (hatchery plus natural) for years 1998-2002. Geometric mean of natural origin spawners noted in parentheses. From Good et al. 2005.
[c]Percent hatchery-origin from 1997-2001. Estimates are from the TRT database and reported in Good et al. 2005.
[d]Short-term median population growth rate estimates assume that the reproductive success of naturally spawning hatchery fish is equivalent to that of natural origin fish. Except estimates noted * where an estimate of the fraction of hatchery fish is not available then λ represents hatchery fish + natural-origin spawners. Data years used for calculation 1990-2002 (Good et al. 2005).

Status and Trends

NMFS listed Puget Sound Chinook salmon as threatened in 1999 (64 FR 14308); that status was reaffirmed on June 28, 2005 (70 FR 37160). This ESU has lost 15 spawning aggregations (nine from the early-run type) that were either independent historical populations or major components of the remaining 22 existing independent historical populations identified (Good et al. 2005). The disproportionate loss of early-run life history diversity represents a significant loss of the evolutionary legacy of the historical ESU.

Data reported by Good et al. (2005) indicate that long term trends in abundance for this ESU are split with about half of the populations declining, and the other half increasing. In contrast, the short-term trend for four populations is declining. The overall long-term trend in abundance indicates that, on average, populations are just replacing themselves. Estimates of the short-term median population growth rate (λ) (data years 1990-2002) indicate an even split between populations that are growing and those that are declining, although estimates would be lower for several populations if the fraction of naturally spawning hatchery fish were available for all populations within the ESU. For available data, when λ is calculated assuming that hatchery fish have the equivalent success of natural spawners then the largest estimated decline occurs in the Green River. Populations with the largest positive short and long-term trends include the White River and the North Fork Nooksack River (Good et al. 2005). Lambda for the Skagit River, which produces the most Chinook salmon in this ESU, has increased slightly. Overall, the recent analysis by Good et al. (2005) illustrated that there has not be much change in this ESU since NMFS' first status review (Busby et al. 1996). Individual populations have improved, while others have declined. However, the lack of information on the fraction of naturally spawning, hatchery-origin fish for 10 of the 22 populations within this ESU limits our understanding of the trends in naturally spawning fish for a large portion of the ESU.

The estimated total run size of Chinook salmon in Puget Sound in the early 1990s was 240,000 fish, representing a loss of nearly 450,000 fish from historic numbers. During a recent 5-year period, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over 9,489 fish. Most populations had natural spawners numbering in the hundreds (median recent natural escapement is 766), and of the six populations with greater than 1,000 natural spawners, only two have a low fraction of hatchery fish. The populations with the greatest estimated component of hatchery fish tend to be in mid- to southern Puget Sound, Hood Canal, and the Strait of Juan de Fuca regions. Estimates of the historical equilibrium abundance, based on pre-European settlement habitat conditions, range from 1,700 to 51,000 potential Puget Sound Chinook salmon spawners per population. The historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for Puget Sound Chinook salmon on September 2, 2005 (70 FR 52630). The specific geographic area includes portions of the Nooksack River, Skagit River, Sauk River, Stillaguamish River, Skykomish River, Snoqualmie River, Lake Washington, Green River, Puyallup River, White River, Nisqually River, Hamma Hamma River and other Hood Canal watersheds, the Dungeness/Elwha Watersheds, and nearshore marine areas of the Strait of Georgia, Puget Sound, Hood Canal and the Strait of Juan de Fuca. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high water line is not defined the lateral extent is defined as the bankfull elevation.

The designation for this ESU includes sites necessary to support one or more Chinook salmon life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Specific primary constituent elements include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 49 subbasins (5th field Hydrological Units) reviewed in NMFS' assessment of critical habitat for the Puget Sound ESUs, nine subbasins were rated as having a medium conservation value, 12 were rated as low, and the remaining subbasins (40), where the bulk of Federal lands occur for this ESU, were rated as having a high conservation value to Puget Sound Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes (such as diking, revetments, loss of secondary channels in floodplains, widespread blockages of streams, and changes in peak flows), degraded freshwater and marine habitat affected by agricultural activities and urbanization, and upper river tributaries widely affected by poor forest practices. Changes in habitat quantity, availability, diversity, flow, temperature, sediment load, and channel stability are common limiting factors in areas of critical habitat.

Sacramento River Winter-Run Chinook Salmon

Distribution and Description of the Listed Species

The Sacramento River winter-run Chinook salmon ESU includes all naturally spawned populations of winter-run Chinook salmon in the Sacramento River and its tributaries in California. Two artificial propagation programs are included in this ESU: winter-run Chinook salmon from the Livingston Stone National Fish Hatchery, and winter-run Chinook salmon in a captive broodstock program maintained at the Livingston Stone National Fish Hatchery and the University of California Bodega Marine Laboratory. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

This ESU consists of a single spawning population that enters the Sacramento River and its tributaries in California from November to June and spawns from late April to mid-August, with a peak from May to June (Table 8). Sacramento River winter-run Chinook salmon historically occupied cold, headwater streams, such as the upper reaches of the Little Sacramento, McCloud, and lower Pit Rivers. Young winter-run Chinook salmon venture to sea in November and December, after only four to seven months in fresh water (Groot et al. 1991).

Table 8. Sacramento River winter-run Chinook salmon abundance and selected measures of population viability

Population	Historical Abundance[a]	Mean number of Spawners (Range)[b]	Percent Hatchery Contribution	Population growth rate (λ)[c]
Sacramento River winter-run	200,000	2,191 (364-65,683)	<10	0.97 (0.87, 1.09)

[a]Historical abundance for the total ESU based on commercial fishery landings in the 1870s (Fisher 1994). Individual river estimates of historical abundance not provided.
[b]Recent geometric mean number of spawners from Good et al. 2005.
[c]Lambda value reported by Good et al. 2005. The 90% confidence intervals are noted in parentheses.

Status and Trends

NMFS listed Sacramento River winter-run Chinook salmon as endangered on January 4, 1994 (59 FR 440), and reaffirmed their status as endangered on June 28, 2005 (70 FR 37160), because dams restrict access to a small fraction of their historic spawning habitat and the habitat remaining to them is degraded. Sacramento River winter-run Chinook salmon consist of a single self-sustaining population which is entirely dependent upon the provision of suitably cool water from Shasta Reservoir during periods of spawning, incubation and rearing.

Construction of Shasta Dams in the 1940s eliminated access to historic spawning habitat for winter-run Chinook salmon in the basin. Winter-run Chinook salmon were not expected to survive this habitat alteration (Moffett 1949). However, cold water releases from Shasta Dam have created conditions suitable for winter Chinook salmon for roughly 60 miles downstream from the dam. As a result the ESU has been reduced to a single spawning population confined to the mainstem Sacramento River below Keswick Dam, although some adult winter-run Chinook salmon were recently observed in Battle Creek, a tributary to the upper Sacramento River.

Quantitative estimates of run-size are not available for the period before 1996, the completion of Red Bluff Diversion Dam. However, winter-runs may have been as large as 200,000 fish based upon commercial fishery records from the 1870s (Fisher 1994). The California Department of Fish and Game estimated spawning escapement of Sacramento River winter-run Chinook salmon at 61,300 (60,000 in the mainstem, 1,000 in Battle Creek, and 300 in Mill Creek) in the early 1960s. During the first 3 years of operation of the county facility at the Red Bluff Diversion Dam (1967 to 1969), the spawning run of winter-run Chinook salmon averaged 86,500 fish. From 1967 through the mid-1990s, the population declined at an average rate of 18% per year, or roughly 50% per generation. The population reached critically low levels during the drought of 1987 to 1992; the 3-year average run size for the period of 1989 to 1991 was 388 fish. Based on the Red Bluff Diversion Dam counts, the population has been growing rapidly since the 1990s. Mean run size from 1995-2000 has been 2,191, but have ranged from 364 to 65,683 (Good et al.

2005). Most recent estimates indicate that the short term trend is 0.26, while the population growth rate is still less than 1 (Table 8). The draft recovery goal for the ESU is an average of 10,000 female spawners per year and a population growth rate >1.0, calculated over 13 years of data (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for Sacramento River winter-run Chinook salmon on June 16, 1993 (58 FR 33212). The following areas consisting of the water, waterway bottom, and adjacent riparian zones: the Sacramento River from Keswick Dam, Shasta County (river mile 302) to Chipps Island (river mile 0) at the westward margin of the Sacramento-San Joaquin Delta, and other specified estuarine waters. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Factors contributing to the downward trends in this ESU include reduced access to spawning/rearing habitat, possible loss of genetic integrity through population bottlenecks, inadequately screened diversions, predation at artificial structures and by nonnative species, pollution from Iron Mountain Mine and other sources, adverse flow conditions, high summer water temperatures, unsustainable harvest rates, passage problems at various structures, and vulnerability to drought (Good et al. 2005).

Snake River Fall-Run Chinook Salmon

Distribution and Description of the Listed Species

The Snake River fall-run Chinook salmon ESU includes all naturally spawned populations of fall-run Chinook salmon in the mainstem Snake River below Hells Canyon Dam, and in the Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River subbasins. Four artificial propagation programs are part of this ESU: The Lyons Ferry Hatchery, Fall Chinook salmon Acclimation Ponds Program, Nez Perce Tribal Hatchery, and Oxbow Hatchery fall-run hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Historically, the primary fall-run Chinook salmon spawning areas occurred on the upper mainstem Snake River (Connor et al. 2005). A series of Snake River dams blocked access to the upper reaches, which significantly reduced spawning and rearing habitat. Consequently, salmon now reside in waters that are generally cooler than pre-dam habitats. Currently, natural spawning occurs at the upper end of Lower Granite Reservoir to Hells Canyon Dam, the lower reaches of the Imnaha, Grande Ronde, Clearwater, and Tucannon rivers, and small mainstem sections in the tailraces of the lower Snake River hydroelectric dams.

Adult Snake River fall-run Chinook salmon enter the Columbia River in July and August, and spawning occurs from October through November. Juveniles emerge from the gravels in March and April of the following year, moving downstream from natal spawning and early rearing areas from June through early fall. Prior to dam construction, fall Chinook salmon were primarily ocean-type (migrated downstream and reared in the mainstem Snake River during their first year). However, today both an ocean-type and reservoir-type occur (Connor et al. 2005). The reservoir-type juveniles overwinter in pools created by dams before migrating to sea; this response is likely due to early development in cooler temperatures which prevents rapid growth. Phenotypic characteristics have shifted in apparent response to environmental changes from hydroelectric dams (Connor et al. 2005). Migration downstream appears to be influenced by flow velocity within both river and reservoir systems (Tiffan et al. 2009).

Status and Trends

NMFS originally listed Snake River fall-run Chinook salmon as endangered in 1992 (57 FR 14653) but reclassified their status as threatened on June 28, 2005 (70 FR 37160). Estimated annual returns for the period 1938 to 1949 was 72,000 fish, and by the 1950s, numbers had declined to an annual average of 29,000 fish (Bjornn and Horner 1980). Numbers of Snake River fall-run Chinook salmon continued to decline during the 1960s and 1970s as approximately 80% of their historic habitat was eliminated or severely degraded by the construction of the Hells Canyon complex (1958 to 1967) and the lower Snake River dams (1961 to 1975). Counts of natural-origin adult Snake River fall-run Chinook salmon at Lower Granite Dam were 1,000 fish in 1975, and ranged from 78 to 905 fish (with an average of 489 fish) over the ensuing 25-year period (Good et al. 2005). Numbers of natural-origin Snake River fall-run Chinook salmon have increased over the last few years, with estimates at Lower Granite Dam of 2,652 fish in 2001, 2,095 fish in 2002, and 3,895 fish in 2003.

Snake River fall-run Chinook salmon have exhibited an upward trend in returns over Lower Granite Dam since the mid 1990s. Returns classified as natural-origin spawners exceeded 2,600 fish in 2001, compared to a 1997 to 2001 geometric mean natural-origin count of 871 (35% of the proposed delisting abundance criteria of 2,500 natural spawners averaged over 8 years). Both the long- and short-term trends in natural returns are positive. Harvest impacts on Snake River fall Chinook salmon declined after listing and have remained relatively constant in recent years. Mainstem conditions for subyearling Chinook migrants from the Snake River have generally improved since the early 1990s. The hatchery component, derived from outside the basin, has decreased as a percentage of the run at Lower Granite Dam from the 1998/99 status reviews (5-year average of 26.2%) to 2001 (8%). This reflects an increase in the Lyons Ferry component, systematic removal of marked hatchery fish at the Lower Granite trap, and modifications to the Umatilla supplementation program to increase homing of fall Chinook salmon release groups. Hatcheries stocking fish to the Snake River fall run produce genetic affects in the population due to three major components: natural-origin fish (which may be progeny of hatchery fish), returns of Snake River fish from the Lyons Ferry Hatchery program, and strays from hatchery programs outside the Snake River.

Critical Habitat

NMFS designated critical habitat for Snake River fall-run Chinook salmon on December 28, 1993 (58 FR 68543). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to Hells Canyon Dam. Critical habitat also includes several river reaches presently or historically accessible to Snake River fall-run Chinook salmon. Limiting factors identified for Snake River fall-run Chinook salmon include: mainstem lower Snake and Columbia hydrosystem mortality, degraded water quality, reduced spawning and rearing habitat due to mainstem lower Snake River hydropower system, harvest impacts, impaired stream flows, barriers to fish passage in tributaries, excessive sediment, and altered floodplain and channel morphology (NMFS 2005a).

Snake River Spring/Summer-Run Chinook Salmon

Distribution and Description of the Listed Species

The Snake River spring/summer-run Chinook salmon ESU includes all naturally spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins. Fifteen artificial propagation programs are part of the ESU: The Tucannon River conventional Hatchery, Tucannon River Captive Broodstock Program, Lostine River, Catherine Creek, Lookingglass Hatchery Reintroduction Program (Catherine Creek), Upper Grande Ronde, Imnaha River, Big Sheep Creek, McCall Hatchery, Johnson Creek Artificial Propagation Enhancement, Lemhi River Captive Rearing Experiment, Pahsimeroi Hatchery, East Fork Captive Rearing Experiment, West Fork Yankee Fork Captive Rearing Experiment, and the Sawtooth Hatchery spring/summer-run Chinook salmon hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU. The Interior Columbia Basin Technical Recovery Team has identified 32 populations in five major population groups (Upper Salmon River, South Fork Salmon River, Middle Fork Salmon River, Grande Ronde/Imnaha, Lower Snake Mainstem Tributaries) for this species. Historic populations above Hells Canyon Dam are considered extinct (ICBTRT 2003). Table 9 identifies extant populations within the Snake River spring/summer Chinook salmon ESU, their abundances, and the relative contribution of hatchery fish.

Snake River spring/summer-run Chinook salmon have a stream-type life history. Spawning occurs in late summer and early fall and eggs incubate over the following winter and hatch in late winter and early spring of the following year. Juveniles mature in the river for one year before migrating to the ocean in the spring of their second year. Larger outmigrants have a higher survival rate during outmigration (Zabel and Williams 2002; Zabel and Achord 2004). Depending on tributary and the specific habitat conditions, juveniles may migrate widely from natal reaches into alternative summer-rearing or overwintering areas. Spawners return to spawn primarily as 4- and 5-year-olds after 2 to 3 years in the ocean. A small fraction return as 3-year-old “jacks” (although sexually mature upon return, these fish are smaller in body and 1-2 years younger than most males on the spawning ground).

Table 9. Snake River spring/summer Chinook salmon populations and selected measures of population viability

Current Populations	Mean Number of Spawners (Range)[a]	Percent Hatchery Contribution[b]	Short-term Trend (Previous)[c]
Tucannon River	303 (128-1,012)	76	-4.1 (-11.0)
Wenaha River	225 (67-586)	64	-9.4 (-23.6)
Wallowa River	0.57 redds (0-29)	5	11.5
Lostine River	34 redds (9-131)	5	12.7
Minam River	180 (96-573)	5	3.3 (-14.5)
Catherine Creek	50 (13-262)	56	-25.1 (-22.5)
Upper Grande Ronde River	46 (3-336)	58	-9.4
South Fork Salmon River	496 redds (277-679)	9	1.1 (-13.6)
Secesh River	144 redds (38-444)	4	9.8
Johnson Creek	131 redds (49-444)		-1.5
Big Creek spring run	53 (21-296)		5.4 (-34.2)
Big Creek summer run	5 redds (2-58)		1.7 (-27.9)
Loon Creek	27 redds (6-255)		12.2
Marsh Creek	53 (0-164)		-4.0
Bear Valley/Elk Creek	266 (72-712)		6.2
North Fork Salmon River	5.6 redds (2-19)		
Lemhi River	72 redds (35-216)		12.8 (-27.4)
Pahsimeroi River	161 (72-1,097)		12.8
East Fork Salmon spring run	0.27 rpm (0.2-1.41)		-5.7
East Fork Salmon summer run	1.22 rpm 0.35-5.32)		0.9 (-32.9)
Yankee Fork spring run	0 rpm		-6.3
Yankee Fork summer run	2.9 redds (1-18)		4.1
Valley Creek spring run	7.4 redds (2-28)		14.9 (-25.9)
Valley Creek summer run	2.14 rpm(0.71-9.29)		5.8 (-29.3
Upper Salmon spring run	69 redds (25-357)		5.3
Upper Salmon summer run	0.24 rpm (0.07-0.58)		-3.3
Alturas Lake Creek	2.7 redds (0-18)		10.2
Imnaha River	564 redds (194-3,041)	62	12.8(-24.1)
Big Sheep Creek	0.25 redds (0-1)	97	0.8
Lick Creek	1.4 redds (0-29)	59	11.7

[a]All data reported in Good et al. 2005. Except where noted values represent the recent geometric mean number of spawners. RPM =redds per mile.
[bc]Reported in Good et al. 2005.
[c]For details on data series used in calculating the population's short term trend see Good et al. 2005.

Status and Trends

NMFS originally listed Snake River spring/summer-run Chinook salmon as threatened on April 22, 1992 (57 FR 14653), and reaffirmed their status as threatened on June 28, 2005 (70 FR 37160). Although direct estimates of historical annual Snake River spring/summer Chinook salmon returns are not available, returns may have declined by as much as 97% between the late 1800s and 2000. According to Matthews and Waples (1991), total annual Snake River spring/summer Chinook salmon production may have exceeded 1.5 million adult fish in the late 1800s. Total (natural plus hatchery origin) returns fell to roughly 100,000 spawners by the late 1960s and were below 10,000 by 1980 (Fulton 1968). Between 1981 and 2000, total returns fluctuated between extremes of 1,800 and 44,000 fish. The 2001 and 2002 total returns increased to over 185,000 and 97,184 adults, respectively. The 1997 to 2001 geometric mean total return for the summer run component at Lower Granite Dam was slightly more than 6,000 fish, compared to the geometric mean of 3,076 fish for the years 1987 to 1996. The 2002 to 2006 geometric mean of the combined Chinook salmon runs at Lower Granite Dam was over 18,000 fish. However, it is important to note that over 80% of the 2001 return and over 60% of the 2002 return originated in hatcheries (Good et al. 2005). Good et al. (2005) reported that risks to individual populations within the ESU may be greater than the extinction risk for the entire ESU due to low levels of annual abundance and the extensive production areas within the Snake River basin. Although the average abundance in the most recent decade is more abundant than the previous decade, there is no obvious long-term trend.

Critical Habitat

NMFS designated critical habitat for Snake River spring/summer-run Chinook salmon on October 25, 1999 (64 FR 57399). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to Hells Canyon Dam; the Palouse River from its confluence with the Snake River upstream to Palouse Falls, the Clearwater River from its confluence with the Snake River upstream to its confluence with Lolo Creek; the North Fork Clearwater River from its confluence with the Clearwater river upstream to Dworshak Dam. Critical habitat also includes several river reaches presently or historically accessible to Snake River spring/summer Chinook salmon. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Limiting factors identified for this species include hydrosystem mortality, reduced stream flow, altered channel morphology and floodplain, excessive fine sediment, and degraded water quality (NMFS 2006c).

Upper Willamette River Chinook Salmon

Distribution and Description of the Listed Species

The Upper Willamette River Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Clackamas River and in the Willamette River, and its tributaries, above Willamette Falls, Oregon. Seven artificial propagation programs are part of the ESU: The McKenzie River Hatchery, Marion Forks/North Fork Santiam River, South Santiam Hatchery in the South Fork Santiam River, South Santiam Hatchery in the Calapooia River, South Santiam Hatchery in the Mollala River, Willamette Hatchery, and Clackamas hatchery spring-run Chinook salmon hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Upper Willamette River Chinook salmon occupy the Willamette River and its tributaries. All spring-run Chinook salmon in the ESU, except those entering the Clackamas River, must pass

Willamette Falls. In the past, this ESU included sizable numbers of spawning salmon in the Santiam River, the middle fork of the Willamette River, and the McKenzie River, as well as smaller numbers in the Molalla River, Calapooia River, and Albiqua Creek. Historically, access above Willamette Falls was restricted to the spring when flows were high. In autumn, low flows prevented fish from ascending past the falls. The Upper Willamette spring-run Chinook salmon are one of the most genetically distinct Chinook salmon groups in the Columbia River Basin. Upper Willamette River Chinook salmon enter the Columbia River and estuary earlier than other spring Chinook salmon ESUs (Meyers et al. 1998). Fall-run Chinook salmon spawn in the Upper Willamette but are not considered part of the ESU because they are not native.

Status and Trends

NMFS originally listed Upper Willamette River Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their status as threatened on June 28, 2005 (70 FR 37160). The total abundance of adult spring-run Chinook salmon (hatchery-origin plus natural-origin fish) passing Willamette Falls has remained relatively steady over the past 50 years (ranging from approximately 20,000 to 70,000 fish), but it is an order of magnitude below the peak abundance levels observed in the 1920s (approximately 300,000 adults). Until recent years, interpretation of abundance levels has been confounded by a high but uncertain fraction of hatchery-produced fish. Although the number of adult spring-run Chinook salmon crossing Willamette Falls is in the same range (about 20,000 to 70,000 adults) it has been for the last 50 years, a large fraction of these are hatchery produced. Estimates of the percentage of hatchery fish range according to tributary, several of which exceed 70 percent (Good et al. 2005). The Calapooia River is estimated to contain 100 percent hatchery fish. Insufficient information on hatchery production in the past prevents a meaningful analysis of the population trend; therefore no formal trend analysis is available.

Most natural spring Chinook salmon populations of the Upper Willamette River are likely extirpated or nearly so, with only one remaining naturally reproducing population identified in this ESU: the spring Chinook salmon in the McKenzie River. Unfortunately, recently short-term declines in abundance suggest that this population may not be self-sustaining (Myers et al. 1998; Good et al. 2005). Abundance in this population has been relatively low (low thousands) with a substantial number of these fish being of hatchery origin. The population increased substantially from 2000 to 2003, probably due to increased survival in the ocean. Future survival rates in the ocean are unpredictable, and the likelihood of long-term sustainability for this population has not been determined. Of concern is that a majority of the spawning habitat and approximately 30 to 40% of total historical habitat are no longer accessible because of dams (Good et al. 2005). Individuals from the ESU migrate far north and are caught incidentally in ocean fisheries, particularly off southeast Alaska and northern Canada, and in the mainstem Columbia and Willamette rivers during spring.

Critical Habitat

NMFS designated critical habitat for Upper Willamette River Chinook salmon on September 2, 2005 (70 FR 52630). Critical habitat for upper Willamette River Chinook salmon includes defined areas within subbasins of the middle fork Willamette River, upper Willamette River, McKenzie River, Santiam River, Crabtree Creek, Molalla River, and Clackamas River. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning and rearing sites, freshwater migration corridors. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 65 subbasins reviewed in NMFS' assessment of critical habitat for the Upper Willamette River Chinook salmon ESU, 19 subbasins were rated as having a medium conservation value, 19 were rated as low, and the 27 remaining subbasins were rated as having a high conservation value to Upper Willamette River Chinook salmon. Federal lands were generally rated as having high conservation value to the species' spawning and rearing. Factors contributing to the downward trends in this ESU include reduced access to spawning/rearing habitat in tributaries, hatchery impacts, altered water quality and temperature in tributaries, altered stream flow in tributaries, and lost or degraded floodplain connectivity and lowland stream habitat.

Chum Salmon

Description of the Species

Chum salmon are more widely distributed than other salmon, and may have at one time made up nearly 50% of the Pacific salmon biomass in the Pacific Ocean (Salo 1991). Historically, chum salmon were distributed throughout the coastal regions of western Canada and the United States, as far south as Monterey Bay, California, to the Arctic coast and east to the Mackenzie River, in the Beaufort Sea. They also ranged in Asia from Korea to the Arctic coast of the Soviet Union and west to the Lena River. Presently, major spawning populations on the west coast of the United States are found only as far south as Tillamook Bay on the northern Oregon coast. In this section of our Opinion, we discuss the distribution, status, and critical habitats of the two listed species of threatened chum salmon separately; however, because chum salmon in the wild are virtually indistinguishable between listed ESUs, and are the same biological species sharing the same generalized life history, we begin this section describing those characteristics common across ESUs.

Chum salmon exhibit obligatory anadromy (there are no recorded landlocked or naturalized freshwater populations), and like Chinook salmon, chum salmon are semelparous so they die after one spawning event. Their general life cycle spans fresh and marine waters, although chum salmon are more marine oriented than the other Pacific salmon, in that they spend very little time rearing in fresh water. Chum salmon spend 2 to 5 years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175° E longitude. North American chum salmon migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska, although some data suggest that Puget Sound chum, including Hood Canal summer run chum, may not make extended migrations into northern British Columbian and Alaskan waters, but instead may travel directly offshore into the north

Pacific Ocean.

Spawning migrations generally occur in the summer and fall; the precise spawn timing and migration varies across populations. Stream flows and water temperatures can influence stream entry. Sexual differences in the timing of returns to spawning grounds are apparent with males generally arriving early and females later in the run. Once on the spawning grounds mate competition is intense with males competing to fertilize eggs and females competing for optimal nest site selection. Size and age at maturity is partially under genetic control, but can be influenced by environment and migration behavior. Generally, spawning runs consist of fish between 2 and 5 years of age, and like Chinook salmon, chum females will build large redds that consist of four or five egg pockets laid in succession. Chum salmon fecundity is highly variable, and is correlated with body size and region (latitudinal trends are evident with northern population having lower absolute and relative fecundities; Salo 1991).

The time necessary for egg incubation until emergence of alevins in fresh water varies among basins and among years within a basin, and is closely correlated to water temperatures such that low temperatures prolong incubation. Egg and alevin survival, and the fitness of emerging fry are affected by sediment loading, intergravel water flow and dissolved oxygen levels, gravel composition, spawning time and density, and water temperatures. Once they emerge from their gravel nests, chum salmon fry outmigrate to seawater almost immediately (Salo 1991). This ocean-type migratory behavior contrasts with the stream-type behavior of other species in the genus Oncorhynchus (e.g., coastal cutthroat trout, steelhead, coho salmon, and most types of Chinook and sockeye salmon, exception pink salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. Because of their small size chum salmon will form loosely aggregated schools, presumably to reduce predation by swamping predators (Miller and Brannon 1982; Pitcher 1986).

Generally, chum fry emigrate to estuaries between March through May where they forage on epibenthic and neritic food resources. The timing of juvenile entry into sea water is commonly correlated with nearshore warming and associated plankton blooms (Groot et al. 1991). As food resources decline and the fish grow, they move further out to forage on pelagic and nektonic organisms (Simenstad and Salo 1982; Salo 1991). Migratory studies indicate that chum salmon in their first year of life will typically maintain a coastal migratory pattern although the pattern is variable as they mature at sea. At sea chum salmon feed on pteropods, euphausiids, amphipods, fish and squid larvae (Salo 1991).

Threats

Natural Threats. Chum salmon are exposed to high rates of natural predation each stage of their life stage, and in particular during migration. Mortality at emergence or prior to emergence is significant because eggs develop in the interstitial spaces in the stream gravel, and storm surges that redeposit gravels and wash out eggs or introduce silt to the interstitial spaces can reduce egg survival. Other factors that reduce egg survival and larvae development include low dissolved oxygen, poor percolation, and extreme cold or warm temperatures.

Anthropogenic Threats. Chum salmon, like the other listed salmon, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the fresh water, estuarine, and coastal ecosystems throughout the Pacific Northwest. These threats for are summarized in detail under Chinook salmon.

Columbia River Chum Salmon

Distribution and Description of the Listed Species

The Columbia River chum ESU includes all naturally spawned populations of chum salmon in the Columbia River and its tributaries in Washington and Oregon. Three artificial propagation programs are part of the ESU: The Chinook River (Sea Resources Hatchery), Grays River, and Washougal River/Duncan Creek chum hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Most of the chum within this ESU return to northern tributaries of the Columbia River (in Washington State), primarily the Grays River, in areas immediately below Bonneville Dam, and in smaller numbers under the I-205 bridge near Vancouver. Chum populations that formerly occupied tributaries on the south bank of the Columbia (in Oregon) are considered extirpated or nearly so. Observers have documented spawning over multiple years in the mainstem Columbia River, near McCord Creek and Multnomah Falls in Oregon, although the number of spawners in these areas are generally quite low (McElhany et al. 2007).

Chum salmon return to the Columbia River in late fall (mid-October to December).

Table 10. Columbia River chum salmon populations and selected measures of population viability

Current Populations	Historical Abundance[a]	Recent Spawner Abundance	Short-Term Median Growth Rate (λ)[c]
Youngs Bay			
Gray's River	7,511	331/704[b]	1.043 (0.957-1.137)
Big Creek			
Elochoman River			
Clatskanie River			
Mill, Abernathy, and Germany Creeks			
Scappoose Creek			
Cowlitz River	141,582		
Kalama River	9,953		
Lewis River	89,671		
Salmon Creek			
Clackamus River			
Sandy River			
Washougal River	15,140		

Lower gorge tributaries	>3,141	425[b]	0.984 (0.883-1.096)
Upper gorge tributaries	>8,912		

[a]Estimated total historical abundance for this ESU was about 283,421 fish, but is not meant to reflect a summation of individual river historic estimates. Individual river estimates of historical abundance are based on an EDT analysis using equilibrium abundance under historical conditions. All data are reported in Good et al. 2005.

[b]Two different time series estimate are available but based on overlapping years. The first estimate is based on 1996-2000 data, while the second is based on 1996-2000 data.

[c]The λ calculation is an estimate of what the natural growth rate would have been after accounting for hatchery-origin spawners. Two different estimates are available for the Grays River population; the Rawlings estimate (depicted in the table above) is believed to be more accurate. Other estimates, long- and short-term trends, suggest the population is declining (see Good et al. 2005).

Status and Trends

NMFS listed Columbia River chum salmon as threatened on March 25, 1999, and reaffirmed their threatened status on June 28, 2005 (71 FR 37160). Chum salmon in the Columbia River once numbered in the hundreds of thousands of adults and were reported in almost every river in the Lower Columbia River basin, but by the 1950s most runs disappeared (Rich 1942; Marr 1943; Fulton 1970). The total number of chum salmon returning to the Columbia River in the last 50 years has averaged a few thousand per year, with returns limited to a very restricted portion of the historical range. Significant spawning occurs in only two of the 16 historical populations, meaning that 88% of the historical populations are extirpated, or nearly so. The two remaining populations are the Grays River and the lower Columbia Gorge tributaries (Good et al. 2005). Both long- and short-term trends for Grays River abundance are negative, but the current trend in abundance for the lower Columbia Gorge tributaries is slightly positive. Chum salmon appear to be extirpated from the Oregon portion of this ESU. In 2000, ODFW conducted surveys to determine the abundance and distribution of chum salmon in the Columbia River, and out of 30 sites surveyed, only one chum salmon was observed.

Few Columbia River chum salmon have been observed in tributaries between The Dalles and Bonneville dams. Surveys of the White Salmon River in 2002 found one male and one female carcass, with no evidence of spawning (Ehlke and Keller 2003). Chum salmon were not observed in any upper Columbia Gorge tributaries during the 2003 and 2004 spawning ground surveys. Finally, most Columbia River chum populations have been functionally extirpated or are presently at very low abundance levels.

Historically, the Columbia River chum salmon supported a large commercial fishery in the first half of this century which landed more than 500,000 fish per year as recently as 1942. Commercial catches declined beginning in the mid-1950s, and in later years rarely exceeded 2,000 per year. During the 1980s and 1990s, the combined abundance of natural spawners for the lower Columbia Gorge, Washougal, and Grays River populations was below 4,000 adults. In 2002, however, the abundance of natural spawners exhibited a substantial increase at several locations (estimate of natural spawners is approximately 20,000 adults). The cause of this dramatic increase in abundance is unknown. However, long- and short-term productivity trends for populations are at or below replacement. The loss of off-channel habitat and the extirpation of approximately 17 historical populations increase this species' vulnerability to environmental variability and catastrophic events. Overall, the populations that remain have low abundance, limited distribution, and poor connectivity (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for Columbia River chum salmon on September 2, 2005 (70 FR 52630). The designated includes defined areas in the following subbasins: Middle Columbia/Hood, Lower Columbia/Sandy, Lewis, Lower Columbia/Clatskanie, Lower Cowlitz, Lower Columbia subbasin and river corridor. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation.

The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more chum salmon life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding and are rated as having high conservation value to the species. Columbia River chum salmon have primary constituent elements of freshwater spawning, freshwater rearing, freshwater migration, estuarine areas free of obstruction, nearshore marine areas free of obstructions, and offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 21 subbasins reviewed in NMFS' assessment of critical habitat for the Columbia River chum salmon ESU, three subbasins were rated as having a medium conservation value, no subbasins were rated as low, and the majority of subbasins (18), were rated as having a high conservation value to Columbia River chum salmon. The major factors limiting recovery for Columbia River chum salmon are altered channel form and stability in tributaries, excessive sediment in tributary spawning gravels, altered stream flow in tributaries and the mainstem Columbia River, loss of some tributary habitat types, and harassment of spawners in the tributaries and mainstem.

Hood Canal Summer-Run Chum Salmon

Distribution and Description of the Listed Species

The Hood Canal summer-run chum salmon ESU includes all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries as well as populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington (64 FR 14508) from mid-September to mid-October (WDF (Washington Department of Fisheries) 1993), but may enter natal rivers in late August. Eight artificial propagation programs are considered to be part of the ESU: the Quilcene National Fish Hatchery, Hamma Hamma Fish Hatchery, Lilliwaup Creek Fish Hatchery, Union River/Tahuya, Big Beef Creek Fish Hatchery, Salmon Creek Fish Hatchery, Chimacum Creek Fish Hatchery, and the Jimmycomelately Creek Fish Hatchery summer-run chum hatchery programs. NMFS determined that these artificially propagated populations are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the species. Table 11 identifies populations within the Hood Canal summer-run chum salmon ESU, their abundances, and hatchery input.

On average Hood Canal chum salmon reside in estuaries for 23 days; daily tidal migrations have not been observed, but prey availability does influence movement patterns (Bax 1983). Upon leaving their natal estuaries summer-run chum salmon generally migrate through Hood Canal and into the main body of Puget Sound.

Status and Trends

NMFS listed Hood Canal summer-run chum salmon as threatened on March 25, 1999 (64 FR 14508), and reaffirmed as threatened on June 28, 2005 (70 FR 37160). Historically, Hood Canal summer-run chum salmon comprised an estimated 16 populations. Only eight extant populations remain within this ESU (Good et al. 2005). Most of the extirpated populations historically occurred on the eastern side of Hood Canal, which is cause for concern over the current spatial structure of this ESU. The widespread loss of estuary and lower floodplain habitat is a continuing threat to ESU spatial structure and connectivity.

Although many population remain adult returns for some populations showed modest improvements in 2000, with upward trends continuing in 2001 and 2002. The recent 5-year mean abundance is variable among populations in the species, ranging from one fish to nearly 4,500 fish in the Big/Little Quilcene rivers. Hood Canal summer-run chum are the focus of an extensive rebuilding program developed and implemented since 1992 by the state and tribal comanagers. Two populations (the combined Quilcene and Union River populations) are above the conservation thresholds established by the rebuilding plan. However, most populations remain depressed. Estimates of the fraction of naturally spawning hatchery fish exceed 60% for some populations, indicating that reintroduction programs are supplementing the numbers of total fish spawning naturally in streams. Long-term trends in productivity are above replacement for only the Quilcene and Union River populations. Buoyed by recent increases, seven populations are exhibiting short-term productivity trends above replacement.

Table 11. Hood Canal summer-run chum populations and selected measures of population viability

Populations [a]	1999-2002 Mean Escapement (range)	Percent Hatchery Contributions (1995-2001)	λ (+/- SE)
Jimmycomelately Creek	10 (1-192)		0.85 (0.16)
Salmon/Snow creeks	1,521 (463-5,921)	0-69	1.23 (0.10)
Big/Little Quilcene rivers	4,512 (3,065-6,067)	5-51	1.39(0.22)
Lilliwaup Creek	13 (1-775)		1.19 (0.44)
Hamma Hamma River	558 (173-2,260)		1.3 (0.19)
Duckabush River	382 (92-942)		1.1 (0.17)
Dosewallips River	919 (351-1,627)		1.17 (0.24)
Union River			1.15 (0.10))
Chimacum Creek*	198 (0-903)[c]	100	
Big Beef Creek*	17 (0-826)[c]	100	
Dewatto Creek*	9 (2-32)[d]		

[a]All data is reported in Good et al. 2005. * Denotes extinct populations that have recently had some natural recolonization or have been seeded with hatchery fish.

Of the eight programs releasing summer-run chum salmon that are considered to be part of the Hood Canal summer chum ESU, six of the programs are supplementation programs implemented to preserve and increase the abundance of native populations in their natal watersheds. NMFS' assessment of the effects of artificial propagation on ESU extinction risk concluded that these hatchery programs collectively do not substantially reduce the extinction risk of the ESU. The hatchery programs are reducing risks to ESU abundance by increasing total ESU abundance as well as the number of naturally spawning summer-run chum salmon.

Critical Habitat

NMFS designated critical habitat for Hood Canal summer-run chum salmon on September 2, 2005 (70 FR 52630). The specific geographic area includes the Skokomish River, Hood Canal subbasin, which includes the Hamma Hamma and Dosewallips rivers and others, the Puget Sound subbasin, Dungeness/Elwha subbasin, and nearshore marine areas of Hood Canal and the Strait of Juan de Fuca from the line of extreme high tide to a depth of 30 meters. This includes a narrow nearshore zone from the extreme high-tide to mean lower low tide within several Navy security/restricted zones. This also includes about 8 miles of habitat that was unoccupied at the time of the designation Finch, Anderson and Chimacum creeks (69 FR 74572; 70 FR 52630), but has recently been re-seeded. Chimacum Creek, however, has been naturally recolonized since at least 2007 (T. Johnson, pers. comm., Jan. 2010). The designation for Hood Canal summer-run chum, like others made at this time, includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation.

The specific primary constituent elements identified for Hood Canal summer-run chum salmon are areas for spawning, freshwater rearing and migration, estuarine areas free of obstruction, nearshore marine areas free of obstructions, and offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 17 subbasins reviewed in NMFS' assessment of critical habitat for the Hood Canal chum salmon ESU, 14 subbasins were rated as having a high conservation value, while only three were rated as having a medium value to conservation. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Limiting factors identified for this species include degraded floodplain and mainstem river channel structure, degraded estuarine conditions and loss of estuarine habitat, riparian area degradation and loss of in-river wood in mainstem, excessive sediment in spawning gravels, and reduced stream flow in migration areas.

Coho Salmon

Description of the Species

Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle et al. 1986). The typical life history of coho salmon is similar to most of the other large bodied Pacific salmonids, in as much as adult fish spawn in the fall and winter, young emerge in the spring, rear in fresh water and saltwater and return to spawn as adults. Sympatric in many river basins with Chinook, chum, sockeye, and pink salmon, partitioning occurs through the species use of different areas of a river for reproduction and rearing, and the length of time they spend in these ecosystems. For instance, Chinook salmon spawn in fast flowing mainstem riverine reaches with large substrate; sockeye salmon spawn in rivers and lakes, and chum salmon spawn in mid- to lower reaches of rivers and have been observed spawning in areas of tidal influence. Coho salmon characteristically spawn in tributaries and slow-flowing shallow creeks in tributaries with gradients of three percent or less, which may be fed by cool groundwater sources, and are often widely dispersed within watershed. Adult coho salmon may remain in fresh water three or more months before spawning, with early migrants often moving farther upstream than later migrants (Sandercock 1991).

Most coho salmon enter rivers between September and February, but entry is influenced by discharge and other factors. In many river systems, coho salmon and other Pacific salmon are unable to enter the rivers until sufficiently strong flows open passages and provide sufficient depth. First fall freshets combined with high tides triggers the upstream migration of coho salmon in many basins. Until then, if river flows are low or warm summer temperatures persist, fish may congregate in pools near the mouth of the river or natal stream until conditions change. Typically coho salmon spawn from November to January, although there are many exceptions throughout their range. Spawning duration usually spans about three months in most basins, with individual fish actively spawning for several days to weeks. Spawning occurs in a few third-order streams, but most spawning activity occurs in fourth- and fifth-order streams. As with other Pacific salmon, coho salmon fecundity varies with the size of the fish and latitudinally with coho salmon in northern climes generally exhibiting higher rates of fecundity (Sandercock 1991). Most coho salmon mature and spawn at age 3, although there are exceptions; in many basins in the northern portion of the species' range coho salmon spawn at age 2.

Rates of incubation are largely temperature dependent: colder water temperatures will slow the rate of development. Generally, in optimal temperatures eggs incubate for about 35 to 50 days, and fry start emerging from the gravel two to three weeks after hatching. Incubation and emergence success are also influenced by dissolved oxygen levels, sediment loading, and scouring high flows. Following emergence, fry aggregate and move to shallow areas near the stream banks. Most coho salmon rear in fresh water for about 15 to 18 months. As fry grow, they disperse upstream and downstream to establish and defend territories. Juvenile rearing usually occurs in tributaries with gradients of three percent or less, although they may move to streams with gradients of four to five percent. Juvenile coho salmon are often found in small streams less than five feet wide, and may migrate considerable distances to rear in lakes and off-channel ponds. During the summer, fry prefer pools featuring adequate cover such as large woody debris, undercut banks, and overhanging vegetation. Overwintering tends to occur in larger pools, backwater areas and off stream channels and ponds (e.g., wall-based channels that are groundwater fed).

At not quite 2 years of age, coho salmon will migrate downstream where they, like other anadromous fish, undergo the physiological transition to salt water. The outmigration of coho smolts begins as early as February and may continue through the summer and fall, with peak outmigration often between March and June, although this varies among basins and environmental conditions (Sandercock 1991). Once in the ocean, coho salmon generally migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. During this migration, juvenile coho salmon tend to occur in both coastal and offshore waters. During spring and summer, coho salmon will forage in waters between 46º N, the Gulf of Alaska, and along Alaska's Aleutian Islands.

Coho salmon, like many other salmon, are opportunistic feeders. While at sea, coho salmon tend to eat fish including herring, sand lance, sticklebacks, sardines, shrimp and surf smelt. While in estuaries and in fresh water coho salmon are significant predators of Chinook, pink, and chum salmon, as well as aquatic and terrestrial insects. Smaller fish, such as fry, eat chironomids, plecoptera, and other larval insects, and typically use visual cues to find their prey.

Threats

Natural Threats. Coho salmon, like other salmon, are exposed to high rates of natural predation at each life stage. Most mortality, however, occurs in the freshwater life stages. Winter mortality may be significant for coho salmon because they overwinter in fresh water, where they can be swept downstream from freshets or eaten by raccoon, cutthroat trout, or other small animals. Once coho reach the ocean, survival is high (Sandercock 1991).

Anthropogenic Threats. Coho salmon have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile coho salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the fresh water, estuarine, and coastal ecosystems throughout the species' range. These threats for are summarized in detail under Chinook salmon.

Central California Coast Coho Salmon

Distribution and Description of the Listed Species

The Central California Coast coho salmon ESU extends from Punta Gorda in northern California south to and including the San Lorenzo River in central California (Weitkamp et al. 1995). The ESU includes all naturally spawned populations of coho salmon from Punta Gorda in northern California south to and including the San Lorenzo River in central California, as well as populations in tributaries to San Francisco Bay, excluding the Sacramento-San Joaquin River system. Four artificial propagation programs are part of the Central California Coast coho salmon ESU: the Don Clausen Fish Hatchery Captive Broodstock Program, Scott Creek/King Fisher Flats Conservation Program, Scott Creek Captive Broodstock Program, and the Noyo River Fish Station egg-take Program coho hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Coho salmon in this ESU enter rivers to spawn very late (peaking in January), with little time spent in fresh water between river entry and spawning. This compressed adult freshwater residency appears to coincide with the single, brief peak of river flow characteristic of this region.

Status and Trends

NMFS originally listed the central California coast coho salmon ESU as threatened on October

31, 1996 (61 FR 56138) and later reclassified their status to endangered June 28, 2005 (70 FR 37160). Information on the abundance and productivity trends for the naturally spawning component of the central California coast coho ESU is extremely limited. There are no long-term time series of spawner abundance for individual river systems. Historical estimated escapement for this ESU is 56,100 for 1963, and more recent estimates suggest the ESU dropped to about one-fourth that size by the late 1980s and early 1990s (Good et al. 2005).

Where data are available, analyses of juvenile coho presence-absence information, juvenile density surveys, and irregular adult counts for the South Fork Noyo River indicate low abundance and long-term downward trends for the naturally spawning populations throughout the ESU. Improved ocean conditions coupled with favorable stream flows and harvest restrictions have contributed to increased returns in 2001 in streams in the northern portion of the ESU, as indicated by an increase in the observed presence of fish in historically occupied streams. Data are particularly lacking for many river basins in the southern two-thirds of the ESU where naturally spawning populations are considered to be at the greatest risk. The extirpation or near extirpation of natural coho salmon populations in several major river basins, and across most of the southern historical range of the ESU, represents a significant risk to ESU spatial structure and diversity (Good et al. 2005).

Artificial propagation of coho salmon within the Central California Coast ESU has declined since the ESU was listed in 1996 though it continues at the Noyo River and Scott Creek facilities, and two captive broodstock populations have recently been established. Genetic diversity risk associated with out-of-basin transfers appears to be minimal, but diversity risk from domestication selection and low effective population sizes in the remaining hatchery programs remains a concern. An out-of-ESU artificial propagation program for coho was operated at the Don Clausen hatchery on the Russian River through the mid 1990s, but was terminated in 1996. Termination of this program was considered by the biological review team as a positive development for naturally produced coho in this ESU.

For the naturally spawning component of the ESU, the biological review team found very high risk of extinction for the abundance, productivity, and spatial structure of the Viable Salmon Population (VSP) parameters and comparatively moderate risk with respect to the diversity VSP parameter. The lack of direct estimates of the performance of the naturally spawned populations in this ESU, and the associated uncertainty this generates, was of specific concern to the biological review team. Informed by the VSP risk assessment and the associated uncertainty, the strong majority opinion of the biological review team was that the naturally spawned component of the Central California Coast coho ESU was “in danger of extinction.” The minority opinion was that this ESU is “likely to become endangered within the foreseeable future.” (70 FR 37160) Accordingly, NMFS upgraded the status of central California coast coho ESU to endangered on June 28, 2005 (70 FR 37160).

Central California Coast coho salmon populations continue to be depressed relative to historical numbers. Strong indications show that breeding groups have been lost from a significant percentage of historical stream range. A number of coho populations in the southern portion of the range appear to be either extinct or nearly so, including those in Gualala, Garcia, and Russian rivers, as well as smaller coastal streams in and south of San Francisco Bay (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for central California coast coho salmon on May 5, 1999 (64 FR 24049). The designation encompasses accessible reaches of all rivers (including estuarine areas and riverine reaches) between Punta Gorda and the San Lorenzo River (inclusive) in California, including two streams entering San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creek. This critical habitat designation includes all waterways, substrate, and adjacent riparian zones of estuarine and riverine reaches (including off-channel habitats) below longstanding naturally impassable barriers (i.e. natural waterfalls in existence for at least several hundred years). These areas are important for the species' overall conservation by protecting growth, reproduction, and feeding.

Lower Columbia River Coho Salmon

Distribution and Description of the Listed Species

The lower Columbia River coho salmon ESU includes all naturally spawned populations of coho salmon in the Columbia River and its tributaries in Washington and Oregon, from the mouth of the Columbia up to and including the Big White Salmon and Hood Rivers, and includes the Willamette River to Willamette Falls, Oregon, Twenty-five artificial propagation programs are part of this ESU: Grays River, Sea Resources Hatchery, Peterson Coho Project, Big Creek Hatchery, Astoria High School (STEP) Coho Program, Warrenton High School (STEP) Coho Program, Elochoman Type-S Coho Program, Elochoman Type-N Coho Program, Cathlamet High School FFA Type-N Coho Program, Cowlitz Type-N Coho Program in the Upper and Lower Cowlitz Rivers, Cowlitz Game and Anglers Coho Program, Friends of the Cowlitz Coho Program, North Fork Toutle River Hatchery, Kalama River Type-N Coho Program, Kalama River Type-S Coho Program, Lewis River Type-N Coho Program, Lewis River Type-S Coho Program, Fish First Wild Coho Program, Fish First Type-N Coho Program, Syverson Project Type-N Coho Program, Washougal River Type-N Coho Program, Eagle Creek NFH, Sandy Hatchery, and the Bonneville/Cascade/ Oxbow complex coho hatchery programs.

Two distinct runs distinguished by the timing of adult returns to fresh water (early returners and later returners) occur within the ESU. Early returning adults generally migrate south of the Columbia River once they reach the ocean, returning to fresh water in mid-August and to spawning tributaries in early September. Peak spawning of early returning adults occurs from mid-October to early November. Late returning adult coho salmon exhibit a northern oceanic distribution, return to the Columbia River from late September through December, and enter tributaries from October through January. Most late return adults spawn between November through January, although some spawn in February and as late as March (LCFRB 2004). Almost all Lower Columbia River ESU coho salmon females and most males spawn at 3 years of age.

Status and Trends

NMFS listed Lower Columbia River coho salmon as endangered on June 28, 2005 (70 FR 37160). The vast majority (over 90%) of the historic population in the Lower Columbia River coho salmon ESU appear to be either extirpated or nearly so. Recent counts of natural-origin spawners and the recent fraction of hatchery-origin spawners are noted in Table 12 , where available.

Only two populations of coho salmon within this ESU produce a sizeable number of naturally spawned fish, the upper Sandy River population above Marmot Dam and the Clackamas River population above the North Fork Dam. Most of the other populations are believed to have very little, if any, natural production. The long-term and short-term trends for Marmot Dam counts are both negative. The long-term median growth rate is slightly positive for both the Sandy and Clackamas rivers, but the confidence intervals for each are very wide indicating there is a large amount of uncertainty. Both populations within the Sandy and Clackamas rivers have suffered from recruitment failure a number of times over the past 15 years, despite the reductions in harvest.

Table 12. Lower Columbia River coho salmon populations and selected measures of population viability

River	2002 Spawner Count[a]	Geometric Mean Abundance 2000-2002[b]	Percent Hatchery Contributions[c]	Long-term Median Growth Rate (λ)[d]
Youngs Bay and Big Creek	4,473		91	
Grays River				
Elochoman River				
Clatskanie River	229		60	
Mill, Germany, and Abernathy creeks				
Scappoose Rivers	458		0	
Cispus River				
Tilton River				
Upper Cowlitz River				
Lower Cowlitz River				
North Fork Toutle River				
South Fork Toutle River				
Coweeman River				
Kalama River				
North Fork Lewis River				
East Fork Lewis River				
Upper Clackamas River	1,001	2,122	12	1.009 (0.898-1.177)
Lower Clackamas River	2,402		78	
Salmon Creek				
Upper Sandy River	310	643	0	1.012 (0.874-1.172)
Lower Sandy River	271		97	
Washougal River				
Columbia River Gorge – lower tributaries				
White Salmon				
Columbia River Gorge – upper tributaries	1,317		>65	
Hood River				

[a]All data are reported in Good et al. 2005. Spawner data from 2002 only.
[b]Geometric mean number of coho salmon above the dams. * is a combined totoal for the upper and lower Clackamas River. Reported in Good et al. 2005
[c]Hatchery production likely dominates yearly returns for the ESU as a whole.
[d]The λ calculated estimates the natural growth rate after accounting for hatchery-origin spawners. The estimate provided above assumes that hatchery-origin spawners make no reproductive contribution. The λ for the Clackamas River is calculated with data spanning 1973-2002, and for the Sandy River covers 1977-2002. The Clackamas River value includes both early-run and late-run coho salmon.

The most serious threat facing this ESU is the scarcity of naturally-produced spawners, with attendant risks associated with small populations, loss of diversity, and fragmentation and isolation of the remaining naturally-produced fish. Spatial structure has been substantially reduced by the loss of access to upper basins from tributary hydro development (i.e., Condit Dam on the Big White Salmon River and Powerdale Dam on the Hood River). The diversity of populations in all three areas has been eroded by large hatchery influences and periodically, low effective population sizes.

Critical Habitat

NMFS has not designated critical habitat for Lower Columbia River coho salmon.

Southern Oregon/Northern California Coast Coho Salmon

Distribution and Description of the Listed Species

Southern Oregon/Northern California coast coho salmon consists of all naturally spawning populations of coho salmon that reside below long-term, naturally impassible barriers in streams between Punta Gorda, California and Cape Blanco, Oregon, as well as three artificial propagation programs: the Cole Rivers Hatchery, Trinity River Hatchery, and Iron Gate Hatchery coho hatchery programs. The three major river systems supporting Southern Oregon – Northern Coastal California coast coho are the Rogue, Klamath (including the Trinity), and Eel rivers.

Southern Oregon and Northern California coast coho immigrate to natal rivers in September or October. River entry is much later south of the Klamath River Basin, occurring in November and December, as well as in basins south of the Klamath River to the Mattole River, California. River entry occurs from mid-December to mid-February in rivers farther south. Because individuals enter rivers late, they spend much less time in the river. Coho salmon adults spawn at age 3, spending just over 1 year in fresh water and a year and a half in the ocean.

Status and Trends

Southern Oregon/Northern California coast coho salmon were listed as threatened on May 7, 1997 (62 FR 24588); they retained that classification when their status was reviewed on June 28, 2005 (70 FR 37160). Southern Oregon/Northern California Coast coho salmon extend from Cape Blanco in southern Oregon to Punta Gorda in northern California (Weitkamp et al. 1995). The status of coho salmon coast-wide, including the Southern Oregon/Northern California Coast coho salmon ESU, was formally assessed in 1995 (Weitkamp et al. 1995). Two subsequent status review updates have been published by NMFS, one addressing all West Coast coho salmon ESUs and a second specifically addressing the Oregon Coast Southern Oregon/Northern California Coast coho salmon ESUs (NMFS 1996, 1997). In the 1997 status update, estimates of natural population abundance were based on very limited information. New data on presence/absence in northern California streams that historically supported coho salmon were even more disturbing than earlier results, indicating that a smaller percentage of streams contained coho salmon compared to the percentage presence in an earlier study. However, it was unclear whether these new data represented actual trends in local extinctions or were biased by sampling effort.

Data on population abundance and trends are limited for the California portion of this ESU. No regular estimates of natural spawner escapement are available. Historical point estimates of coho salmon abundance for the early 1960s and mid-1980s suggest that statewide coho spawning escapement in the 1940s ranged between 200,000 and 500,000 fish. Numbers declined to about 100,000 fish by the mid-1960s with about 43% originating from this ESU. Brown et al. (1994) estimated that the California portion of this ESU was represented by about 7,000 wild and naturalized coho salmon (Good et al. 2005). In the Klamath River, the estimated escapement has dropped from approximately 15,400 in the mid-1960s to about 3,000 in the mid 1980s, and more recently to about 2,000 (Good et al. 2005). The second largest producing river in this ESU, the Eel River, dropped from 14,000, to 4,000 to about 2,000 during the same period. Historical estimates are considered “best guesses” made using a combination of limited catch statistics, hatchery records, and the personal observations of biologists and managers.

Most recently, Williams et al. (2006) described the structure of historic populations of Southern Oregon/Northern California Coast coho salmon. They described three categories of populations: functionally independent populations, potentially independent populations and dependent populations. Functionally independent populations are populations capable of existing in isolation with a minimal risk of extinction. Potentially independent populations are similar but rely on some interchange with adjacent populations to maintain a low probability of extinction. Dependent populations have a high risk of extinction in isolation over a 100-year timeframe and rely on exchange of individuals from adjacent populations to maintain themselves.

Critical Habitat

NMFS designated critical habitat for Southern Oregon/Northern California Coast coho salmon on May 5, 1999 (64 FR 24049). Critical habitat for this species encompasses all accessible river reaches between Cape Blanco, Oregon, and Punta Gorda, California. Critical habitat consists of the water, substrate, and river reaches (including off-channel habitats) in specified areas. Accessible reaches are those within the historical range of the ESU that can still be occupied by any life stage of coho salmon. Of 155 historical streams for which data are available, 63% likely still support coho salmon. These river habitats are important for a variety of reasons, such as supporting the feeding and growth of juveniles and serving as spawning habitat for adults. Limiting factors identified for this species include: loss of channel complexity, connectivity and sinuosity, loss of floodplain and estuarine habitats, loss of riparian habitats and large in-river wood, reduced stream flow, poor water quality, temperature and excessive sedimentation, and unscreened diversions and fish passage structures.

Oregon Coast Coho Salmon

Distribution and Description of the Listed Species

The Oregon Coast coho salmon ESU includes all naturally spawned populations of coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco (63 FR 42587; August 1998). One hatchery population, the Cow Creek hatchery coho salmon, is considered part of the ESU. Table 13 identifies populations within the Oregon Coast coho salmon ESU, their abundances, and hatchery input.

Table 13. Oregon Coast coho populations and selected measures of population viability

Basin[a]	Mean Spawner Abundance[b]	13-Year Spawner Trend (SE)[c]	Percent Hatchery Contribution[d]
Necanicum	1,889	1.169 (0.860)	2.9-6.4
Nehalem	18,741	1.206 (0.889)	0.5-26.0
Tillamook Bay	3,949	1.191 (1.084)	0-5.6
Nestucca	3,846	1.230 (1.015)	0-10.4
Siletz	2,295	1.070 (0.760)	1.8-100
Yaquima	3,665	1.204 (1.205)	0-37.5
Alsea	3,621	1.042 (0.960)	0-87.5
Siuslaw	16,213	1.120 (1.037)	0.3-11.1
Umpqua	24,351	1.182 (0.662)	2.1-8.3
Coos	20,136	1.088 (1.066)	0-1.9
Coquille	8,847	1.070 (0.649)	0-6.0

[a]Population structure is unclear. The above data reflects the assumption that spawners from major river basins are largely isolated, and each basin comprises a population. All data are reported in Good et al. 2005.
[b]Recent 3-year geometric mean of natural-origin spawners.
[c]Data years 1990-2002.
[d]Data represents the range of percent hatchery contributions from 1998 through 2002 (from Jacobs et al. 2002, 2001, and 2002 as cited in Good et al. 2005).

Status and Trends

The Oregon coast coho salmon ESU was listed as a threatened species under the ESA on February 11, 2008 (73 FR 7816), the conclusion to a 13-year history of court cases. The most recent NMFS status review for the Oregon Coast coho ESU was conducted by the biological review team in 2003, which assessed data through 2002. The abundance and productivity of Oregon Coast coho since the previous status review represented some of the best and worst years on record (NMFS 1997a). Yearly adult returns for the Oregon Coast coho ESU were over 160,000 natural spawners in 2001 and over 260,000 in 2002, far exceeding the abundance observed for the past several decades. These increases in spawner abundance in 2000 to 2002 followed three consecutive brood years (the 1994 to 1996 brood years returning in 1997 to 1999, respectively) exhibiting recruitment failure (recruitment failure is when a given year class of natural spawners fails to replace itself when its offspring return to the spawning grounds 3 years later). These 3 years of recruitment failure were the only such instances observed thus far in the entire 55-year abundance time series for Oregon Coast coho salmon (although comprehensive population-level survey data have only been available since 1980). The 2000 to 2002 increases in natural spawner abundance occurred in many populations in the northern portion of the ESU, which were the most depressed at the time of the last review (NMFS 1997a). Although encouraged by the increase in spawner abundance in 2000 to 2002, the biological review team noted that the long-term trends in ESU productivity were still negative due to the low abundances observed during the 1990s.

Since the biological review team convened, the total abundance of natural spawners in the Oregon Coast coho ESU has declined each year (i.e., 2003 to 2006). The abundance of total natural spawners in 2006 (111,025 spawners) was approximately 43 % of the recent peak abundance in 2002 (255,372 spawners). In 2003, ESU-level productivity (evaluated in terms of the number of spawning recruits resulting from spawners 3 years earlier) was above replacement, and in 2004, productivity was approximately at replacement level. However, productivity was below replacement in 2005 and 2006, and dropped to the lowest level since 1991 in 2006 (73 FR 7816).

Preliminary spawner survey data for 2007 (the average peak number of spawners per mile observed during random coho spawning surveys in 41 streams) suggest that the 2007 to 2008 return of Oregon Coast coho is either (1) much reduced from abundance levels in 2006, or (2) exhibiting delayed run timing from previous years. As of December 13, 2007, the average peak number of spawners per mile was below 2006 levels in 38 of 41 surveyed streams (ODFW 2007 *in* 73 FR 7816). It is possible that the timing of peak spawner abundance is delayed relative to previous years, and that increased spawner abundance in late December and January 2008 will compensate for the low levels observed thus far.

The recent 5-year geometric mean abundance (2002 to 2006) of approximately 152,960 total natural spawners remains well above that of a decade ago (approximately 52,845 from 1992 to 1996). However, the decline in productivity from 2003 to 2006, despite generally favorable marine survival conditions and low harvest rates, is of concern (73 FR 7816).

Critical Habitat

NMFS designated critical habitat for Oregon Coast coho on February 11, 2008 (73 FR 7816). The designation includes 72 of 80 watersheds occupied by Oregon Coast coho salmon, and totals about 6,600 stream miles including all or portions of the Nehalem, Nestucca/Trask, Yaguina, Alsea, Umpqua and Coquille basins. These areas are essential for feeding, migration, spawning, and rearing. The specific primary constituent elements include: spawning sites with water and substrate quantity to support spawning, incubation, and larval development; freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth, foraging, behavioral development (e.g., predator avoidance, competition), and mobility; freshwater migratory corridors free of obstruction with adequate water quantity and quality conditions; and estuarine, nearshore and offshore areas free of obstruction with adequate water quantity, quality and salinity conditions that support physiological transitions between fresh- and saltwater, predator avoidance, foraging and other life history behaviors.

Sturgeon

Description of the Genus

Sturgeon (*Acipenseridae*) are one of the oldest *Osteichthyes* (bony fish) in existence, and are native to rivers and coastal areas of North America. The two listed sturgeon, discussed below, are part of the genus *Acipenser*, and share some common characteristics. Sturgeon, in general, have a characteristic external morphology distinguished by the inferior mouth typical of bottom-feeders. Most species are anadromous, although a few species are entirely fresh water and many species can survive if they become land-locked. Both listed species (discussed below) are anadromous and tend to remain in coastal waters. As an anadromous fish, sturgeon spawn in fresh water and feed and rear in marine or estuarine waters. Sturgeon are also iteroparous spawners and tend to be very long-lived.

Threats

Natural Threats. Freshwater predation of eggs and larvae from birds and larger fish, and marine predation of adult and subadult fish by sharks, pinnipeds and other large body predators.

Anthropogenic Threats. In general sturgeon have declined from the combined effects from the construction of dam and water diversion projects, dredging and blasting, water pollution, and fisheries. As a result of their longevity, slow rate of growth and delayed maturation, and bottom-feeding habits, in general sturgeon have a life history that makes them susceptible to over-harvest and exposure to (and the accumulation of) contaminants. Many sturgeon also do not spawn on an annual basis, but may spawn every other year or even more infrequently. Thus even small increases in mortality can affect population productivity (Heppell 2007). The body form and feeding habits of sturgeon may expose them to a different suite of contaminants or contaminant properties than pelagic fish due to their affinity with bottom sediments. Exposure pathways would include a dissolved or water borne exposure, but for sediment-associated contaminants the sediment exposure pathway may be more significant. Benthic dwelling fish may be exposed through the direct contact with sediment, exposed to the boundary layer over the sediment, and commonly have a higher rate of incidental ingestion and exposure through direct consumption of sediments.

Southern Green Sturgeon

Distribution and Description of the Listed Species

Green sturgeon occur along the west coast of North America from Mexico to the Bering Sea (Adams et al. 2002; Moyle 2002; Colway and Stevenson 2007). Distinguished primarily according genetic differences and spawning locations, NMFS recognizes two distinct population segments (DPS) of green sturgeon: a northern DPS whose populations are relatively healthy, and a Southern DPS that has undergone significant decline (Adams et al. 2007). NMFS listed the Southern DPS of green sturgeon as threatened in 2006 (71 FR 17757).

Green sturgeon are considered one of the most marine-oriented sturgeon species, spending much of their lives in coastal marine waters, estuaries and bays. Early life stages rear in fresh water, and adults return to fresh water when they are 15 years old or older to spawn. Across the species' range only three rivers contain documented spawning and only one of the rivers is part of the southern green sturgeon DPS, the Sacramento River (Moyle et al. 1992; CDFG 2002). Outside of natal rivers, the distribution of southern green sturgeon and northern green sturgeon overlap. Both northern DPS and southern DPS green sturgeon occupy coastal estuaries and coastal marine waters from southern California to Alaska, including Humbolt Bay, the lower Columbia River estuary, Willapa Bay, Grays Harbor and southeast Alaska. In general, green sturgeon are more common north of Point Conception, California.

Green sturgeon are spring spawners and initiate spawning migrations as early as March, spawn late spring to early summer, hold in deep pools and return to salt water in the fall early, often with the first increases in fall flows. There may a be a latitudinal cline in the timing of upstream spawning migrations, as fish in the Klamath River have been observed initiating migrations between April and June, Rogue River fish between May and July, whereas Heubein et al. (2009) observed Sacramento River fish making their upstream migrations between March and April. Spawning generally occurs in deep pools of large rivers or off-channel coves (Moyle et al. 1992, 1995; Erickson and Webb 2007; Erickson et al. 2001; Heublein et al. 2009; Rien et al. 2001). Fish then tend to aggregate in deep pools, where they will over-summer before outmigrating in the fall, although some fish have been observed outmigrating relatively soon after presumed spawning events (Heubein et al. 2009). In the Sacramento River adult green sturgeon spawn in late spring and early summer above Hamilton City, above Red Bluff Diversion Dam, and possibly as far upstream as Keswick Dam (CDFG 2002; Heubein et al. 2009). It appears that specific habitat for spawning includes large cobblestones (where eggs can settle between), although spawning is known to occur over clean sand or bedrock.

Green sturgeon are a long-lived fish, and likely live for 60 to 70 years (Moyle 2002). Age at first maturation for green sturgeon is at least 15 years old, after which adults likely return every 2 to 5 years to spawn (Adams et al. 2002; Moyle 2002; Van Eenennaam et al. 2006). Most male spawners are young (17 to 18 years) while females on the spawning grounds are often older (27 to 28 years). Females produce roughly 60,000 to 140,000 eggs per spawning event (Scott and Crossman 1973; Moyle et al. 1992). Temperature may trigger spawning behavior, with ranges of 48º to 62º F being influential (Moyle et al. 1995). Water temperature is also critical for egg survival with temperatures above 68º F being fatal to developing embryos (Cech et al. 2000).

Green sturgeon spend their first 1 to 4 years in their natal streams and rivers (Nakamoto et al. 1995; Beamsesderfer and Webb 2002), although they are believed to be physiologically adapted to sea water survival at 6 months of age (Allen and Cech 2007; Allen et al. 2009a, b). Larvae are active at night, a behavior that likely reduces predation and avoids being moved downstream more than necessary (Cech et al. 2000). Green sturgeon larvae grow very rapidly, reaching about 300 mm by age one (Deng 2000). Temperature is strongly correlated with growth rates, with optimal growth rates occurring at about 59º F (Cech et al 2000). While in fresh water, juveniles feed on a variety of fishes and invertebrates (Moyle et al. 1992). One juvenile from the Sacramento-San Joaquin estuary was found to have preyed most commonly upon opisthobranch mollusks (*Philline* sp.), with bay shrimp (*Crangon* sp.) and overbite clams (*Potamocorbula amurensis*) as secondary prey. Other juveniles in the Sacramento River delta feed on opossum shrimp (*Neomysis mercedis*) and *Corophium* amphipods (Radtke 1966).

Upon outmigration from fresh water, subadult green sturgeon disperse widely along through continental shelf waters of the west coast within the 110 meter contour (Moyle et al. 1992; NMFS 2005b; Erikson and Hightower 2007). Biologists have recaptured fish tagged in the Sacramento River, in coastal and estuarine waters to the north. It appears that green sturgeon generally distribute north of the river mouth from whence they emerge as juveniles during fall and move into bays and estuaries during summer and fall (Israel et al. 2009; Moser and Lindley 2007). The limited feeding data available for subadult and adult green sturgeon show that they consume benthic invertebrates including shrimp, clams, chironomids, copepods, mollusks, amphipods, and small fish (Houston 1988; Moyle et al. 1992; Wilson and McKinley 2004; Dumbauld et al. 2008). Starting as larvae, sturgeon use electroreception to identify prey. Olfaction and taste may also be important to foraging, while vision is thought play a minor role in prey capture (Miller 2004).

Status and Trends

NMFS listed the southern population of the North American green sturgeon as threatened on April 7, 2006 (71 FR 17757). Trend data for green sturgeon is severely limited. Available information comes from two predominant sources, fisheries and tagging. Only three data sets were considered useful for the population time series analyses by NMFS' biological review team: the Klamath Yurok Tribal fishery catch, a San Pablo sport fishery tag returns, and Columbia River commercial landings (NMFS 2005b). Using San Pablo sport fishery tag recovery data, the California Department of Fish and Game produced a population time series estimate for the southern DPS. San Pablo data suggest that green sturgeon abundance may be increasing, but the data showed no significant trend. The data set is not particularly convincing, however, as it suffers from inconsistent effort and since it is unclear whether summer concentrations of green sturgeon provide a strong indicator of population performance (NMFS 2005b). Although there is not sufficient information available to estimate the current population size of southern green sturgeon, catch of juveniles during state and federal salvage operations in the Sacramento delta are low in comparison to catch levels before the mid-1980s.

Threats

Natural Threats. Green sturgeon eggs and larvae are likely preyed upon by a variety of larger fish and animals, while sub-adult and adult sturgeon may occasionally be preyed upon by shark sea lions, or other large body predators. Physical barriers, changes in water flow and temperatures may also affect fresh water survival.

Anthropogenic Threats. The principle threat to southern green sturgeon comes from a drastic reduction in available spawning area from impassible barriers (e.g., Oroville, Shasta and Keswick dams). Other threatens include potentially lethal temperature limits, harvest, entrainment by water projects and toxins and invasive species (Adams et al. 2007; Erickson and Webb 2007; Lackey 2009). Since this DPS is composed of a single spawning population within the Sacramento River, stochastic variation in environmental conditions and significant fluctuations in demographic rates increases the risk of extinction for this DPS.

Climate change has the potential to affect sturgeon in similar, if not more significant ways it affects salmonids. Elevated air temperatures could lead to precipitation falling as rain instead of snow. Additionally, snow would likely melt sooner and more rapidly, potentially leading to greater flooding during melting and lower water levels at other times, as well as warmer river temperatures. Although sturgeon can spawn over varied benthic habitat, they prefer localized depressions in riverbeds (Erickson et al. 2001; Moyle et al. 1992; Moyle et al. 1995; Rien et al. 2001). Increased extremes in river flow (i.e., periods of flooding and low flow) can alternatively disrupt and fill in spawning habitat that sturgeon rely upon (ISAB 2007). If water flow is low during migration events, it is likely that new obstacles can impede or block sturgeon movement. As with other anadromous fishes, sturgeon are uniquely evolved to the environments that they live in. Because of this specificity, broad scale changes in environment can be difficult to adapt to, including changes in water temperature (Cech et al. 2000). Sturgeon are also sensitive to elevated water temperatures. Temperature triggers spawning behavior. Warmer water temperatures can initial spawning earlier in a season for salmon and the same can be true for sturgeon (ISAB 2007). If river and lake temperatures become anomalously warm, juvenile sturgeon may experience elevated mortality due to lack of cooler water refuges in freshwater habitats. Apart from direct changes to sturgeon survival, altered water temperatures may disrupt habitat, including the availability of prey (ISAB 2007). Warmer temperatures may also have the effect of increasing water use in agriculture, both for existing fields and the establishment of new ones in once unprofitable areas (ISAB 2007). This means that streams, rivers, and lakes will experience additional withdrawal of water for irrigation and increasing contaminant loads from returning effluent. Overall, it is likely that global warming will increase pressures on sturgeon survival and recovery.

Studies from other sturgeon species indicate that sturgeon readily bioaccumulate contaminants. White sturgeon from the Kootenai River have been found to contain aluminum, arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium, zinc, DDE, DDT, PCBs, and other organochlorines (Kruse and Scarnecchia 2001). Mercury has also been identified from white sturgeon of the lower Columbia River (Webb et al. 2006). Numerous organochlorines, including DDT, DDD, DDE, chlordane, and dieldrin have also been identified in these fish (Foster et al. 2001). Observed concentrations are likely sufficient to influence reproductive physiology.

Critical Habitat

On October 9, 2009, NMFS designated critical habitat for southern green sturgeon (74 FR 52300). The geographical area identified as critical habitat is based upon the overlapping distribution of the southern and northern DPS, and encompasses all areas where the presence of southern green sturgeon have been confirmed or where their presence is likely. Therefore the geographical area defined as critical habitat is the entire range of the biological species, green sturgeon, from the Bering Sea, AK, to Ensenada, Mexico. Specific fresh water areas include the Sacramento River, Feather River, Yuba River, and the Sacramento-San Joaquin Delta. Specific coastal bays and estuaries include estuaries from Elkhorn Slough, California, to Puget Sound, Washington. Coastal marine areas include waters along the entire biological species' range within a depth of 60 fathoms. The principle biological or physical constituent elements essential for the conservation of southern green sturgeon in fresh water include: food resources; substrate of sufficient type and size to support viable egg and larval development; water flow, water quality such that the chemical characteristics support normal behavior, growth and viability;

migratory corridors; water depth; and sediment quality. Primary constituent elements of estuarine habitat include food resources, water flow, water quality, migratory corridors, water depth, and sediment quality. The specific primary constituent elements of marine habitat include food resources, water quality, and migratory corridors.

Critical habitat of the Southern DPS of green sturgeon is threatened by several anthropogenic factors. Four dams and several other structures currently are impassible for green sturgeon to pass on the Sacramento, Feather, and San Joaquin rivers, preventing movement into spawning habitat. Threats to these riverine habitats also include increasing temperature, insufficient flow that may impair recruitment, the introduction of striped bass that may eat young sturgeon and compete for prey, and the presence of heavy metals and contaminants in the river.

Shortnose Sturgeon

Distribution and Description of the Listed Species

Shortnose sturgeon occur along the Atlantic Coast of North America, from the St. John River in Canada, south to the St. John's River in Florida. NMFS' recovery plan (1998a) recognized 19 wild populations based on their strong fidelity to their natal streams, and several captive populations (from a Savannah River broodstock) that are maintained for educational and research purposes (NMFS 1998a; Table 14).

Shortnose sturgeon are generally anadromous (they migrate between sea and fresh water for reproductive purposes) or amphidromous (some fish migrate between fresh and salt water for reasons other than spawning, such as feeding), but such migratory behavior may not be obligatory for the species as they can also maintain land-locked (freshwater resident) populations. In general, shortnose sturgeon are benthic fish that occupy the deep channel sections of large rivers or estuarine waters of their natal rivers, and will migrate considerable distances. Dadswell (1979 in Dadswell et al. 1984) observed shortnose sturgeon traveling up 160 km between tagging and recapture in the St. John estuary, and it is not uncommon for adults to migrate 200 km or more to reach spawning areas (Kynard 1997).

The general migratory strategy of shortnose sturgeon is similar to many fresh water and diadromous fishes, which probably optimizes feeding opportunities, minimizes losses due to unfavorable conditions (winter refuge migrations), and optimizes spawning success (Northcote 1978; Harden-Jones 1968 in Dadswell 1984). Water temperatures, flow regimes, and barriers influence their movement patterns (Kynard 1997; Kynard et al. 2000). Adult shortnose sturgeon will migrate upstream to spawning areas in the spring or in the fall. Fish that migrate upstream in the fall generally overwinter in areas just downstream of spawning sites, while others including non-spawners will overwinter in estuarine waters. After spawning in the spring, spent (post-spawned) adults tend to migrate rapidly downstream to feeding areas in the estuary or to tidally influence fresh water (see Dadswell et al. 1984 for a review).

Young-of-the year shortnose sturgeon move downstream after hatching, remaining in fresh water for about 1 year (Kynard 1997). Initially, young shortnose sturgeon will reside short distances from spawning areas, and as they grow will tend to move further downstream (Dadswell et al.

1984). By age 3 or older juvenile sturgeon will spend a large portion of their year at the salt- and fresh water interface of coastal rivers (NMFS 1998a).

Habitat use in fresh water during summer and winter months overlaps between adult and age-1 shortnose sturgeon (O'Herron et al. 1993; Rogers and Weber 1995 in Moser et al. 2000; Kynard et al. 2000). Kynard et al. (2000) found that both age classes preferred deep-water curves with sand and cobble to higher velocity runs, particularly during winter months, and shifted to channel habitat as water temperatures rose in summer months. Many fish also exhibited diel movement patterns between deeper waters during the day and shallower waters at night (Kynard et al. 2000). During the summer, at the southern end of their range, shortnose sturgeon congregate in cool, deep, areas of rivers where adult and juvenile sturgeon can take refuge from high temperatures (Flournoy et al. 1992, Rogers and Weber 1995, and Weber 1996 cited in Moser et al. 2000). In the Connecticut River and the Merrimack, Kynard et al. (2000) found shortnose generally used water about 3 meters deep, ranging from less than a meter to about 15 meters deep.

Sturgeon are iteroparous, and based on limited data it appears that females sturgeon spawn every three to five years while males spawn every other year, although some may spawn in consecutive years (Dovel et al. 1992; Collins and Smith 1993; Kieffer and Kynard 1993; NMFS 1998a). Spawning typically occurs during the spring, between mid-March and late May. Spawning areas are often located just below the fall line at the farthest accessible upstream reach of the river (NMFS 1998a). The onset of spawning may be cued to decreasing river discharge following the peak spring freshet, when water temperatures range from 8 to 12 °C and bottom water velocities range between 25-130 cm/s, although photoperiod appears to control spawning readiness (Dadswell et al. 1984; NMFS 1998a; Kynard et al., in draft).

Length at maturity is about 45-55 cm fork length for shortnose sturgeon and age at first spawning appears to vary along a latitudinal cline. According to spawning checks, it appears that male shortnose sturgeon in southern rivers will first spawn between ages 2 and 5, while fish as far north as the St. Johns River, Canada first spawn at about 10 to 11 years of age (Dadswell et al. 1984; NMFS 1998a). Age at first spawning for female shortnose sturgeon varies from about age 6 to 18 years, like males, varying on a latitudinal cline (Dadswell et al. 1984; NMFS 1998a). In general, fish in the northern portion of the species' range live longer than individuals in the southern portion of the species' range (Gilbert 1989). The maximum age reported for a shortnose sturgeon in the St. John River in New Brunswick is 67 years (for a female), 40 years for the Kennebec River, 37 years for the Hudson River, 34 years in the Connecticut River, 20 years in the Pee Dee River, and 10 years in the Altamaha River (Gilbert 1989 using data presented in Dadswell et al. 1984). Male shortnose sturgeon appear to have shorter life spans than females (Gilbert 1989).

Like all sturgeon, shortnose have ventrally located, sucker-like mouths, structured for feeding on benthos. Foraging generally occurs in areas with abundant macrophytes, where juvenile and adult shortnose sturgeon feed on amphipods, polychaetes, and gasteropods (Dadswell et al. 1984; Moser and Ross 1995; NMFS 1998a). Starting as larvae sturgeon use electroreception to identify prey. Olfaction and taste are also likely important to foraging, while vision is thought to play a minor role (Miller 2004). As adults, a significant portion of a shortnose sturgeon's diet may consist of freshwater mollusks (Dadswell et al. 1984). Based on observations by Kynard et al.

(2000), shortnose sturgeon will consume the entire mollusk, excreting the shell after ingestion.

Table 14. Shortnose sturgeon populations and their estimated abundances

Population (Location)[a]	Data Series	Abundance Estimate (C.I.)[b]	Population Segment	Reference
Saint John River (Canada)	1973-1977	18,000 (+/-30%)	Adults	Dadswell 1979
Kennebecasis River (Canada)	1998-2005	2,068 (801-11,277)		COSEWIC 2005
Kennebecasis River	2005	4,836 (+/-69)		Li et al. 2007, NMFS unpubl.
Penobscot River (ME)	2006-2007	1,049 (673-6,939)		UME 2008
	2008	1739 (846-3653)	Summer	P. Dionne, pers. comm..
		667 (451-1013)	Fall	P. Dionne, pers. comm..
Kennebec River (ME)	1977-1981	7,222 (5,046-10,765)	Adult	Squiers et al. 1982
	2003	9,488 (6,942-13,358)	Adults	Squiers 2003
Merrimack River (MA)	1987-1991	32 (20-79)	Adults	Kynard & Kieffer, unpubl.; NMFS unpubl.
Connecticut River (MA, CT)	1989-2002	1,042-1,580 [c]	Adults	Savoy 2004
Upper Connecticut River[d]	1976-1977	516 (317–898)	Total	Taubert 1980; NMFS 1998a
	1977-1978	370 (235–623)	Total	Taubert 1980; NMFS 1998a
	1976-1978	714 (280-2,856)	Total	Taubert 1980; NMFS 1998a
	1976-1978	297 (267–618)	Total	Taubert 1980; NMFS 1998a
	1994	328 (188-1,264)	Adults	Kynard & Kieffer, unpubl.; NMFS unpubl.
	1994-2001	143 (14-360)	Spawning Adults	Kynard & Kieffer, unpubl.; NMFS unpubl.
Lower Connecticut River[e]	1988-1993	895 (799-1,018)	Adult	Savoy and Shake 1992; NMFS 1998a
Hudson River (NY)	1980	30,311	Total	Dovel 1979; NMFS 1998a
	1994-1997	61,057 (52,898-72,191)	Total	Bain et al. 2007
Delaware River (NJ, DE, PA)	1981-1984	12,796 (10,288-16,267)	Partial	Hastings et al. 1987
	1981-1984	14,080 (10,079-20,378)	Partial	Hastings et al. 1987
	1999-2003	12,047 (10,757-13,589)		Brundage and O'Herron 2003
Chesapeake Bay (MD, VA)				
Cape Fear River (NC)				
Winyah Bay (NC, SC)				
Santee River (SC)				
Cooper River (SC)	1996-1998	300	Adults	Cooke et al. 2004
ACE Basin (SC)				
Savannah River (SC, GA)		1,000 - 3,000	Adults	B Post, SCDNR 2003; NMFS unpubl.
Ogeechee River (GA)	1993	266 (236 – 300)		Weber 1996, 1998
	1993	361 (326 – 400)	Total	Rogers and Weber 1994, NMFS 1998a
	1999-2004	147 (104-249)		Fleming et al. 2003; NMFS unpubl.
Altamaha River (GA)	1988	2,862 (1,069 - 4,226)	Total	NMFS 1998a
	1990	798 (645 - 1,045)	Total	NMFS 1998a

	1993	468 (316 – 903)	Total	NMFS 1998a
		6,320 (4,387-9,249)	Total	DeVries 2006
Satilla River (GA)				
Saint Mary's River (FL)				
Saint Johns River (FL)				FFWCC 2007c

[a]The original 19 populations identified by NMFS in the 1998 recovery plan are left aligned in this column. Estimates for a tributary or river segment are indented.
[b]Population estimates are established using different techniques and should be viewed with caution. In some cases, sampling biases may have violated the assumptions of the procedures used or resulted in inadequate representation of a population segment. Some estimates (e.g., those without confidence intervals or are depicted by ranges only) are the "best professional judgment" of researchers based on their sampling effort and success.
[c]Range represents total population estimates using four different techniques. All techniques suggest the population increased during the sampling period (see Savoy 2004 for more details).
[d]Above Holyoke Dam.
[e]Below Holyoke Dam.

Status and Trends

Shortnose sturgeon were listed as endangered on March 11, 1967, under the Endangered Species Preservation Act (32 FR 4001) and remained on the endangered species list with enactment of the ESA of 1973, as amended. Although the original listing notice did not cite reasons for listing the species, a 1973 Resource Publication issued by the U.S. Department of Interior (USDOI), stated that shortnose sturgeon were "in peril ... gone in most of the rivers of its former range [but] probably not as yet extinct" (USDOI 1973). Pollution and overfishing, including bycatch in the shad fishery, were listed as principal reasons for the species' decline. Shortnose sturgeon are listed as an endangered species throughout all of its range

Northern shortnose sturgeon population abundances are generally larger than southern populations (Kynard 1997). Updated population estimates also suggest that three of the largest populations (Kennebec, Hudson, and Delaware River) may be increasing or stable, although data is limited. The New York (Hudson River) shortnose sturgeon population is the largest extant population of this species and based on available data exhibits appears to have increased (NMFS 1998a; Bain *et al.* 2000). The most recent population estimate indicates this population consists of about 61,000-shortnose sturgeon (95% confidence interval [CI] was between 52,898 and 72,191 fish [Bain et al. 2000]). A comparison of the Bain estimate to the 1979/1980 population estimate of spawning adults by Dovel et al. (1992; about 13,000 fish) led Bain et al. (2000) to conclude that the population had made a dramatic increase (about 400 % increase) between 1979 and 1997. While still evidence of an increasing population, a comparison of total population estimates (30,000:60,000) would suggest the population has only doubled in size during the study years. Similarly, the Kennebec River population appears to be increasing. Early estimates suggest that the Kennebec River contained an estimated 7,200 adult shortnose sturgeon in 1977-81 (Squiers et al. 1982), while the most recent estimate for this population is about 9,500 fish (Squiers 2003), suggesting the population has increased by about 30 % in about a twenty year period.

Data from the Delaware River, suggests that the population may be stable. Brundage and O'Herron (2003) estimate that the current population for the Delaware River is 12,047 adult fish (1999-2003; 95% CI: 10,757-13,589), which is similar to the 1981/84 estimate by Hastings *et al.* (1987) of 12,796 fish (95% CI: 10,288-16367). The recent capture of several fish that were tagged as adults by Hastings et al. (1987) suggests that older fish may comprise a substantial portion of the Delaware River population. Based on studies from other sturgeon species we know of no evidence of senescence in sturgeon, and we would expect that these fish are reproductively active (Paramian et al. 2005). Despite their longevity, the viability of sturgeon populations is sensitive to variability in juvenile recruitment and survival (Anders et al. 2002; Gross et al. 2002; Secor et al. 2002). Although interannual variation in juvenile recruitment would be expected as a result of stochastic factors that influence spawning and egg/larval survival, if the mean population size does not change over the long-term it then it would appear there is sufficient juvenile survival to provide at least periodic recruitment into the adult age classes. Data on juvenile recruitment or age-1+ survival would, however, establish whether this population is at a stable equilibrium.

South of Chesapeake Bay, populations are relatively small compared to their northern counterparts. The largest of the southern populations of shortnose sturgeon is the Altamaha River population. Population estimates have been calculated several times for sturgeon in the Altamaha since 1993, and s. Total population estimates shown pretty sizeable interannual variation is occurring; estimates have ranged from as low as 468 fish in 1993 to over 6,300 fish in 2006 (NMFS 1998a; DeVries 2006). The Ogeechee River is the next most studied river south of Chesapeake Bay, and abundance estimates indicate that the shortnose sturgeon population in this river is considerably smaller than that in the Altamaha River. The highest point estimate in 1993 using a modified Schnabel technique resulted in a total population estimate of 361 shortnose sturgeon (95% CI: 326-400). In contrast the most recent survey resulted in an estimate of 147 shortnose sturgeon (95% CI: 104-249), suggesting that the population may be declining.

Annual variation in population estimates in many basins is due to changes in yearly capture rates, which are strongly correlated with weather conditions (river flow and water temperatures). In "dry years" fish move into deep holes upriver of the saltwater/freshwater interface, which can make them more susceptible to gillnet sampling. Consequently, rivers with limited data sets among years and limited sampling periods within a year may not offer a realistic representation of the size or trend of the shortnose sturgeon population in the basin. As a whole, the data on shortnose sturgeon populations is rather limited and some of the differences observed between years may be an artifact of the models and assumptions used by the various studies. Long-term data sets and an open population model would likely provide for more accurate population estimates across the species' range, and could provide the opportunity to more closely link strong-year classes to habitat conditions.

Throughout the species' range there are other extant populations, or at least evidence that several other basins are used periodically. That is, shortnose sturgeon have been documented in the St. John's River (FL), the St. Mary's River, Chesapeake Bay, Potomac River, Piscataqua River, the Housatonic River, and others. Some basins probably previously contained shortnose populations, but recent sampling has been largely unsuccessful. Despite the occasional observations of shortnose sturgeon, populations may be extinct in several basins (e.g., St. John's (FL), St. Mary's, Potomac, Housatonic, and Neuse rivers). Those few fish that have been observed in these basins are generally presumed to be immigrants from neighboring basins. In some cases, (e.g. Chesapeake Bay) migratory information collected from tagged fish and genetic evidence confirms that fish captured in Chesapeake Bay were part of the Delaware River population (Grunwald et al. 2002; Wirgin et al. 2005; and T. King, in progress)..

Threats

Natural Threats. Yellow perch, sharks, and seals are predators of shortnose sturgeon juveniles (NMFS 1998a). The effects of disease and parasites are generally unknown.

Anthropogenic Threats. Shortnose sturgeon have declined from the combined effects from the construction of hydropower and water diversion projects, dredging and blasting, water pollution, fisheries, and hatcheries. The construction of dams has resulted in substantial loss of shortnose sturgeon habitat along the Atlantic seaboard. In many cases dams divide shortnose sturgeon spawning habitat (e.g., Connecticut River, Penobscot River) and impede passage or block it completely. Where it has occurred, remediation measures, such as obstruction removal or modification to allow for fish passage have improved shortnose sturgeon habitat and likely improved productivity and more such modifications are planned in certain basins. For instance, with the breaching of the Bangor Dam in the Penobscot River in 1977 five river kilometers were opened to sturgeon and other anadromous fishes. With the recent signing of the Penobscot River Restoration Trust, access may be restored to another 29 km of habitat.

Historic fishery harvests, as well as the incidental harvest in current fisheries, have had lasting effects on shortnose sturgeon. In the late nineteenth and early twentieth centuries shortnose sturgeon commonly were harvested incidental to Atlantic sturgeon, the larger and more commercially valuable of these two sympatric sturgeon species (NMFS 1998a). The effects of these harvests may have latent and long-lasting impacts on some populations. At present there is no legal directed fishing effort for shortnose sturgeon in the United States, although some illegal poaching is suspected. Additionally, shortnose sturgeon are often caught incidental to other fisheries. For instance, shortnose are caught incidentally by bass anglers, and incidentally to alewife/gaspereau and shad fisheries in the St. John's River in Canada, shad fisheries in the Altamaha River, Hudson River, and others (COSEWIC 2005; Bahn & Peterson 2009).

Habitat alterations from discharges, dredging or disposal of material into waterways, and other developmental activities along riverine and estuarine systems threaten shortnose sturgeon habitat. Periodic maintenance of harbors and rivers likely results in the direct take of some sturgeon, but perhaps of greater impact is the manner in which dredging alters benthic topography and community structure, and water quality (increase in suspended sediments). Shoreline development of liquefied natural gas facilities and alternative power sources also alters coastal habitats through changes in benthic communities by dredging, changes in water quality and water temperatures, and may increase the potential of ship strikes. In the Bay of Fundy, a tidal turbine killed at least three Atlantic salmon in the 1980s, and may be a threat to shortnose sturgeon as well (Dadswell and Rulifson 1994). Although currently the only example of this type of turbine in North America, increasing interests in finding alternative energy sources is expected to result in an increase in the number of marine turbines along the coast.

Fish kills have also been observed where estuaries are affected by urban and agricultural discharges that cause vegetative blooms and eutrophic conditions. Extreme declines in dissolved oxygen levels have occurred periodically throughout the species' range. In the late 1960s and early 1970s, dissolved oxygen levels reached zero ppm in the Penobscot, Kennebec, and Androscoggin rivers and estuaries during the summer. Extreme low dissolved oxygen levels have also plagued Chesapeake Bay. In most cases, dissolved oxygen levels have improved through improved treatment and control of waste discharges in the past twenty years, but degraded conditions of benthos are still common in many estuaries throughout the species' range as a result of this historic loading of organic materials, waste, and legacy toxins such as dioxin. As a result, shortnose sturgeon and other benthic organisms are regularly in direct contact with legacy pollutants, as well as a suite of common contaminants added from more current industrial and agricultural practices. Studies demonstrate that shortnose sturgeon carry a wide number of potentially hazardous contaminants. Individuals from the Delaware River contain numerous metals (mercury, aluminum, antimony, barium, cadmium, calcium, chromium, copper, iron, magnesium, manganese, nickel, potassium, sodium, vanadium, and zinc), PCDDs, PCDFs, PCBs, DDE, DDD, bis (2-ethylhexyl) phthalate, di-n-butylphthalate, and chlordane (ERC 2002). Most of these metals, PCDDs, PCDFs, and PCBs were also found in shortnose sturgeon in the Kennebec River (ERC 2003).

Climate change has the potential to affect sturgeon in similar, if not more significant, ways than it affects salmonids. Elevated air temperatures could lead to precipitation falling as rain instead of snow. Additionally, snow would likely melt sooner and more rapidly, potentially leading to greater flooding during melting and lower water levels at other times, as well as warmer river temperatures (ISAB 2007). Although sturgeon can spawn over varied benthic habitat, they prefer localized depressions in riverbeds (Erickson et al. 2001; Moyle et al. 1992; Moyle et al. 1995; Rien et al. 2001). Increased extremes in river flow (i.e., periods of flooding and low flow) can alternatively disrupt and fill in spawning habitat that sturgeon rely upon (ISAB 2007). If water flow is low during migration events, it is likely that new obstacles can impede or block sturgeon movement. As with other anadromous fishes, sturgeon are uniquely evolved to the environments that they live in. Because of this specificity, broad scale changes in environment can be difficult to adapt to, including changes in water temperature (Cech et al. 2000). Sturgeon are also directly sensitive to elevated water temperatures. Temperature triggers spawning behavior. Warmer water temperatures can initiate spawning earlier in a season for salmon and the same can be true for sturgeon (ISAB 2007). If river and lake temperatures become anomalously warm, juvenile sturgeon may experience elevated mortality due to lack of cooler water refuges in freshwater habitats. Apart from direct changes to sturgeon survival, altered water temperatures may disrupt habitat, including the availability of prey (ISAB 2007). Warmer temperatures may also have the effect of increasing water use in agriculture, both for existing fields and the establishment of new ones in once unprofitable areas (ISAB 2007). This means that streams, rivers, and lakes will experience additional withdrawal of water for irrigation and increasing contaminant loads from returning effluent. Overall, it is likely that global warming will increase pressures on sturgeon survival and recovery.

Critical Habitat

NMFS has not designated critical habitat for shortnose sturgeon.

Sockeye Salmon

Description of the Species

Sockeye salmon are the second most abundant of the seven Pacific salmon species, and occur in the North Pacific and Arctic oceans and associated freshwater systems. This species' ranges south as far as the Sacramento River in California and northern Hokkaido in Japan, to as far north as far as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia (Burgner 1991). The largest populations, and hence the most important commercial populations, occur north of the Columbia River

Sockeye salmon exhibit a very diverse life history, characteristically using both riverine and lake habitat throughout their range, exhibiting both freshwater resident and anadromous forms. The vast majority of sockeye salmon are anadromous fish that make use of lacustrine habitat for juvenile rearing. These “lake-type” fish typically spawn in the outlet streams of lakes and occasionally in the lakes themselves. Juvenile sockeye salmon will then use the lake environment for rearing for up to 3 years before migrating to sea. After 1 to 4 years at sea, sockeye salmon will return to their natal lake to spawn. Some sockeye, however, spawn in rivers without lake habitat for juvenile rearing. Offspring of these riverine spawners tend to use the lower velocity sections of rivers as the juvenile rearing environment for 1 to 2 years, or may migrate to sea in their first year.

Sockeye salmon also have a wholly freshwater life history form, called kokanee (Burgner 1991). In some cases a single population will give rise to both the anadromous and freshwater life history form. While in fresh water juveniles of both life history types prey primarily upon insects. The presence of both life history types may be related to the energetic costs of outmigrating to sea, and the productivity of the lacustrine system they inhabit. In coastal lakes, where the migration to sea is relatively short and energetic costs are minimal, kokanee populations are rare.

Once smolts enter the Pacific Ocean, they distribute widely across the North Pacific, generally above 40ºN where a current boundary is located. Season, temperature, salinity, life stage, age, size, availability of prey and population-of-origin are all factors that influence offshore movements (Burgner 1991). Sockeye tend to stay within several dozen feet of the surface, although they tend to be closer to the surface at night versus daytime (Manzer 1964; French et al. 1976). However, they may migrate several thousand miles in search of prey and are considered to travel continuously (Royce et al. 1968). While at sea, sockeye prey upon a variety of organisms, including small fish (capelin, lantern fish, cod, sand lance, herring, and pollock), squid, crustacean larvae, krill, and other invertebrates (Foerster 1968; French et al. 1976; Wing 1977). Thermoclines may also influence vertical distribution, with fish only mingling between surface and deeper waters when the boundary temperature difference is weak. Sockeye appear to prefer cooler waters relative to other salmon species, but younger salmon may prefer warmer sea surface temperatures (39 to 50º F) than larger, older fish (37 to 41º F), possibly an artifact of older fish being distributed further north. Adult upstream migration may be blocked by temperatures above 70º F (McCullough 1999). However, temperatures below 70º F can stress fish by increasing their susceptibility to disease and elevating their metabolism (Brett 1979; Berman 1990). Maturation and timing of return to spawn by sockeye appears to be linked to water temperature, with gonad development increasing in late May through early July (Nishiyama 1984).

Spawning generally occurs in late summer and autumn, but the precise time can vary greatly among populations. Age at maturity varies by region from 2 to 5 years, but is generally 2 to 4 years in Washington State (Burgner 1991). Males often arrive earlier than females on the spawning grounds, and will persist longer during the spawning period. Average fecundity ranges from about 2,000 to 2,400 eggs per female to 5,000 eggs, depending upon the population and average age of the female. Fecundity in kokanee is much lower and may range from about 300 to less than 2,000 eggs.

Incubation is a function of water temperatures, but generally lasts between 100 and roughly 200 days (Burgner 1991). After emergence, fry move rapidly downstream or upstream along the banks to the lake rearing area. Fry emerging from lakeshore or island spawning grounds may simply move along the shoreline of the lake (Burgner 1991).

Ozette Lake Sockeye Salmon

Distribution and Description of the Listed Species

This ESU includes all naturally spawned sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake, Washington. Composed of only one population, the Ozette Lake sockeye salmon ESU consists of five spawning aggregations or subpopulations which are grouped according to their spawning locations. The five spawning locations are Umbrella and Crooked creeks, Big River, and Olsen’s and Allen’s beaches (NMFS 2009).

Adult Ozette Lake sockeye salmon enter Ozette Lake through the Ozette River from mid-April to mid-August, holding three to nine months in Ozette Lake prior to spawning in late October through January. Sockeye salmon spawn primarily in lakeshore upwelling areas in Ozette Lake (particularly at Allen's Bay and Olsen's Beach), and in two tributaries Umbrella Creek and Big River. Minor spawning may occur below Ozette Lake in the Ozette River or in Coal Creek, a tributary of the Ozette River. Beach spawners are almost all age-4 adults, while tributary spawners are ages 3 and 5 (Haggerty et al. 2009 in NMFS 2009). Spawning occurs in the fall through early winter, with peak spawning in tributaries in November and December. Eggs and alevins remain in the gravel until the fish emerge as fry in spring. Fry then migrate immediately to the limnetic zone in Ozette Lake, where the fish rear. After one year of rearing, in late spring, Ozette Lake sockeye salmon emigrate seaward as age-1+ smolts, where they spend between 1 and 3 years in ocean before returning to fresh water.

Status and Trends

NMFS originally listed Ozette Lake sockeye salmon ESU as a threatened species in 1999 (64 FR 14528). This classification was retained on June 28, 2005 (70 FR 37160). This ESU includes all naturally spawned populations of sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake, Washington. Two artificial propagation programs are considered part of this ESU: The Umbrella Creek and Big River sockeye salmon hatchery programs. NMFS considers these artificially propagated populations no more divergent relative to the local natural population than would be expected between closely related natural populations (70 FR 37160).

The historical abundance of Ozette Lake sockeye salmon is poorly documented, but may have been as high as 50,000 individuals (Blum 1988). The overall abundance of naturally–produced Ozette Lake sockeye salmon is believed to have declined substantially from historical levels. In the first study of lake escapement of Ozette Lake sockeye salmon (Kemmerich 1945), the run size entering the lake was estimated at a level of several thousand fish. These counts appear to be roughly double the current mean lake abundance, considering that they were likely conducted upstream from fisheries in or near to the Ozette River. Makah Fisheries Management (MFM 2000 *in* Good et al. 2005) concluded that there appears to be a substantial decline in the Tribal catch of Ozette Lake sockeye salmon beginning in the 1950s and a similar decline in the run size since the 1920s weir counts reported by Kemmerich (1945).

An analysis of total annual Ozette Lake sockeye salmon abundance (based on adult run size data presented in Jacobs et al. 1996) indicates a trend in abundance averaging -2% per year over the period 1977 through 1998 (NMFS 1998b). The current tributary-based hatchery program was planned and initiated in response to the declining population trend identified for the Ozette Lake sockeye salmon population. The most recent (1996 to 2003) run-size estimates range from a low of 1,609 in 1997 to a high of 5,075 in 2003, averaging approximately 3,600 sockeye per year (NMFS 2009). For return years 2000 to 2003, the 4-year average abundance estimate was slightly over 4,600 sockeye. Because run-size estimates before 1998 are likely to be even more unreliable than recent counts, and new counting technology has resulted in an increase in estimated run sizes, no statistical estimation of trends is reported. The current trends in abundance are unknown for the beach spawning aggregations. Although overall abundance appears to have declined from historical levels, whether this resulted in fewer spawning aggregations, lower abundances at each aggregation, or both, is not known (Good et al. 2005). Based on estimates of habitat carrying capacity, a viable sockeye salmon population in Lake Ozette watershed would range between 35,500 to 121,000 spawners (Rawson et al. 2008 in NMFS 2009).

There has been no harvest of Ozette Lake sockeye salmon for the past four brood-cycle years (since 1982). Prior to that time, ceremonial and subsistence harvests by the Makah Tribe were low, ranging from 0 to 84 fish per year. Harvest has not been an important mortality factor for the population in over 35 years. In addition, due to the early river entry timing of returning Ozette Lake sockeye salmon (beginning in late April, with the peak returns prior to late-May to mid-June), the fish are not intercepted in Canadian and United States marine area fisheries directed at Fraser River sockeye salmon. There are currently no known marine area harvest impacts on Ozette Lake sockeye salmon.

Overall abundance is substantially below historical levels (Good et al. 2005). Declines in abundance have been attributed to a combination of introduced species, predation, loss of tributary populations, a loss of quality of beach spawning habitat, temporarily unfavorable ocean conditions, habitat degradation, and excessive historical harvests (Jacobs et al. 1996). In the last few years the number of returning adults has increased, although some of these individuals are of hatchery origin. This produces uncertainty regarding natural growth rate and productivity of the ESU’s natural component. In addition, genetic integrity has perhaps been compromised due to the artificial supplementation that has occurred in this population, since approximately one million sockeye have been released into the Ozette watershed from the late 1930s to present (Kemmerich 1945; Boomer 1995).

Critical Habitat

On September 2, 2005, NMFS designated critical habitat for the Ozette Lake sockeye salmon ESU (70 FR 52630). The specific geographic areas designated as critical are the Hoh/Quillayute Subbasin, Ozette Lake and the Ozette Lake watershed, and include: the Ozette River upstream to endpoints in Big River, Coal Creek, East Branch Umbrella Creek, the North and South Fork of Crooked Creek and several other tributaries. The specific primary constituent elements identified for Lake Ozette sockeye salmon are areas for spawning, freshwater rearing and migration, estuarine areas free of obstruction, nearshore marine areas free of obstructions, and offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, and adequate passage conditions. Only one watershed supports this ESU, and it is rated as having a high conservation value. This watershed is essential to the species' overall conservation by protecting quality growth, reproduction, and feeding.

Snake River Sockeye Salmon

Distribution and Description of the Listed Species

Snake River sockeye salmon are unique compared to other sockeye salmon populations: it spawns at a higher elevation (6,500 feet) and a longer freshwater migration (approximately 900 miles) than any other sockeye salmon population in the world. Sockeye salmon in this ESU spawn in Redfish Lake in Idaho's Stanley Basin (Bjornn et al. 1968; Foerster 1968). Stanley Basin sockeye salmon are separated by 700 or more river miles from two other extant upper Columbia River populations in the Wenatchee River and Okanogan River drainages. These latter populations return to lakes at substantially lower elevations (Wenatchee at 1,870 feet and Okanagon at 912 feet) and occupy different ecoregions. The Snake River sockeye salmon ESU includes all anadromous and residual sockeye salmon from the Snake River basin of Idaho, as well as hatchery individuals from the Redfish Lake Captive Broodstock Program.

Status and Trends

Snake River sockeye salmon were originally listed as endangered in 1991 and retained that classification when their status was reviewed on June 28, 2005 (70 FR 37160). The only extant sockeye salmon population in the Snake River basin at the time of listing was that in Redfish Lake, in the Stanley Basin (upper Salmon River drainage) of Idaho. Other lakes in the Snake River basin historically supported sockeye salmon populations, including Wallowa Lake (Grande Ronde River drainage, Oregon), Payette Lake (Payette River drainage, Idaho) and Warm Lake (South Fork Salmon River drainage, Idaho; Waples et al. 1997). These populations are now considered extinct. Although kokanee, a resident form of *O. nerka*, occur in numerous lakes in the Snake River basin, other lakes in the Stanley Basin, and sympatrically with sockeye in Redfish Lake, resident *O. nerka* were not considered part of the species at the time of listing (1991). Subsequent to the 1991 listing, a residual form of sockeye residing in Redfish Lake was identified. The residuals are non-anadromous, completing their entire life cycle in fresh water, but spawn at the same time and in the same location as anadromous sockeye salmon. In 1993, NMFS determined that residual sockeye salmon in Redfish Lake were part of the Snake River sockeye salmon. Also, artificially propagated sockeye salmon from the Redfish Lake Captive

Propagation program are considered part of this species (70 FR 37160; June 28, 2005).

NMFS has determined that this artificially propagated population is genetically no more than moderately divergent from the natural population (NMFS 2005a). Five lakes in the Stanley Basin historically contained sockeye salmon: Alturas, Pettit, Redfish, Stanley and Yellowbelly (Bjornn et al. 1968). It is generally believed that adults were prevented from returning to the Sawtooth Valley from 1910 to 1934 by Sunbeam Dam. Sunbeam Dam was constructed on the Salmon River approximately 20 miles downstream of Redfish Lake. Whether Sunbeam Dam was a complete barrier to adult migration remains unknown. It has been hypothesized that some passage occurred while the dam was in place, allowing the Stanley Basin population or populations to persist (Bjornn et al. 1968; Waples et al. 1991).

Adult returns to Redfish Lake during the period 1954 through 1966 ranged from 11 to 4,361 fish (Bjornn et al. 1968). Sockeye salmon in Alturas Lake were extirpated in the early 1900s as a result of irrigation diversions, although residual sockeye may still exist in the lake (Chapman and Witty 1993). From 1955 to 1965, the Idaho Department of Fish and Game eradicated sockeye salmon from Pettit, Stanley, and Yellowbelly lakes, and built permanent structures on each of the lake outlets that prevented re-entry of anadromous sockeye salmon (Chapman and Witty 1993). In 1985, 1986, and 1987, 11, 29, and 16 sockeye, respectively, were counted at the Redfish Lake weir (Good et al. 2005). Only 18 natural origin sockeye salmon have returned to the Stanley Basin since 1987. During the fall of 1990, during the course of NMFS' first status review on the species, no fish were observed at Lower Granit Dam or entering the lake and only one fish was observed in each of the two previous years. The first adult returns from the captive broodstock program returned to the Stanley Basin in 1999. From 1999 through 2005, a total of 345 captive brood program adults that had migrated to the ocean returned to the Stanley Basin.

Recent annual abundances of natural origin sockeye salmon in the Stanley Basin have been extremely low. No natural origin anadromous adults have returned since 1998 and the abundance of residual sockeye salmon in Redfish Lake is unknown. This species is entirely supported by adults produced through the captive propagation program at the present time. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely greater than 0.3% (Hebdon et al. 2004). The status of this ESU is extremely precarious, such that there was unanimous consent among the biological review team members that the species remains in danger of extinction (Good et al. 2005).

Critical Habitat

Critical habitat for these salmon was designated on December 28, 1993 (58 FR 68543), and encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to the confluence of the Salmon River; all

Salmon River reaches to Alturas Lake Creek; Stanley, Redfish, yellow Belly, Pettit, and Alturas Lakes (including their inlet and outlet creeks); Alturas Lake Creek and that portion of Valley Creek between Stanley Lake Creek and the Salmon River. Critical habitat also includes all river lakes and reaches presently or historically accessible to Snake River sockeye salmon. These habitats are critical for the conservation of the species because it provides spawning and juvenile rearing habitat, areas for juvenile growth and development, and migration corridors for smolts to the ocean and adults to spawning habitat from the Pacific Ocean. Limiting factors identified for Snake River sockeye include: reduced tributary stream flow, impaired tributary passage and blocks to migration, and mainstem Columbia River hydropower system mortality.

Steelhead

Description of the Species

Steelhead, the common name of the anadromous form of *O. mykiss*, are native to Pacific Coast streams extending from Alaska south to northwestern Mexico (Moyle 1976; Stolz & Schnell 1991; NMFS 1997b). The life history of this species varies considerably throughout its range. Generally, steelhead can into two races: the stream-maturing type, summer steelhead, enters fresh water in a sexually immature condition and requires several months in fresh water to mature and spawn; and the ocean-maturing type, winter steelhead, enters fresh water with well-developed gonads and spawns shortly after river entry. Variations in migration timing exist between populations, and some river basins have both summer and winter steelhead, while others only have race.

Summer steelhead enter fresh water between May and October in the Pacific Northwest (Nickelson et al. 1992; Busby et al. 1996). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelson et al. 1992). Summer steelhead migrate inland toward spawning areas, overwinter in the larger rivers, resume migration in early spring to natal streams, and then spawn in January and February (Barnhart 1986; Meehan and Bjornn 1991; Nickelson et al. 1992). Winter steelhead enter fresh water between November and April in the Pacific Northwest (Nickelson et al. 1992; Busby et al. 1996), migrate to spawning areas, and then spawn, generally in April and May (Barnhart 1986). Some adults, however, do not enter some coastal streams until spring, just before spawning (Meehan and Bjornn 1991).

There is a high degree of overlap in spawn timing between populations regardless of run type (Busby et al. 1996). Difficult field conditions at that time of year and the remoteness of spawning grounds contribute to the relative lack of specific information on steelhead spawning. Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death, although steelhead rarely spawn more than twice before dying; most that do spawn more than twice tend to be female (Nickelson et al. 1992; Busby et al. 1996). Second time spawners often make up about 70 to 85 % of repeat spawners, with third time spawners make up between 10 to 25 % of repeats (Stolz & Schnell 1991). Iteroparity is more common among southern steelhead populations than northern populations (Busby et al. 1996).

As with other salmonids, the larger the fish the more eggs produced. Egg and hatching success are related to the conditions within the redd, and time to hatching is temperature dependent.

Fertilization to hatching is generally less than a month, after which newly hatched fish will remain in the redd for another 2-3 weeks. In late spring, and following yolk sac absorption, alevins emerge from the gravel and begin actively feeding. After emerging from the gravel, fry usually inhabit shallow water along banks of perennial streams. Fry occupy stream margins (Nickelson et al. 1992). Summer rearing takes place primarily in the faster parts of pools, although young-of-the-year are abundant in glides and riffles. Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelson et al. 1992).

Juvenile steelhead migrate little during their first summer and occupy a range of habitats featuring moderate to high water velocity and variable depths (Bisson et al. 1988). Steelhead hold territories close to the substratum where flows are lower and sometimes counter to the main stream; from these, they can make forays up into surface currents to take drifting food (Kalleberg 1958). Juveniles rear in fresh water from 1 to 4 years, then smolt and migrate to the ocean in March and April (Barnhart 1986). Winter steelhead juveniles generally smolt after 2 years in fresh water (Busby et al. 1996). Juvenile steelhead tend to migrate directly offshore during their first summer from whatever point they enter the ocean rather than migrating along the coastal belt as salmon do. Steelhead typically reside in marine waters for 2 or 3 years prior to returning to their natal stream to spawn as 4- or 5-year olds; fish in the northern portion of the range may spend more time rearing in marine waters (Stolz & Schnell 1991). Juveniles feed primarily on insects (chironomids, baetid mayflies, and hydropsychid caddisflies; Merz 1994). Adults feed on aquatic and terrestrial insects, mollusks, crustaceans, fish eggs, minnows, and other small fishes (including greenling and other trout; Chapman and Bjornn 1969; Stolz & Schnell 1991).

Threats

Natural Threats. Steelhead, like other salmon, are exposed to high rates of natural predation each stage of their life stage. The highest mortality occurs between the egg stage and smolt outmigration, and is highest in the first few months following emergence from the redd (Stolz & Schnell 1991). In fresh water, fry fall prey to older steelhead and other trout, as well as birds, sculpin, and various mammals. In the ocean, marine mammals, and other fish prey on steelhead but the extent of such predation is not well known.

Anthropogenic Threats. Steelhead have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile steelhead; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the fresh water, estuarine, and coastal ecosystems throughout the species' range. These threats for are summarized in detail under Chinook salmon.

Central California Coast Steelhead

Distribution and Description of the Listed Species

The Central California Coast steelhead DPS includes all naturally spawned anadromous steelhead populations below natural and manmade impassable barriers in California streams from the Russian River (inclusive) to Aptos Creek (inclusive), and the drainages of San Francisco, San Pablo, and Suisun Bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers. Tributary streams to Suisun Marsh including Suisun Creek, Green Valley Creek, and an unnamed tributary to Cordelia Slough (commonly referred to as Red Top Creek), excluding the Sacramento-San Joaquin River Basin, as well as two artificial propagation programs: the Don Clausen Fish Hatchery, and Kingfisher Flat Hatchery/ Scott Creek (Monterey Bay Salmon and Trout Project) steelhead hatchery programs.

The DPS is entirely composed of winter run fish, as are those DPSs to the south. As winter-run fish adults migrating upstream from December-April, and smolts emigrating between March-May (Shapovalov and Taft 1954; Hayes et al. 2008). At the time of the 1996 status review and 1997 listing, little information was available on the specific demographics and life history characteristics of steelhead in this DPS. While age at smoltification typically ranges from 1 to 4 years, recent studies by Sogard et al. (2009) that growth rates in Soquel Creek likely prevent juveniles from undergoing smoltification until age 2. Survival in freshwater reaches tends to be higher in summer and lower from winter through spring for year classes 0 and 1 (Sogard et al. 2009). Larger individuals also survive more readily than do smaller fish within year classes (Sogard et al. 2009). Greater movement of juveniles in fresh water has been observed in winter and spring versus summer and fall time periods, with smaller individuals more likely to move between stream areas (Sogard et al. 2009). Growth rates during this time have rarely been observed to exceed 0.3 mm per day and are highest in winter through spring, potentially due to higher water flow rates and greater food availability (Boughton et al. 2007; Hayes et al. 2008; Sogard et al. 2009).

Status and Trends

The Central California Coast steelhead DPS was listed as a threatened species on August 18, 1997 (62 FR 43937); threatened status was reaffirmed on January 5, 2006 (71 FR 834). Table 15 identifies runs within the Central California Coast steelhead DPS and their estimated run sizes.

Table 15. Central California coast steelhead populations and their estimated abundances

Basin	Estimated Abundance[a]	Year
Russian River	65,000	1970
	1,750-7,000	1994
Lagunitas	500	1994
	400-500	1990s
San Gregorio	1,000	1973
Waddell Creek	481	1933-1942
	250*	1982
	150*	1994
Scott Creek	400	1991
	<100	1991

	300	1994
San Vicente Creek	150*	1982
	50*	1994
San Lorenzo River	20,000	Pre 1965
	1,614	1977
	>3,000*	1978
	600	1979
	3,000	1982
	"few"	1991
	<150*	1994
Soquel Creek	500-800*	1982
	<100	1991
	50-100*	1994
Aptos Creek	200*	1982
	<100	1991
	50-75*	1994

[a]A complete list of data sources is available in Good et al. 2005. According to Good et al. the basis for certain estimates is questionable (noted with an asterisk above).

Estimates of historical abundance are provided here only for background, as the accuracy of the estimates is unclear. An estimate of historical abundance for the total DPS is provided by CDFG at 94,000 fish. This estimate is based on a partial data set and "best professional judgment" (see Good et al. 2005 for a discussion). Other estimates of historical abundance are on a per river basis: According to Busby et al. (1996), Shapovalov and Taft (1954) described an average of about 500 adults in Waddell Creek (Santa Cruz County) for the 1930s and early 1940s, whereas Johnson (1964) estimated a run size of 20,000 steelhead in the San Lorenzo River before 1965. Most of the estimates for run sizes within the DPS are more recent (see Table 15). Two rivers thought to have contained the largest populations within the DPS were the Russian River, and the San Lorenzo River. Based on run size estimates from the 1990s, the Russian River is still likely the largest run within the DPS, albeit estimates suggest the population has declined between 90-96 % from 1970 levels.

No current estimates of total population size are available for this DPS, and consequently there is no time series data available to evaluate the central California coast steelhead population trends. Rather, a general dearth of data on adult steelhead within the DPS, led the biological review team to examine data collected on juvenile steelhead (see Good et al. 2005). In general, juvenile data is considered a poor indicator of the reproductive portion of the population as juvenile age classes exhibit greater mortality rates, which are closely tied to stochastic events, and may move widely within a basin (which may include intermixing with other populations). There is no simple relationship between juvenile and adult numbers (Shea and Mangel 2001). Nonetheless, there was not enough adult data upon which the biological review team could base an assessment of the population trends within the DPS. Therefore, the biological review team log-transformed and normalized juvenile survey data from a number of watersheds (presumed populations). As a result, the team derived trend estimates for five populations: the San Lorenzo River, Scott Creek, Waddell Creek, Gazos Creek, and Redwood Creek in Marin County (see Good et al. 2005 for a detailed discussion of the approach). All populations exhibited downward trends in abundance. Accordingly, provided the juvenile data is representative of the true trend, this data suggests that there is an overall downward trend in abundance in the DPS.

In the most recent review of the status of this DPS, most members of the biological review team (69 %) considered this DPS "likely to become endangered" thus supporting the renewal of the threatened status for central California coast steelhead. Notably, 25 % of the team voted that the DPS be upgraded to endangered status (voted the DPS as" in danger of extinction"; Good et al. 2005). Abundance and productivity were of relatively high concern (as a contributing factor to risk of extinction), and spatial structure was also of concern.

Since the original status review, fishing regulations have changed in a way that probably reduces extinction risk for Central California Coast steelhead. Ocean sport harvest is prohibited, and ocean harvest is considered rare. Although freshwater streams are closed to fishing year round, CDFG has identified certain streams as exceptions where they allow catch-and-release angling or summer trout fishing. In catch-and-release streams, all wild steelhead must be released unharmed.

Critical Habitat

Critical habitat was designated for the Central California Coast steelhead DPS on September 2, 2005 (70 FR 52488), and includes areas within the following hydrologic units: Russian River, Bodega, Marin Coastal, San Mateo, Bay Bridge, Santa Clara, San Pablo, and Big Basin. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Central California Coast steelhead occupy 46 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 1,500 miles of stream habitat and about 400 square miles of estuarine habitat (principally Humboldt Bay). This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 46 occupied watersheds reviewed in NMFS' assessment of critical habitat for Central California Coast steelhead, 14 watersheds received a low rating of conservation value, 13 received a medium rating, and 19 received a high rating of conservation value for the species.

California Central Valley Steelhead

Distribution and Description of the Listed Species

California Central Valley steelhead occupy the Sacramento and San Joaquin Rivers and their tributaries, although they were once widespread throughout the Central Valley (Busby et al. 1996; Zimmerman et al. 2009). Steelhead were found from the upper Sacramento and Pit River systems (now inaccessible due to Shasta and Keswick Dams), south to the Kings and possibly the Kern River systems (now inaccessible due to extensive alteration from water diversion projects), and in both east- and west-side Sacramento River tributaries (Yoshiyama et al. 1996). The present distribution has been greatly reduced (McEwan and Jackson 1996). The California Advisory Committee on Salmon and Steelhead (1988) reported a reduction of steelhead habitat from 6,000 miles historically to 300 miles today. Historically, steelhead probably ascended Clear Creek past the French Gulch area, but access to the upper basin was blocked by Whiskeytown Dam in 1964 (Yoshiyama et al. 1996). Steelhead also occurred in the upper drainages of the Feather, American, Yuba, and Stanislaus Rivers which are now inaccessible (McEwan and Jackson 1996; Yoshiyama et al. 1996).

Existing wild steelhead populations in the Central Valley are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks and a few wild steelhead are produced in the American and Feather Rivers (McEwan and Jackson 1996). Recent snorkel surveys (1999 to 2002) indicate that steelhead are present in Clear Creek (J. Newton, FWS, pers. comm. 2002, *in* Good et al. 2005). Because of the large resident *O. mykiss* population in Clear Creek, steelhead spawner abundance has not been estimated. Until recently, steelhead were thought to be extirpated from the San Joaquin River system. Recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, Calaveras, and other streams previously thought to be void of steelhead (McEwan 2001). On the Stanislaus River, steelhead smolts have been captured in rotary screw traps at Caswell State Park and Oakdale each year since 1995 (Demko et al. 2000). It is possible that naturally spawning populations exist in many other streams but are undetected due to lack of monitoring programs.

The Sacramento and San Joaquin Rivers offer the only migration route to the drainages of the Sierra Nevada and southern Cascade mountain ranges for anadromous fish. The CDFG considers all steelhead in the Central Valley as winter steelhead, although “three distinct runs,” including summer steelhead, may have occurred there as recently as 1947 (CDFG 1995 in Good et al. 2005; McEwan and Jackson 1996). Steelhead in these basins travel extensive distances in fresh water (some exceed 300 km to their natal streams), making these the longest freshwater migrations of any population of winter steelhead. The upper Sacramento River essentially receives a single continuous run of steelhead in from July through May, with peaks in September and February. Spawning begins in late December and can extend into April (McEwan and Jackson 1996).

Status and Trends

NMFS originally listed California Central Valley steelhead as threatened in 1998; this status was reviewed and retained on January 5, 2006 (71 FR 834). Historic Central Valley steelhead run size is difficult to estimate given the paucity of data, but may have approached one to two million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock et al. (1961) estimated an average of 20,540 adult steelhead occurred in the Sacramento River (upstream of the Feather River). Steelhead counts at Red Bluff Diversion Dam declined from an average of 11,187 for the period of 1967 to 1977, to an average of approximately 2,000 through the early 1990s, with an estimated total annual run size for the entire Sacramento-San Joaquin system at no more than 10,000 adults (based on Red Bluff Diversion Dam counts; McEwan and Jackson 1996; McEwan 2001). The five-year geometric mean, however, is just under 2,000 steelhead (Table 16), and the long-term trend suggests that the population is declining.

Table 16. California Central Valley steelhead and their long-term trend

Population	5-Year Mean (Min – Max)[a]	λ	Long-term trend[b]
Sacramento River	1,952 (1,425-12,320)	0.95 (0.90,1.02)	-0.09 (-0.13,-0.06)

[a]Refers to the period ending in 1993, when steelhead counts at Red Bluff Diversion dam ended. Data reported in Good et al. 2005.
[b] 90% confidence limits in parentheses.

The only consistent data available on steelhead numbers in the San Joaquin River basin come from CDFG mid-water trawling samples collected on the lower San Joaquin River at Mossdale. These data indicate a decline in steelhead numbers in the early 1990s, which have remained low through 2002 (Good et al. 2005). In 2004, a total of 12 steelhead smolts were collected at Mossdale (CDFG, unpublished data *in* Good et al. 2005).

Reynolds et al. (1993) reported that 95% of salmonid habitat in California's Central Valley has been lost, largely due to mining and water development activities. They also noted that declines in Central Valley steelhead populations are "due mostly to water development, inadequate instream flows, rapid flow fluctuations, high summer water temperatures in streams immediately below reservoirs, diversion dams which block access, and entrainment of juveniles into unscreened or poorly screened diversions." Thus, overall habitat problems in this ESU relate primarily to water development resulting in inadequate flows, flow fluctuations, blockages, and entrainment into diversions (McEwan and Jackson 1996). Other problems related to land use practices (agriculture and forestry) and urbanization have also contributed to population declines. It is unclear how harvest has affected California's Central Valley steelhead, although it is likely a continuing threat. A CDFG creel census in 2000 indicated that most fish are caught and released, but due to the size of the catch and release fishery (more than 14,000 steelhead were caught and released according to the survey) even a small amount of mortality in this fishery could cause declines in the populations.

Critical Habitat

NMFS designated critical habitat for California Central Valley steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Tehama, Whitmore, Redding, Eastern Tehama, Sacramento Delta, Valley-Putach-Cache, American River, Marysville, Yuba, Valley American, Colusa Basin, Butte Creek, Ball Mountain, Shata Bally, North Valley Floor, Upper Calaveras, Stanislaus River, San Joaquin Valley, Delta-Mendota Canal, North Diablo Range, and the San Joaquin Delta. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, California Central Valley steelhead occupy 67 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 2,300 miles of stream habitat and about 250 square miles of estuarine habitat in the San Francisco-San Pablo-Suisan Bay estuarine complex. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 67 watersheds reviewed in NMFS' assessment of critical habitat for California Central Valley steelhead, seven watersheds received a low rating of conservation value, three received a medium rating, and 27 received a high rating of conservation value for the species.

Lower Columbia River Steelhead

Distribution and Description of the Listed Species

Lower Columbia River steelhead include naturally produced steelhead returning to Columbia River tributaries on the Washington side between the Cowlitz and Wind rivers in Washington and on the Oregon side between the Willamette and Hood rivers, inclusive. In the Willamette River, the upstream boundary of this species is at Willamette Falls. This species includes both winter and summer steelhead. Two hatchery populations are included in this species, the Cowlitz Trout Hatchery winter-run population and the Clackamas River population but neither was listed as threatened. Table 17 identifies the populations that comprise Lower Columbia River steelhead and summarizes several measures available to characterize population viability.

Summer steelhead return sexually immature to the Columbia River from May to November, and spend several months in fresh water prior to spawning. Winter steelhead enter fresh water from November to April, are close to sexual maturation during freshwater entry, and spawn shortly after arrival in their natal streams. Where both races spawn in the same stream, summer steelhead tend to spawn at higher elevations than the winter forms.

Status and Trends

NMFS listed Lower Columbia River steelhead as threatened on March 19, 1998 (63 FR 13347), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). The 1998 status review noted that this ESU is characterized by populations at low abundance relative to historical levels, significant population declines since the mid-1980s, and widespread occurrence of hatchery fish in naturally spawning steelhead populations. During this review NMFS was unable to identify any natural populations that would be considered at low risk.

All populations declined between 1980 and 2000, with sharp declines beginning in 1995. Those with adequate data for modeling are estimated to have a high extinction risk (Good et al. 2005). Abundance trends are generally negative, showing that most populations are in decline, although some populations, particularly summer run, have shown higher return in the last 2 to 3 years. Historical counts in some of the larger tributaries (Cowlitz, Kalama, and Sandy Rivers) suggest the population probably exceeded 20,000 fish while in the 1990s fish abundance dropped to 1,000 to 2,000. Recent abundance estimates of natural-origin spawners range from completely extirpated for some populations above impassable barriers to over 700 for the Kalama and Sandy winter-run populations. A number of the populations have a substantial fraction of hatchery-origin spawners in spawning areas, and are hypothesized to be sustained largely by hatchery production. Exceptions are the Kalama, the Toutle, and East Fork Lewis winter-run populations. These populations have relatively low recent mean abundance estimates with the largest being the Kalama (geometric mean of 728 spawners).

Table 17. Lower Columbia River steelhead populations and select measures of population viability

Life History	Population	Historical Abundance[a]	Mean Number of Spawners	Percent Hatchery Contribution	Median Short-term Growth Rate (λ)[b]
Winter	Cispus River				
	Tilton River		2,787[c]	73	
	Upper Cowlitz River				
	Lower Cowlitz River	1,672			
	Coweeman River	2,243	466[d]	50	0.920, 0.787
	South Fork Toutle River	2,627	504[d]	2	0.933, 0.929
	North Fork Toutle River	3,770	196[d]	0	1.038, 1.038
	Kalama River	554	726[d]	0	0.984, 0.922
	North Fork Lewis River	713			
	East Fork Lewis River	3,131			
	Salmon Creek				
	Washougal River	2,497	323[d]	0	
	Clackamas River		560[e]	41	0.875, 0.830
	Sandy River		977e	42	0.866, 0.782
	Lower Columbia Gorge tributaries	793			
	Upper Columbia Gorge tributaries	243			
	Hood River		756[f]	52	
Summer	Wind River	2,288	472[g]	5	0.995, 0.903
	Hood River		931[f]	83	Unknown
	Washougal River	1,419	264[g]	8	1.029, 0.960
	East Fork Lewis River	422	434[g]	25	

North Fork Lewis River				
Kalama River	3,165	474[g]	32	0.900, 0.664

[a]All data reported by Good et al. 2005. Estimate of historical abundance derived through EDT model associated with large uncertainty. Model also incorporates presently available habitat that was not historically available and vice versa.
[b]λ calculation assumed either hatchery fish fail to reproduce or reproduce at the rate of wild individuals, respectively.
[c]Data from 2002.
[d]Data from 1998-2002.
[e]Data from 1997-2001.
[f]Data from 1996-2000.
[g]Data from 1999-2003.

Critical Habitat

NMFS designated critical habitat for Lower Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Middle Columbia/Hood subbasin, Lower Columbia/Sandy subbasin, Lewis subbasin, Lower Columbia/Clatskanie subbasin, Upper Cowlitz subbasin, Cowlitz subbasin, Clackamas subbasin, Lower Willamette subbasin, and the Lower Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52630) contains additional description of the watersheds that are included as part of this designation, and any areas specifically excluded from the designation.

In total, Lower Columbia River steelhead occupy 32 watersheds. The total area of habitat designated as critical includes about 2,340 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 32 watersheds reviewed in NMFS' assessment of critical habitat for Lower Columbia River steelhead, two watersheds received a low rating of conservation value, 11 received a medium rating, and 26 received a high rating of conservation value for the species. Limiting factors identified for Lower Columbia River steelhead include: degraded floodplain and steam channel structure and function, reduced access to spawning/rearing habitat, altered stream flow in tributaries, excessive sediment and elevated water temperatures in tributaries, and hatchery impacts.

Middle Columbia River Steelhead

Distribution and Description of the Listed Species

The Middle Columbia River steelhead DPS includes all naturally spawned anadromous steelhead populations below natural and manmade impassible barriers in Oregon and Washington drainages upstream of the Hood and Wind River systems, up to and including the Yakima River (61 FR 41541). Steelhead from the Snake River Basin (described elsewhere) are excluded from this DPS. Seven artificial propagation program are part of this DPS: The Touchet River endemic, Yakima River kelt reconditioning program (in Satus Creek, Toppenish Creek, Naches River, and the Upper Yakima River), and the Umatilla River and the Deschutes River steelhead hatchery programs. These artificially propagated populations are considered no more divergent relative to the local natural populations than would be expected between closely related natural populations within the DPS.

Middle Columbia River steelhead occupy the intermontane region of the Pacific Northwest, which includes some of the driest areas in the region generally receiving less than 15.7 inches of rainfall annually. Major drainages in this ESU are the Deschutes, John Day, Umatilla, Walla Walla, Yakima, and Klickitat river systems. The area is generally characterized by its dry climate and harsh temperature extremes. Almost all steelhead populations within this DPS are summer-run fish; the only exceptions are the only populations of inland winter steelhead, which occur in the Klickitat River and Fifteenmile Creek (Busby et al. 1996). According to Interior Columbia Basin Technical Recovery Team (ICBTRT 2003) this DPS is comprised of 16 putative populations in four major population groups (Cascades Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River) and one unaffiliated independent population (Rock Creek). See Table 18 for a list of extant (putative) populations that compose this DPS. There are two extinct populations in the Cascades Eastern Slope major population group, the White Salmon River and Deschutes Crooked River above the Pelton/Round Butte Dam complex. Present population structure is delineated largely on the basis of geographical proximity, topography, distance, ecological similarities or differences. Additional genetic studies are needed to describe the DPS substructure, as well as the fine-scale genetic structure of the populations within a particular basin (e.g., John Day River).

Table 18. Middle Columbia River steelhead populations and select measures of population viability

Population[a]	**Major Population Groups**	**Mean Number of Spawners (range)**[b]	**Percent Hatchery Contribution**[c]	**Long-term Growth Rate (λ)**[d]
Klickitat River	Cascade Eastern Slope	155 redds (97-261)		
Fifteenmile Creek	Cascade Eastern Slope	2.87 rpm (1.3-6.0)	0	1.129
Deschutes River - eastside	Cascade Eastern Slope	13,455 (10,026-21,457)	72	1.022, 0.840, 0.942
Descutes River – westside	Cascade Eastern Slope			
John Day lower mainstem tributaries	John Day River	1.4 rpm (0-5.4)		1.013
North Fork John Day	John Day River	Upper NF - 2.57 rpm (1.6-5.0)[e]		1.011
		Lower NF - 3.52 rpm (1.5-8.8)		1.174
Middle Fork John Day	John Day River	3.70 rpm (1.7-6.2)		0.966
South Fork John Day	John Day River	2.52 rpm (0.9-8.2)		0.967
John Day upper mainstem	John Day River	2,122 (926-4,168)	4	0.975, 0.966
Rock Creek	Unaffiliated Area			
Umatilla River	Walla Walla & Umatilla	2,486 (1,480-5,157)	40	1.007, 0.969
Walla Walla	Walla Walla & Umatilla			
Touchet River	Walla Walla & Umatilla	345 (273-527)[f]	16	0.961, 0.939
Toppenish & Satus	Yakima River			

Creek				
Naches River	Yakima River			
Yakima River upper mainstem	Yakima River	1,801 (1,058-4,061)	3	1.009

[a]Population groups defined by the ICBTRT (2003).
[b]Values represent the 5-year geometric mean in spawners, redds, or redds per mile (RPM). Values calculated from data series using years 1997-2001 or 1998-2001. See Good et al. (2005) for details.
[c]Hatchery production in the recent past and at present consists of locally-derived broodstock, although straying of production fish into the Deschutes River has been a persistent problem. Data from Good et al. 2005.
[d]Multiple estimates for long-term growth (λ) presented for some populations representing two different assumptions on the contribution of hatchery fish to the natural production. Where two or more values are presented, the first value reflects the assumption that hatchery fish do not contribute to natural production, and the second value reflects the assumption that hatchery contribute to natural production at the same rate as natural-origin spawners. Deschutes River values are reflective of total population, not eastside only. The λ value is calculated from data (1980-1999) from Warm Springs area. Data series upon which values are calculated varies across basins. See Good et al. (2005) for details on the length and time of data series available by population.

Most Middle Columbia River steelhead smolt at 2 years of age and spend 1 to 2 years at sea prior to re-entering natal river systems. They may remain in such rivers for up to a year prior to spawning (Howell et al. 1985). Within this ESU, the Klickitat River is unusual, as it produces both summer and winter steelhead. The summer steelhead are dominated by year-class-two ocean steelhead, whereas most other rivers in this region produce about equal numbers of both age-one and age-two ocean steelhead. Factors contributing to the decline of Middle Columbia river steelhead include hydropower development and agriculture; these land uses impede or prevent migrations, alter water availability, and alter water chemistry and temperatures.

Status and Trends

Middle Columbia River steelhead were listed as threatened in 1999 (64 FR 14517), and their status was reaffirmed on January 5, 2006 (71 FR 834). The precise pre-1960 abundance of this species is unknown. Based upon the Washington Department of Fish and Wildlife’s estimates of the historic run size for the Yakima River at 100,000 steelhead, Busby et al. (1996) surmised that total DPS abundance likely exceeded 300,000 returning adults. By 1993, the estimated 5-year average size (ending in 1993) of the Middle Columbia steelhead DPS was 142,000 fish (Busby et al. 1996). Survey data collected between 1997 and 2001 indicates that several populations within the DPS have increased since the last status review (Good et al. 2005). However, long-term annual population growth rate (λ) is negative for most populations (see Table 18).

In contrast, short term trends in major areas were positive for 7 of the 12 areas with available data (see Good et al. 2005). Spawner numbers in the Yakima River, the Deschutes River and sections of the John Day River system were substantially higher compared to numbers surveyed between 1992 and 1997 (Good et al. 2005). Similarly, spawner numbers substantially increased in the Umatilla River and Fifteenmile Creek relative to annual levels in the early 1990s. Nonetheless, most populations remain below interim target levels. For instance, the Yakima River returns are still substantially below interim target levels of 8,900 (the current 5-year average is 1,747 fish) and estimated historical return levels. In fact, the majority of spawning occurs in only one tributary, Satus Creek (Berg 2001 in Good et al. 2005). Based on recent 5-year geometric means, only the Deschutes River exceeded interim target levels (Good et al. 2005). While increases in short-term trends could suggest improvements within the DPS, given that the average population growth rate across all streams is negative (0.98 assuming hatchery spawners do not contribute to production, and 0.97 assuming that both hatchery and natural-origin fish contribute equally) and evidence of large fluctuation in marine survival for the species, recent increases in population sizes must be viewed cautiously.

Critical Habitat

NMFS designated critical habitat for Middle Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Upper Yakima, Naches, Lower Yakima, Middle Columbia/Lake Wallula, Walla Walla, Umatilla, Middle Columbia/Hood, Klickitat, Upper John Day, North Fork John Day, Middle Fork John Day, Lower John Day, Lower Deschutes, Trout, and the Upper Columbia/Priest Rapids subbasins, and the Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.

In total, there are 114 watersheds within the range of Middle Columbia River steelhead. The total area of habitat designated as critical includes about 5,800 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 114 watersheds reviewed in NMFS' assessment of critical habitat for Middle Columbia River steelhead, nine watersheds received a low rating of conservation value, 24 received a medium rating, and 81 received a high rating of conservation value for the species. Although pristine habitat conditions are still present in some wilderness, roadless, and undeveloped areas, habitat complexity has been greatly reduced in many areas of designated critical habitat for Middle Columbia River steelhead. Limiting factors identified for Middle Columbia River steelhead include: hydropower system mortality, reduced stream flow, impaired passage, excessive sediment, degraded water quality, and altered channel morphology and floodplain.

Northern California Steelhead

Distribution and Description of the Listed Species

The Northern California DPS of steelhead includes all naturally spawned steelhead populations below natural and manmade impassible barriers in California coastal river basins from Redwood Creek south to, but not including the Russian river, and two artificial propagation programs (Yager Creek Hatchery, and North Fork Gualala River Hatchery). In the recent update on the status of this DPS, the southern boundary of the DPS was redefined to include the small coastal streams south of the Gualala River (between the Gualala River and the Russian River) that support steelhead. This DPS consists of winter and summer-run fish, as well as "half-pounders" – a sexually steelhead that returns from the sea after spending less than a year in the ocean. Generally, a half-pounder will overwinter in freshwater and return to the ocean in the spring.

Status and Trends

NMFS listed Northern California steelhead as threatened on June 7, 2000 (65 FR 36074), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). Long-term data sets are limited for Northern California steelhead. Before 1960, estimates of abundance specific to this DPS were available from dam counts in the upper Eel River (Cape Horn Dam; annual average number of adults was 4,400 in the 1940s), the South Fork Eel River (Benbow Dam; annual average number of adults was 18,000 in the 1940s), and the Mad River (Sweasey Dam; annual average number of adults was 3,800 in the 1940s). According to California Department of Fish & Game nearly 200,000 spawning steelhead may have comprised this DPS in the early 1960s (Good et al. 2005). At the time of the first status review on this population, adult escapement trends could be calculated for seven populations. Five of the seven populations exhibited declines, while two exhibited increases with a range of almost 6% annual decline to a 3.5% increase. At the time, little information was available on the actual contribution of hatchery fish to natural spawning, there was and continues to be insufficient information to calculate an overall abundance estimate for Northern California steelhead (Busby et al. 1996).

Recent time series data is also limited for this DPS, with recent abundance estimates available for only four populations, three summer-run and one winter-run. Similarly, Good et al. (2005) could only calculate the population growth rate for three populations (see Table 19). Population growth rates are negative for two of the three populations, the South Fork Eel River winter-run and the Middle Fork Eel River summer-run. Based on time series data for the Middle Fork Eel River, both the long-term and short-term trends are downward. Due to the lack of adult data on which to base their risk assessment, Good et al. (2005) also examined data on juvenile steelhead, and found both upward and downward trends. The lack of data for the populations within this DPS, particular winter-run fish is of continuing concern.

Table 19. Northern California steelhead salmon populations and select measures of population viability

River	Historical Abundance[a]	Mean Number (CI)[b]	Growth Rate (λ)[c]
Redwood Creek	10,000	3 (n/a)	
Mad River	6,000	162 (162-384)[d]	1.00 (0.93,1.05)[e]
Freshwater Creek winter run		32 (25-32)	
Eel River -Total	82,000		
South Fork Eel River	34,000		0.98 (0.92,1.02)
Middle Fork Eel River	23,000	418 (384-1,246)[e]	0.98 (0.93,1.04)[g]
Mattole River	12,000		
Ten Mile River	9,000		
Noyo River	8,000		
Big River	12,000		
Navarro River	16,000		
Garcia River	4,000		
Gualala River	16,000		
Other Humboldt County streams	3,000		
Other Mendocino County streams	20,000		

[a]Historical abundances (1963) are considered uncertain by the author, California Department of Fish & Game. All data are reported in Good et al. 2005.
[b]Value represents the geometric mean number of fish surveyed by snorkel counts or weir counts (e.g., Mad River and MF Eel counts are from snorkel surveys – for the MF Eel River these are snorkel counts of fish holding in pools of the main stem). See Good et al. 2005 for details.
[c]Growth rate calculated upon method where a λ=1.0 could describe a population that is in decline due to environmental stochasticity.

[d]Five year mean of Mad River summer-run steelhead only.
[e]Population growth rate calculated on Mad River winter-run steelhead only.

Critical Habitat

NMFS designated critical habitat for Northern California steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Redwood Creek, Trinidad, Mad River, Eureka Plain, Eel River, Cape Mendocino, and the Mendocino Coast. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Northern California steelhead occupy 50 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 3,000 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 50 watersheds reviewed in NMFS' assessment of critical habitat for Northern California steelhead, nine watersheds received a low rating of conservation value, 14 received a medium rating, and 27 received a high rating of conservation value for the species. Two estuarine areas used for rearing and migration (Humboldt Bay and the Eel River estuary) also received a rating of high conservation value.

Puget Sound Steelhead

Distribution and Description of the Listed Species

The Puget Sound DPS for steelhead includes all naturally spawned anadromous winter-run and summer-run steelhead populations in watersheds of the Strait of Juan de Fuca, Puget Sound and Hood Canal, Washington. Boundaries of this DPS extend to and include the Elwha River to the west, and the Nooksack River and Dakota Creek to the north. Hatchery production of steelhead is widespread throughout this DPS, but only two artificial propagation programs are part of this DPS: the Green River natural and Hamma Hamma winter-run steelhead hatchery populations. The remaining hatchery programs are not considered part of the Puget Sound steelhead DPS because they are more than moderately diverged from the local native populations (NMFS 2005c).

The oceanic distribution of Puget Sound steelhead is not well understood. Winter and summer runs from multiple DPS' comingle in the North Pacific Ocean and some may undergo extensive migrations as a result of the location of their natal streams and oceanic "centers of abundance" (Light et al. 1989). Tagging and genetic studies indicate that Puget Sound steelhead migrate to the central North Pacific ocean (see French et al. 1975, Hartt and Dell 1986, and Burgner et al. 1992 in NMFS 2005c). However, the fjord-like ecosystem of Puget Sound may affect steelhead migration patterns; for example, some populations of coho and Chinook salmon, at least historically, remained within Puget Sound and did not migrate to the Pacific Ocean itself. Even when Puget Sound steelhead migrate to the high seas, they may spend considerable time as juveniles or adults in the protected marine environment of Puget Sound. Oceanic residence times varies among populations within the DPS, with some populations spending only one season in the ocean and others spending three years in marine waters before returning to their natal stream for spawning. Generally, winter-run steelhead enter their natal freshwater systems later (November to April) in the year than summer-run steelhead (May to October), and thus have a shorter freshwater residence time just prior to spawning. The result is that winter-run steelhead have a lower pre-spawn mortality rate than summer-run steelhead (NMFS 2005c). Winter-run steelhead are also more prevalent than summer-run fish, comprising 37 of the 53 populations within this DPS.

Status and Trends

NMFS listed Puget Sound steelhead as a threatened species on May 11, 2007 (72 FR 26722). At the time of the listing, the biological review team concluded that: the viability of Puget Sound steelhead is at a high risk due to declining productivity and abundance; Puget Sound steelhead are at moderate risk due to reduced spatial complexity and connectivity among populations within the DPS, and reduction in life-history diversity within populations and from the threats posed by artificial propagation and harvest. The Puget Sound steelhead DPS includes 53 putative populations; most of which are composed of winter-run fish. Summer-run populations within Puget Sound are small, with most averaging less than 200 spawners, and most lack sufficient data to estimate population abundance. Table 20 lists several of the populations that comprise Puget Sound steelhead as well as some statistics summarizing their current status.

In general, steelhead are most abundant in the northern Puget Sound streams. The largest populations in this DPS are in the Skagit River and Snohomish River winter-run steelhead populations. The recent geometric mean escapement is 5,608 winter-run steelhead in the Skagit, and 3,230 winter-run steelhead in the Snohomish River. The Green River and Puyallup River populations, in central Puget Sound, are the next largest populations and average approximately 1,500 (Green) and 1,000 (Puyallup) winter-run steelhead spawners annually.

Table 20. Puget Sound steelhead salmon populations and a summary of available demographic data

Population	Life History	Historical Abundance (Percent Annual change[a]	Mean Number of Spawners[b]	Trends in escapement[c]	Median short-term growth rate (λ)[d]
Canyon	Summer				
	Winter				
Skagit	Summer				

	Winter	7,700 (2.0)	5608.5	-0.002	0.997 (0.997-0.998)
Snohomish	Summer				
Snohomish	Winter	8,000 (3.1)	3230.1	-0.019	0.804 S
Dakota	Winter				
Nooksack	Winter	NA (-11.6)			
Samish	Winter		852.2	0.067**	0.988 (0.997-0.998)
Stillaguamish	Winter	NA (-6.3)	550.2	-0.065****	0.885 S (0.884-0.885)
Tolt	Summer		119.0	0.025	1.018 (1.017-1.018)
Green	Summer				
Green	Winter		1625.5	0.008	0.932 (0.932-0.933)
Cedar	Winter		36.8	-0.179**	0.808 S (0.804-0.811)
Lake Washington	Winter	NA (-17.5)	36.8	-0.180****	0.802 (0.800-0.803)
Nisqually	Winter	1,200 (-5.1)	392.4	-0.084****	0.918 (0.917-0.918)
Puyallup	Winter	2,000 (-5.2)	1001.0	-0.062****	0.882 (0.881-0.882)
Dewatto	Winter		24.7		1.020 (1.008-1.020)
Dosewallips	Winter		76.7		
Duckabush	Winter		17.7	0.017	
Hamma Hamma	Winter		51.9	0.291*	1.013
Quilcene	Winter		15.1	-0.006	0.988 S
Skokomish	Winter	NA (-3.5)	202.8	-0.075****	0.865 S
Tahuya	Winter	NA (-0.6)	117.0	0.009	0.983 (0.982-0.983)
Union	Winter		55.3	0.008	0.969 S
Elwha	Summer				
	Winter		210.0		0.966 (0.965-0.966)
Dungeness	Winter	NA (-5.5)	173.8	-0.076	0.924 (0.924-0.924)
Mc Donald	Winter			-0.031	0.732 S
Morse	Winter	200 (-12.3)		-0.006	0.945 (0.945-0.946)

[a]Values of historical abundance represent the total escapement for the subbasin. Data generally span the late 1970s to mid 1990s. All estimates are run reconstructions, except the Nooksack which comes from spawner surveys. Specific data years for each data set and other details are noted in Busby et al. 1996.
[b]Geometric mean estimates of escapement for Puget Sound steelhead are provided for the five year period from 2000-2004, and for hatchery plus natural spawners (NMFS 2005c).
[cr]Estimates of temporal trends in escapement and total run size (transformed by natural log). Estimates are the slopes of the regressions of natural log (spawners or run size) on year. Estimates provided are for the entire available dataset and are based on natural fish (data years noted in NMFS 2005c). *, P<0.05; **, P<0.01; ***, P<0.001; ****, P<0.0001 (all other values are not significant (data from NMFS 2005c)).
[d]Estimates for each population were computed for the most recent 10 years of data (1995-2004). S – Denotes that the estimate is based on natural spawners alone. Values in parentheses represent the 95% Confidence Intervals of the estimate (data from NMFS 2005c).

Estimates of historical abundance for this DPS are largely based on catch data. The earliest catch records from commercial fisheries in the late 1880s indicate that the catch peaked at 163,796 steelhead in Puget Sound in 1895 (NMFS 2005c). Based on this catch data, NMFS (2005c) estimated that the peak run size for Puget Sound steelhead ranged between 300,000 and 550,000 fish. Given that most fish were harvested in terminal fisheries (nets set at the mouth of rivers) NMFS expects that this estimate is a fair estimate of the Puget Sound DPS as it is unlikely to include fish from neighboring rivers outside of the Puget Sound DPS. As early as 1898, Washington officials expressed concerns that the run had declined by half of its size in only three years (NMFS 2005c). Since 1925, Washington has managed steelhead as a game fish, and in 1932 the State prohibited the commercial catch, possession or sale of steelhead.

Run size for this DPS was calculated in the early 1980s at about 100,000 winter-run fish and 20,000 summer-run fish. It is not clear what portion were hatchery fish, but a combined estimate with coastal steelhead suggested that roughly 70% of steelhead in ocean runs were of hatchery origin. Escapement of wild fish to spawning grounds would be much lower without the influx of hatchery fish (Busby et al 1996).

NMFS first status review for Puget Sound steelhead demonstrated that 80 % of the runs for which there was data had declining trends in abundance. Basinwide abundance estimates from Busby et al. (1996) are depicted in Table 20. Busby et al. (1996) noted that the largest decline, an 18% annual decline, occurred in the Lake Washington population. On the contrary, the largest increase in abundance occurred in the Skykomish River winter-run steelhead (the Skykomish River is a tributary to the Snohomish River) at a 7% annual increase. Estimates of spawner abundance in the Skagit and Snohomish rivers, the two largest steelhead producing basins in the DPS, were about 8,000 naturally spawning adult steelhead each (Table 20). These two basins exhibited modest overall upward trends at the time of the first status review. Recent data demonstrates significant declines in the natural escapement of steelhead throughout the DPS, especially in the southern Puget Sound populations. Significant positive trends have occurred in the Samish and the Hamma Hamma winter-run populations. The increasing trend in the Hamma Hamma River appears to be the result of a captive rearing program, rather than due to natural escapement. The predominant downward trends in escapement and run size of natural steelhead in the Puget Sound DPS, both over the long-term and short-term, is of concern particularly given that despite widespread reductions in direct harvest since the mid 1990s (NMFS 2005c).

Critical Habitat

NMFS has not designated critical habitat for Puget Sound steelhead.

Snake River Steelhead

Distribution and Description of the Listed Species

The Snake River Basin steelhead DPS includes all naturally spawned populations of steelhead in streams in the Snake River basins of southeast Washington, northeast Oregon and Idaho. Six artificial propagation programs are considered part of this DPS: The Tucannon River, Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater, East Fork Salmon River, and the Little Sheep Creek/Imnaha river hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than what would be expected between closely related natural populations within the DPS.

Snake River Basin steelhead are distributed throughout the Snake River drainage basin, migrating a considerable distance from the ocean to use high-elevation tributaries (typically 1,000-2,000 m above sea live). Generally, classified as summer-run fish, Snake River steelhead enter the Columbia River from late June to October. After remaining in the river through the winter, Snake River steelhead spawn the following spring (March to May). Managers recognize two life history patterns within Snake River steelhead primarily based on ocean age and adult size upon return: A-run steelhead are typically smaller, have a shorter fresh water and ocean residence (generally 1 year in the ocean), and begin their up-river migration earlier in the year; whereas B-run steelhead are larger, spend more time in fresh water and the ocean (generally 2-years in ocean), and appear to start their upstream migration later in the year. Table 21 lists the life-history type associated with each of the 24 demographically independent populations within this DPS.

Table 21. Snake River steelhead populations and a summary of available demographic data

Populations[a]	**Life History**	**Historical Abundance (Percent Annual change**[b]	**Mean Number of Spawners (range)**[c]	**Percent Hatchery Contribution**[d]	**Long-term growth rate (λ)**[e]
Tucannon River	A-run	400 (-18.3)	407 (257-628)	74	0.886, 0.733
Asotin Creek	A-run	200 (-19.7)	87 exp. redds (0-543)	Unknown	
Lower Clearwater	A-run				
South Fork Clearwater	B-run				
Lolo Creek	B-run				
Selway River	B-run				
Lochsa River	B-run				
North Fork Clearwater River					
Lower Grande Ronde	A-run	(-0.5)			
Joseph Creek	A-run		1,542 (1.077-2,385)	0	1.069
Wallowa River	A-run	(-3.0)			
Upper Grande Ronde	A-run		1.54 rpm (0.3-4.7)	23	0.967, 0.951
Little Salmon and lower Salmon tributaries	A-run				
South Fork Salmon River	B-run	(-8.0)			
Secesh River	B-run				
Chamberlain Creek	A-run				
Lower Middle Fork Salmon	B-run	(-25.8**)			
Upper Middle Fork Salmon	B-run				
Panther Creek	A-run				
North Fork Salmon	A-run				
Lemhi River	A-run				
Pahsimeroi River	A-run	1,400 (0.1)			
East Fork Salmon River	A-run	150*(-6.0)			
Upper Mainstem Salmon River	A-run				
Imnaha River	A-run	(81.2)	3.7 rpm (2.0-6.8)	20	1.042, 1.026
Hells Canyon tributaries	A-run				

[a] Demographically independent populations identified by ICBTRT 2003.
[b]Values of historical abundance represent total escapement as calculated in NMFS' first status review for the DPS. Values with a * are estimates of total run; no escapement estimate was available. Data generally span the late 1980s to mid 1990s. Estimates are calculated from different data types, and include data from spawner surveys, run reconstructions, or dam/weir counts. Specific data years for each data set and other details are noted in Busby et al. 1996. **=Middle Fork and tributaries.
[c]Geometric mean estimates of escapement represent total escapement (hatchery plus natural adult returns).
[c]Estimates of percentage of hatchery returns in Granite dam aggregate counts indicate that returns are predominantly composed of hatchery fish (about 85%). Values from Good et al. 2005.
[c]Multiple estimates for long-term growth (λ) presented for some populations represent two different assumptions on the contribution of hatchery fish to natural production. Where two or more values are presented, the first value reflects the assumption that hatchery fish do not contribute to

natural production, and the second value reflects the assumption that hatchery contribute to natural production at the same rate as natural-origin spawners. Data series upon which values are calculated, varies across basins. See Good et al. (2005) for details on the length and time of data series available by population.

Status and Trends

NMFS listed Snake River steelhead as threatened in 1997 (62 FR 43937), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). NMFS 1997 status review identified sharp declines in the returns of naturally produced steelhead, beginning in the mid-1980s. At the time nine of 13 trend indicators were in decline and the average abundance (geometric mean, 1992-1996) for the DPS was 75,000 adult steelhead (8,900 naturally produced). Of this, about 7,000 were A-run adults, and about 1,400 were B-run adults (Busby et al. 1996).

The paucity of information on adult spawning escapement for specific tributaries of the Snake River Basin DPS continues to make a quantitative assessment of viability difficult. Available data indicate that the overall long-term estimates of population trends have remained negative. Return estimates for the late 1990s to early 2000s are summarized in Table 21. Annual return estimates are limited to counts of the aggregate return over Lower Granite Dam, and spawner estimates for the Tucannon, Asotin, Grande Ronde, and Imnaha Rivers. The 2001 return over Lower Granite Dam was substantially higher relative to the low levels seen in the 1990s; the recent geometric 5-year mean abundance (Total escapement 106,175 with 14,768 natural returns) was approximately 28% of the interim recovery target level (52,000 natural spawners). The 10-year average for natural-origin steelhead passing Lower Granite Dam between 1996 and 2005 is 28,303 adults. Long-term trend estimates of the population growth rate (λ) across the available data set was 0.998 assuming that natural returns are produced only from natural-origin spawners, and 0.733 if both hatchery and wild spawners are contributing to production equally. Parr densities in natural production areas, which are another indicator of population status, have been substantially below estimated capacity for several decades. The Snake River supports approximately 63% of the total natural-origin production of steelhead in the Columbia River Basin. Genetic diversity is currently being depressed by the displacement of natural fish by hatchery fish (declining proportion of natural-origin spawners). Homogenization of hatchery populations occurs within basins and some populations exhibit high stray rates.

Critical Habitat

NMFS designated critical habitat for Snake River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Hells Canyon, Imnaha River, Lower Snake/Asotin, Upper Grand Ronde River, Wallowa River, Lower Grand Ronde, Lower Snake/Tucannon, Upper Salmon, Pahsimeroi, Middle Salmon-Panther, Lemhi, Upper Middle Fork Salmon, Lower Middle Fork Salmon, Middle Salmon, South Fork Salmon, Lower Salmon, Little Salmon, Upper and Lower Selway, Lochsa, Middle and South Fork Clearwater, and the Clearwater subbasins, and the Lower Snake/Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.

There are 289 watersheds within the range of Snake River steelhead. The total area of habitat designated as critical includes about 8,000 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 289 fifth order streams reviewed in this DPS, 231 received a high conservation value rating, 44 received a medium rating, and 14 received a rating of low conservation value for the species. The lower Snake/Columbia rearing/migration corridor downstream of the spawning range has a high conservation value. Limiting factors identified for Snake River Basin steelhead include: hydrosystem mortality, reduced stream flow, altered channel morphology and floodplain, excessive sediment, degraded water quality, harvest impacts, and hatchery impacts.

South-Central California Coast Steelhead

Distribution and Description of the Listed Species

The South-Central California Coast steelhead DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California. No artificially propagated steelhead populations that reside within the historical geographic range of this DPS are included in this designation. The two largest basins within this DPS are the inland basins of the Pajaro River and the Salinas River. Both of these watersheds drain intercoastal mountain ranges and have long alluvial lower stretches. Principle sub-basins in the Pajaro River that support steelhead include: Corralitos Creek, Pescadero Creek, Uvas Creek, and Pacheco Creek. Principle sub-basins in the Salinas River that support steelhead include the Arroyo Seco River, Gabilan Creek, Paso Robles Creek, Atascadero Creek and Santa Margarita Creek. Other important watersheds include the smaller coastal basins of the Carmel River, and St. Rosa and San Luis Obispos creeks.

Status and Trends

NMFS listed South-Central California Coast steelhead as threatened in 1997, and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). Historical data on the South-Central California Coast steelhead DPS are sparse and no credible historic or recent estimates of total DPS size are available. Steelhead are present in a large portion of the historically occupied basins within this DPS (estimated 86-95 %) but observed and inferred abundance suggest many of this basins support a small fragment of their historic run size. Present population trends within individual watersheds continuing to support runs is generally unknown, but may vary widely between watersheds. No data are available to estimate the steelhead abundance or trends in the two largest watersheds in the DPS, the Pajaro and Salinas basins, although these basins are highly degraded and expected to support runs much reduced in size from historical levels.

Steelhead in the Carmel Basin have been monitored at San Clemente Dam since 1964, representing one of the longest data sets available for steelhead in this DPS. However, this data is also limited because a nine year gap exists in the series, a large portion of the run spawns below the dam, and the older dam counts may be incomplete. Between NMFS' 1997 status review and 2005 status update, continuous data from San Clement dam suggests that the abundance of adult spawners in the Carmel River has increased. Carmel River time series data indicate that the population declined by about 22% per year between 1963 and 1993, and between 1991 and 1997 the population increased from one adult to 775 adults at San Clemente Dam. Good et al. (2005) deemed this increase too great to attribute simply to improved reproduction and survival of the local steelhead population. Other possibilities were considered, including that the substantial immigration or transplantation occurred, or that resident trout production increased as a result of improved environmental conditions within the basin. The five-year geometric mean calculated by Good et al. (2005) for the Carmel River population (1998-2002) was 611 steelhead (range 1-881).

Critical Habitat

NMFS designated critical habitat for South-Central California Coast steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Pajaro River, Carmel River, Santa Lucia, Salinas River, and Estero Bay. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, South-Central California Coast steelhead occupy 30 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 1,250 miles of stream habitat and about 3 square miles of estuarine habitat (e.g., Morro Bay). This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 30 watersheds reviewed in NMFS' assessment of critical habitat for South-Central California Coast steelhead, six watersheds received a low rating of conservation value, 11 received a medium rating, and 13 received a high rating of conservation value for the species.

Southern California Steelhead

Distribution and Description of the Listed Species

The Southern California steelhead DPS includes all naturally spawned populations of steelhead in streams from the Santa Maria River, San Luis Obispo County, California (inclusive) to the United States-Mexico border. Artificially propagated steelhead that reside within the historical geographic range of this DPS are not included in the listing.

A comprehensive assessment of the distribution of steelhead within the Southern California DPS indicates that steelhead occur in most of the coastal basins (Boughton and Fish 2003 in Good et al. 2005). Major watersheds occupied by steelhead in this DPS include the Santa Maria, Santa Ynez, Ventura, Santa Clara rivers. Smaller watersheds that support steelhead include the Los Angeles, San Gabriel, San Luis Rey, and Sweetwater rivers, and San Juan and San Mateo creeks. Significant portions of several upper watersheds are contained with four national forests (Los Padres, Angeles, Cleveland, and San Bernardino National Forests), whereas coastal and inland valleys are dominated by urban development, with the Los Angeles basin being the most expansive and densest urban area in the DPS. Populations within the southernmost portion of the DPS (San Juan Creek, San Luis Rey River, and San Mateo Creek) are separated from the northernmost populations by about 80 miles.

Status and Trends

NMFS listed Southern California steelhead as endangered in 1997 (62 FR 43937), and reaffirmed their status as endangered on January 5, 2006 (71 FR 834). Historical and recent data is generally lacking for Southern California steelhead, making a general assessment of their status difficult. The historical run size estimate for the entire DPS was between 32,000-46,000 steelhead, but this estimate omits the Santa Maria system and basins south of Malibu Creek (Busby et al. 1996). Estimates for the Santa Ynez River Basin, probably the largest run historically, range from 13,000 to 30,000 spawners, although this number may underestimate the steelhead abundance in the basin prior to the construction of Juncal and Gibraltar dams (Busby et al. 1996; Good et al. 2005). No recent data are available for steelhead in the Santa Ynez basin, and most of the historical spawning habitat was blocked by Bradbury and Gibraltar dams. Steelhead and rainbow trout are known to occur in streams downstream of Bradbury Dam, but no estimates of abundance or trends are available. Similarly, Twitchell Dam in the Santa Maria River, and Casitas Dam on Coyote Creek and Matilija Dam on Matilija Creek block access to significant portions of historical spawning and rearing habitat, and alter the hydrology of the basins. A fish ladder and counting trap at the Vern Freeman Diversion Dam on the Santa Clara River is thought to be dysfunctional (Good et al. 2005). In general run sizes in river systems within the DPS are believed to range between less than five anadromous adults per year, to less than 100 anadromous adults per year. An estimated 26-52% of historically occupied basins are believed to still contain some steelhead, and about 30% are believed vacant, extirpated or nearly extirpated due to dewatering or barriers that block spawning habitat.

Critical Habitat

NMFS designated critical habitat for Southern California steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Santa Maria River, Santa Ynez, South Coast, Ventura River, Santa Clara Calleguas, Santa Monica Bay, Callequas, and San Juan hydrological units. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Southern California steelhead occupy 32 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 700 miles of stream habitat and about 22 square miles of estuarine habitat, mostly within Humboldt Bay. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 32 watersheds reviewed in NMFS' assessment of critical habitat for Southern California steelhead, five watersheds received a low rating of conservation value, six received a medium rating, and 21 received a high rating of conservation value for the species.

Upper Columbia River Steelhead

Distribution and Description of the Listed Species

The Upper Columbia River steelhead DPS includes all naturally spawned populations of steelhead in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the United States-Canada border. Six artificial propagation programs are part of this DPS: the Wenatchee River, Wells Hatchery (in the Methow and Okanogan rivers), Winthrop National Fish Hatchery, Omak Creek, and the Ringold steelhead hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this DPS.

Rivers in this DPS primarily drain the east slope of the northern Cascade Mountains and include the Wenatchee, Entiat, Methow, and Okanogan River Basins. Some of these upper Columbia River subbasins, including the Okanogan River and the upper Columbia River proper, extend into British Columbia although steelhead do not occur in significant numbers in British Columbia and thus were not included in the DPS. Identified largely on the basis of spawning distributions, this DPS is composed of four putative populations defined by the Wenatchee, Entiat, Methow, and Okanogan rivers (Table 22). Historically (before the construction of Grand Coulee Dam blocked 50% of the river to Upper Columbia steelhead) major watershed that may have supported steelhead within this DPS were the Sanpoil, Spokane, Colville, Kettle, Pend Oreille and Kootenai rivers (ICBTRT 2003).

All upper Columbia River steelhead are summer-run steelhead. Adults return in the late summer and early fall, with most migrating relatively quickly to their natal tributaries. A portion of the returning adult steelhead overwinters in mainstem reservoirs, passing over upper-mid-Columbia dams in April and May of the following year. Spawning occurs in the late spring of the year following river entry. Juvenile steelhead spend 1 to 7 years rearing in fresh water before migrating to sea. Smolt outmigrations are predominantly year class two and three (juveniles), although some of the oldest smolts are reported from this DPS (7 years). Most adult steelhead return to fresh water from sea after 1 or 2 years.

Status and Trends

NMFS originally listed Upper Columbia River steelhead as endangered in 1997 (62 FR 43937). On January 5, 2006, after reviewing the status of Upper Columbia River steelhead and noting an increase in abundance and more widespread spawning, NMFS reclassified the status of Upper Columbia River threatened (71 FR 834). In accordance with a United States District Court decision, NMFS reinstated the endangered status of Upper Columbia River steelhead in June 2007 (62 FR 43937). NMFS appealed the Court's decision, and on June 18, 2009, the District Court revised its ruling, effectively reinstating threatened status for Upper Columbia River steelhead (74 FR 42605). Thus, consistent with the court's rulings and the NMFS' listing determination of January 5, 2006, Upper Columbia River steelhead are listed as threatened under the ESA.

Since the 1940s, artificially propagated steelhead have seeded this DPS to supplement the numbers lost with the construction Grand Coulee Dam. Abundance estimates of returning naturally produced Upper Columbia River steelhead have been based on extrapolations from mainstem dam counts and associated sampling information (e.g., hatchery/wild fraction, age composition). Early estimates of steelhead in this DPS may be based on runs that were already depressed due to dams and steelhead fisheries. Nevertheless, these early dam counts are the best source of available data on the former size of the populations within this DPS. From 1933-1959 counts at Rock Island Dam averaged between 2,600 and 3,700 steelhead adults, which suggested the pre-fishery run size likely exceeded 5,000 adults destined for tributaries above Rock Island Dam (Chapman et al. 1994 in Busby et al. 1996). Using counts at Priest Rapids Dam (located below the production areas for this DPS) as an indicator of DPS size and trends suggests that the total number of spawners has increased since NMFS' 1996 status review. The 1992-1996 average annual total returns (hatchery plus natural) of steelhead spawners was 7,800, and the 1997-2001 average is 12,900 steelhead (hatchery plus natural). The natural component increased in these same periods from 1,040 to 2,200, respectively (Good et al. 2005).

Table 22. Upper Columbia River steelhead salmon populations and a summary of demographic data

Population	**Historical Abundance (Percent Annual change)[a]**	**Mean Number of Spawners (range)[b]**	Percent Hatchery Contribution[c]	Long-term growth rate (λ)[d]
Wenatchee River	2,500 (2.6)	3,279** (1,899-8,036)	71 (65)	1.067, 0.733
Entiat River				

Methow River Okanogan River	2,400* (-12.0)	3,714** (1,879-12,801)	91 (81)	1.086, 0.589

[a]Values of historical abundance represent total escapement as calculated in NMFS' first status review for the DPS. * = value represents a combined total escapement for the Methow and Okanogan rivers. Available data series: Wenatchee = 1962-1993, Methow and Okanogan = 1982-1993; calculations represent the geometric mean 1989-1993. Estimates are run reconstructions. Demographically independent populations identified by ICBTRT 2003.
[b]Geometric mean estimates of escapement represent total escapement (hatchery plus natural adult returns). ** Estimates of the mean number of spawners is a combined estimate for the Wenatchee and Entiat rivers, and the Methow and Okanogan rivers are also combined.
[c]Estimates of percentage of hatchery returns are from Good et al. 2005, and are based on extrapolations from mainstem dam counts and sampling. Parenthetical values are from Busby et al. 1996, and are provided for comparison.
[d]Multiple estimates for long-term growth (λ) are provided by Good et al. (2005) and represent two different assumptions on the contribution of hatchery fish to natural production. The first value reflects the assumption that hatchery fish do not contribute to natural production, and the second value reflects the assumption that hatchery fish contribute to natural production at the same rate as natural-origin spawners. Data series: 1976-2001.

While the total number of naturally produced fish in this DPS increased between status reviews, the proportion of naturally produced steelhead to hatchery-origin fish has declined. Total escapement increased in the combined estimate for the Wenatchee and Entiat rivers to a geometric mean of 3,279 spawners (900 natural spawners) over NMFS' previous estimate of 2,500 hatchery and natural steelhead spawners (1989 to 1993, natural component 800 steelhead). Estimates of the hatchery contribution to this population increased from 65% to 71% of total escapement (Table 22). A comparison of estimates for the Methow and Okanogan rivers during the same periods indicate that the total escapement increased from 2,400 to 3,714 while naturally produced steelhead declined from 450 to 358. Thus, the contribution of naturally produced steelhead declined from 19% to only 9% of total escapement between the 1993 and 2001 estimates (Good et al. 2005).

The assumptions of the role that hatchery fish play in the overall productivity and health of the DPS strongly influence estimates of population growth rates. Estimates based on the assumption that hatchery fish contribute to natural production at the same rate as natural-origin spawners consistently result in long-term population growth rates (expressed as λ) that are consistently below 1 (Table 22). Under the assumption that hatchery fish do not contribute to natural production, estimates of long term population growth rate suggest the population is growing. Determining the actual contribution of hatchery fish to natural production is important for understanding the true status of this DPS, particularly given that the proportion of naturally produced steelhead to hatchery-origin steelhead continues to decline. The extremely low replacement rate of naturally produced steelhead in this DPS is of concern, and the returns of natural steelhead remain well below recovery target levels.

The majority of the biological review team (54%) felt that this DPS warranted an "endangered" listing due to the growth rate and productivity, and uncertainty over the contribution of hatchery fish to natural production. NMFS, after convening a review of the artificial propagation programs of the six hatcheries in the DPS concluded that the programs collectively mitigate the immediacy of extinction risk in the DPS. Thus, NMFS listed the DPS as threatened rather than threatened (71 FR 834). NMFS concluded that the hatchery programs have increased total escapement and the distribution of spawning areas, and minimize the potential risks associated with artificial propagation. However, the abundance and productivity of naturally spawned steelhead remains a concern.

Critical Habitat

NMFS designated critical habitat for Upper Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Chief Joseph, Okanogan, Similkameen, Methow, Upper Columbia/Entiat, Wenatchee, Lower Crab, and the Upper Columbia/Priest Rapids subbasins, and the Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.

There are 42 watersheds within the range of Upper Columbia River steelhead. The total area of habitat designated as critical includes about 1,250 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 42 watersheds reviewed in NMFS' assessment of critical habitat for Upper Columbia River steelhead, three watersheds received a low rating of conservation value, eight received a medium rating, and 31 received a high rating of conservation value for the species. In addition, the Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Limiting factors identified for the Upper Columbia River steelhead include: mainstem Columbia River hydropower system mortality, reduced tributary stream flow, tributary riparian degradation and loss of in-river wood, altered tributary floodplain and channel morphology, and excessive fine sediment and degraded tributary water quality.

<u>Upper Willamette River Steelhead</u>

Distribution and Description of the Listed Species

The Upper Willamette River steelhead DPS includes all naturally spawned populations of winter-run steelhead in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River (inclusive). No artificially propagated populations that reside within the historical geographic range of this DPS are included in this listing. Hatchery summer-run steelhead occur in the Willamette Basin but are an out-of-basin population that is not included in this DPS.

The native (late) winter-run steelhead, with spring Chinook salmon, are the only two populations of salmon believed to historically occur above Willametter Falls (RKm 77). The construction of a fish ladder at the falls in the late 1880s, allowed for the passage of summer steelhead from Skamania Creek and winter-run steelhead from Big Creek (i.e., Gnat Creek). The two groups of winter-run steelhead exhibit different return times. The later run exhibits the historical phenotype adapted to passing the seasonal barrier that existed at Willamette Falls prior to construction of the fish ladder. The early run of winter-run steelhead are considered non-native, and were derived from Columbia River steelhead outside the Willamette River (Good et al. 2005). While the release of these hatchery winter-run fish was recently discontinued, some fish from earlier releases now reproduce naturally within the upper Willamette River Basin. Nonnative summer-run hatchery steelhead continue to be released within the upper basin (Good et al. 2005).

Native steelhead in the Upper Willamette are a late-migrating winter group that enters fresh water in January and February (Howell et al. 1985). They do not ascend to their spawning areas until late March or April (Dimick and Merryfield 1945) and spawning occurs from April to June 1. The smolt migration past Willamette Falls also begins in early April and proceeds into early June, peaking in early- to mid-May (Howell et al. 1985). Smolts generally migrate through the Columbia via Multnomah Channel rather than the mouth of the Willamette River. Most spend 2 years in the ocean before re-entering natal rivers to spawn (Busby et al. 1996). Steelhead in the Upper Willamette River DPS generally spawn once or twice, although some may spawn three times. Repeat spawners are predominantly female and generally account for less than 10% of the total run size (Busby et al. 1996).

Status and Trends

NMFS originally listed Upper Willamette River steelhead as threatened in 1999 (64 FR 14517), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). The Upper Willamette steelhead DPS consists of four demographically independent populations, each of which remains extant although depressed from historical levels (Table 23). Available data for this DPS comes from a combination of dam counts, redd count index surveys, and hatchery trap counts. Estimates of abundance from NMFS 1996 status review on this DPS, demonstrate a mix of trends with the largest populations, Mollala and North Santiam rivers, declining over the period of analysis. The 2005 review of the status of the Upper Willamette steelhead DPS indicated that each population showed a declining trend over the data series that extended to 2000 and 2001, while one population, the Calapooia River, increased over the short-term (1990-2000/1; Good et al. 2005).

More recently, data reported in McElhany et al. (2007) indicate that currently the two largest populations within the DPS are the Santiam River populations. Mean spawner abundance in both the North Santiam River and the South Santiam River is about 2,100 native winter-run steelhead. Long-term growth is negative for three of the populations within the DPS, with Calapooia River demonstrating a lambda of >1 indicating long-term growth in this population (McElhany et al. 2007). Spatial structure for the North and South Santiam populations has been substantially reduced by the loss of access to the upper North Santiam basin and the Quartzville Creek watershed in the South Santiam subbasin due dam construction lacking passage facilities (McElhany et al. 2007). Additionally, habitat in the Molalla subbasin has been reduced significantly by habitat degradation and in the Calapooia by habitat degradation as well as passage barriers. Finally, the diversity of some populations has been eroded by small population size, the loss of access to historical habitat, legacy effects of past winter-run hatchery releases, and the ongoing release of summer steelhead (McElhany et al. 2007).

Table 23. Upper Willamette river steelhead populations and a summary of available demographic data

Population[a]	Historical Abundance (Percent Annual change)[b]	Mean Number of Spawners (range)[c]	Long-term trend in redds per mile (95% CI)[d]	λ[e]
Mollala River	2,300 (-4.9)	914 (655-1275)	0.947 (0.918, 0.977)	0.988 (0.79, 1.235)
North Santiam River	2,000 (-4.0)	2,109 (1,485-2,994)	0.941 (0.906, 0.977)	0.983 (0.786, 1.231)
South Santiam River	550 (2.4)	2,149 (1,618-2,853)	0.936 (0.904, 0.907)	0.976 (0.855, 0.998)
Calapooia River	700	339 (206-560)	0.968 (0.933, 1.003)	1.023 (0.743, 1.409)

[a]Demographically independent populations identified by Myers et al. 2002 cited in Good et al. 2005.
[b]Values of historical abundance represent total escapement, with the exception of the Calapooia River which represents total run, as calculated in NMFS' first status review for the DPS. Data were collected using different methods (Angler Catch vs. Dam Counts) and represent data series ending in the early 1990s or earlier. Details on data types and the data series used for these calcuations are available in Busby et al. (1996).
[c]The geometric mean natural orgin spawner abudance calculated for the data series 1990-2005, and reported in McElhany et al. 2007.
[d]Long term trends are estimated using the entire data set, which is 1980 to 2000 for the Mollala River, and 1980-2001 for the remaining populations. Trends calculated by Good et al. 2005.
[e]Long-term growth rate (λ) reported by McElhany et al. 2007, and relects spawner escapement for the total available data series (1980-2005 – Molalla, Calappia & N Santiam Rivers; 1968-2005-S.Santiam River).

Critical Habitat

NMFS designated critical habitat for Upper Willamette River steelhead on September 2, 2005 (70 FR 52488). Designated critical habitat includes the following subbasins: Upper Willamette, North Santiam, South Santiam, Middle Willamette, Molalla/Pudding, Yamhill, Tualatin, and the Lower Willamette subbasins, and the lower Willamette/Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.

There are 38 watersheds within the range of Upper Willamette River steelhead. The total area of habitat designated as critical includes about 1,250 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 38 watersheds reviewed in NMFS' assessment of critical habitat for Upper Willamette River steelhead, 17 watersheds received a low rating of conservation value, six received a medium rating, and 15 received a high rating of conservation value for the species. In addition, the lower Willamette/Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value.

Marine Mammals

Cook Inlet Beluga Whale

Distribution and Description of the Listed Species

Beluga whales are widely distributed in Arctic and subarctic waters, and in Alaska five putative populations exist (Beaufort Sea, eastern Chukchi Sea, Bristol Bay, eastern Bering Sea, and Cook Inlet). Cook Inlet beluga whales are the only population that is listed under the ESA. Mitochondrial and nuclear DNA distinguish Alaskan beluga whales from those that occur in Hudson Strait, Baffin Bay and the St. Lawrence River, with the Cook Inlet population demonstrating the strong evidence of genetic isolation from the other Alaskan populations and other populations demonstrating weak to moderate evidence of genetic isolation (O'Corry-Crowe et al. 2007 in Hobbs et al. 2008; O'Corry-Crowe 2008; O'Corry-Crowe et al. 2008). Analysis of mtDN variation indicates strong philopatry to summering areas and low rates of dispersal between Cook Inlet beluga whales and other populations. The phylogenetic structure of the Cook Inlet beluga whale population suggests isolation of the population over evolutionary time scales.

Beluga whales are observed year-round in Cook Inlet although less is known about their winter movements than summer movements (see Hobbs et al. 2008 for a review). Data from satellite tagging studies suggest that movements of Cook Inlet beluga whales during summer months are short and largely focused around river estuaries and inlets (e.g., Chickaloon Bay, Turnagain Arm, Susitna River, and Knik Arm in the upper inlet and in many cases the animals exhibited very little movement for weeks during the summer (Hobbs et al. 2005). Dense groupings in these areas during June and July are the focus of NMFS aerial surveys, but numbers drop substantially in the upper inlet by November (Hobbs et al. 2005). Outside of Cook Inlet in the Gulf of Alaska beluga whale sightings are extremely rare (Laidre et al. 2000). Hobbs et al. (2005) found that tagged beluga whales moved to farther offshore during winter months, but remained within Cook Inlet. However travel distance appeared to increase during winter months, and exhibited more widely dispersed patterns both within and among individuals (Hobbs et al. 2005). Distribution during all months is likely influenced by prey distribution, where salmon and eulachon are concentrated in river mouths during summer months and other prey like sand lance are found in mid and bottom waters of the inlet during winter months, albeit in more dispersed patterns leading to the wider dispersal of the whales.

Based on past studies of the summer distribution of beluga whales in Cook Inlet, it appears that the population has experienced a contraction in its overall distribution (Speckman and Piatt 2000; Hobbs et al. 2008). Aerial surveys in the 1970s indicated that at least 10% of the population used areas south of Kenai River and Kalgin Island (mid- to lower Cook Inlet) during summer months, whereas more recent surveys (1993-2007) observed more than 90% of the beluga whales in upper Cook Inlet in shallow waters. According to Hobbs et al. (2008) 90% of the whales in the 1970s were observed within 70 nmi of the western tip of Anchorage (Point Woronzof), whereas more recently (1998-2007) 90% were detected within 20 nmi. Although the precise reason for the range contraction is not known, the shrinking summer distribution likely reflects the reduction in the population size over the same intervals and the beluga whale's preference for dense aggregations of preferred prey species.

Analyses of beluga whale stomach contents indicate that beluga whales are opportunistic feeders, but specific species form the bulk of the prey when they are seasonally abundant (Hobbs et al. 2008). For instance, eulachon (*Thaleichthys pacificus*) also known as smelt or candlefish, are a small anadromous fish return that their natal rivers in spring for spawning. In the Susitna River, the eulachon spawning migration has a bimodal peak, with fish entering the estuary in May and again in June, and represents a significant biomass of prey, with estimates of several thousand fish entering the river in the first wave and several million entering the river in June (Calkins 1989). The common name candlefish is derived from the fact the fish is so high in fat content during spawning, with up to 15% of total body weight as fat, that when caught and dried and strung on a wick the fish could be burned like a candle. This high fat content confers a significant source of energy for beluga whales, including calving whales that occur in the upper inlet during the same period (Calkins 1989). The stomach contents of one beluga whale harvested in upper Cook Inlet in 1998 near the Susitna River contained only eulachon. Based on stomach sample analyses from 2002-2007 fish compose the majority of the prey species, with gadids (cod and walleye pollock) and salmonids composing the majority of the fish eaten (Hobbs et al. 2008). Anadromous salmonids begin concentrating at the river mouths and intertidal flats in upper Cook Inlet in late spring and early summer as emigrating smolts and immigrating adult spawners. Like eulachon, salmon are another source of lipid-rich prey for the beluga whale and represent the greatest percent frequency of occurrence of the prey species found in Cook Inlet beluga whale stomachs (Hobbs et al. 2008). As salmonid numbers dwindle in the fall and winter, beluga whales return to feed on nearshore or deeper water species including cod, sculpin, flounder, sole, shrimp, crab and others (Hobbs et al. 2008).

Cook Inlet experiences some of the most extreme tidal fluctuations in the world (see NMFS 2008 for a discussion), and beluga whales in the inlet have adapted to these tidal cycles and seemingly take advantage of them, although the precise causal reasons are not well known. Presumably, the feeding opportunities these tidal cycles proffer the beluga whale are a contributing factor. Aerial surveys and predictive models of habitat us indicate that beluga whale movement patterns are closely correlated to tidal patterns, flow accumulation and mudflats, with a preference for medium and high flow inlets of larger river basins (Ezer et al 2008; Goetz et al. 2007). More information, however, is needed to link these habitat attributes to causative reasons for this preference. Besides feeding, studies have suggested this preference for tidal mudflats may also be attributed to calving and breeding, molting, or shelter from predators like killer whales (Calkins 1989; Huntington 2000; Moore et al. 2000; Sheldon et al. 2003).

Beluga whale calving is not well documented but the presence of cow/calf pairs in large river estuaries in the upper inlet, and accounts of Alaskan Natives, suggests that calving and nursery areas are located near the mouths of the Beluga and Susitna Rivers, Chickaloon Bay and Turnagain Arm (NMFS 2008). According to surveys by LGL (Funk et al. 2005 as cited in NMFS 2008) cow/calf pairs also make extensive use of Knik Arm in the summer and fall. Neonates are often not seen until June in Cook Inlet (Burns and Seaman 1986a). NMFS (2008) and others have suggested that the shallow waters of Cook Inlet may be important for reproduction and calving, as the shallower water is warmer which may confer an important thermal advantage for calf survival as they have relatively limited fat deposits at birth. Breeding is presumed to occur shortly after calving, in the late summer with a female's first parturition at age 5 or 6 after about 14-15 months of gestation (Calkins 1989). Lactation lasts about two years, with breeding occurring during lactation (Calkins 1989).

Calculation of beluga whale age is based on growth layers in teeth. Some debate exists as to whether a beluga whale tooth contains two growth layer groups (GLG) per year or one growth layer per year (See Hobbs et al. 2008 for discussion). Due to this ambiguity, Hobbs et al. (2008) summarized life history parameters according to tooth growth layers rather than years (Table 24 from Hobbs et al. 2008).

Table 24. Review of Female beluga life history parameters found in the published literature (from Hobbs et al. 2008; GLG=growth layer groups)

Parameter	Data	Sources
Age at sexual maturity	8-15 GLG	Brodie 1971; Sergeant 1973; Ognetov 1981; Seaman and Burns 1981; Braham 1984; Burns and Seaman 1986
	0% at 8-9 GLGs 33% at 10-11 GLGs 94% at 12-13 GLGs	Burns and Seaman 1986[a]
	9.1 +/- 2.8 GLGs	Robeck et al. 2005
Age at color change (gray to white)	12 GLGs	Brodie 1971
	22 GLGs	Sergeant 1973
Age at 1st conception	54% at 8-9 GLGs 41% at 10-11 GLGs 94% at 12-13 GLGs	Burns and Seaman 1986[b]
Age at senescence	42-43 GLGs	Brodie 1971
Pregnancy and birth rates	Small fetuses: 0.055 at 0-11 GLGs 0.414 at 12-21 GLGs 0.363 at 22-45 GLGs 0.267 at 46-57 GLGs 0.190 at 58-77 GLGs With full-term fetuses/neonates: 0.000 at 0-11 GLGs 0.326 at 12-21 GLGs 0.333 at 22-45 GLGs 0.278 at 46-51 GLGs 0.182 at 52-57 GLGs 0.125 at 58-77 GLGs	Burns and Seaman 1986
Lifespan	>60 GLGs (Oldest female estimated at 70+ GLGs)	Burns and Seaman 1986
	64-65 GLGs	Khuzin 1961 (cited in Ohsumi 1979)
	60-61 GLGs	Brodie 1971
	50-51 GLGs	Sergeant 1973
Adult annual survival	0.96-0.97	Béland et al. 1992
	0.955 (based on pilot whale data)	Brodie et al. 1981
	0.935	Lesage and Kingsley 1998
	0.91-0.92	Allen and Smith 1978
	0.906 (includes natural & human-caused mortality)	Burns and Seaman 1986

Parameter	Data	Sources
	0.84-0.905 (based on body length and lifespan	Ohsumi 1979
Immature annual survival	0.905 (for neonates in first half year)	Sergeant 1973
Reproductive rate	0.010-012	Perrin 1982[c]
	0.11[d]	Burns and Seaman 1986
	0.13[d]	Sergeant 1973
	0.09[d]	Brodie 1971
	0.09-0.12 [d]	Braham 1984
	0.09-0.14[e]	Braham 1984
	0.12 [e]	Sergeant 1973; Ray et al. 1984
	0.08-0.14[e]	Davis and Evans 1982
	0.06-0.10[e]	Davis and Finley 1979
	0.08-0.10[e]	Brodie et al. 1981
	0.08 (unknown)	Breton-Provencher 1981
Calving Interval	<3 years	Burns and Seaman 1986[f]
	2 yrs and 3 years	Sergeant 1973[g]

[a]Alaska sample (52 whales). Sampling occring in June when most Alaskan beluga whales are born. Hobbs et al. 2008 note that it is possible that non-pregnant 8-9 GLGs beluga whales would have conceived before their 10-11 GLG birth date.
[b]Alaska sample of 22 whales.
[c]Based on literature review and adopted by the International Whaling Commission
[d]Based on annual calf production rates
[e]Based on calf counts
[f]For some female beluga whales. This was a tentative conclusion based on high conception rates noted in some females between the ages of 12-13 GLGs and 44-45 GLGs.
[g]Two-year intervals were for 25% of mature female belugas in eastern Canada (7 of 29 sampled); presumed after noting pregnancies occurred during lactation. Three-year intervals were for 75% of mature females in eastern Canada. Sergeant (1973) concluded that the “overlap of pregnancy and previous lactation is infrequent so that calving occurs about once in three years.”

Status

On October 22, 2008, NMFS listed the Cook Inlet beluga whale as endangered (73 FR 62919). Historic numbers of beluga whales in Cook Inlet are unknown. Dedicated surveys began in earnest in the 1990s when NMFS began conducting aerial surveys for beluga whales in Cook Inlet. Prior to then, survey efforts were inconsistent, part of larger sea bird and marine mammal surveys, made by vessel, or estimated following interviews with fishermen (Klinkhart 1966). In many cases the survey methodology or confidence intervals were not described. For instance, Klinkhart (1966) conducted aerial surveys in 1964 and 1965, where he describes having estimated the populations at 300-400 whales, but the methodology was not described nor did he report the variance around these estimates. Other estimates were incomplete due to the small area the survey focused upon (e.g. river mouth estimates; e.g., Hazard 1988). The most comprehensive survey effort prior to the 1990s occurred in 1979 and included transects from Anchorage to Homer, and covered the upper, middle and lower portions of Cook Inlet. From this effort, and using a correction factor of 2.7 to account for submerged whales Calkins (1989 cited in NMFS 2008) estimated the 1979 abundance at about 1,293 whales.

In 1993, NMFS began systematic aerial surveys of beluga whales in Cook Inlet and like the 1979 survey cover the upper, middle and lower portions of Cook Inlet. The survey protocol involves using paired observers who make independent counts at the same time a video of the whale grouping is recorded. Each group size estimate is corrected for subsurface and missed animals, or if video counts are not available then additional corrections are made (Allen and Angliss 2010).

Table 25. Estimated abundance of Cook Inlet beluga whales with coefficient of variation and 95% confidence intervals.

Year	**Estimate**[1]	**CV**	**95% CI**[2]	
			Lower	**Upper**
1979	1,293			
1994	653	0.43	291	1464
1995	491	0.44	215	1120
1996	594	0.28	347	1018
1997	440	0.14	335	578
1998	347	0.29	199	606
1999	367	0.14	279	482
2000	435	0.23	279	679
2001	386	0.087	326	458
2002	313	0.12	248	396
2003	357	0.107	290	440
2004	366	0.2	290	440
2005	278	0.18	196	394
2006	302	0.16	221	412
2007	375	0.14	285	492
2008	375	0.23	240	585
2009[2]	321	0.18	226	456

[1]All estimates, except 1979 estimate, reported in Hobbs & Shelden 2008. The 1979 estimate is from Calkins 1989 as cited in NMFS 2008.
[2]Data from R. Hobbs, pers. comm., to A. Garrett, Apr. 2010.

Between 1979 and 1994, according to above noted population estimates, Cook Inlet beluga whales declined by 50%, with another 50% decline observed between 1994 and 1998. Using a growth fitted model Hobbs et al. 2008 observed an average annual rate of decline of -2.91% (SE = 0.010) from 1994 to 2008, and a -15.1% (SE=0.047) between 1994 and 1998. A comparison with the 1999-2008 data suggests the rate of decline at -1.45% (SE=0.014) per year (Hobbs et al. 2008). Given that harvest was curtailed significantly between 1999 and 2008, NMFS had expected the population would begin to recover at a rate of 2-6% per year. However, abundance estimates demonstrate that this is not the case (Hobbs & Shelden 2008).

In conducting its status review, NMFS ran a number of population viability analyses (PVAs) to estimate the time to extinction for Cook Inlet beluga whales. The models were sensitive to a variety of parameters such as killer whale predation, allee effects, and unusual mortality events. The best approximation of the current population incorporated killer whale predation at only one beluga whale per year, and allowed for an unusual mortality event occurring on average every 20 years. According to this model, there is an 80% probability that the population is declining, a 26% probability that the population will be extinct in 100 years (by 2108) and a 70% probability that the population will be extinct within 300 years (by 2308).

Social Behavior

Beluga whales are highly social animals. The highly developed vocal repertoire of the beluga whale may play a substantial role in the formation of groups and communication among individuals. According to O'Corry-Crowe (2002), the beluga whale has long been called the "sea canary" by mariners because of the wide variety of sounds they make and can be heard reverberating through ship hulls. About 50 types of calls are recognized, typically ranging from

0.1 to 12 kHz, and include groans, whistles, buzzes, trills, roars and others, allow them to communicate over long distance and through icy arctic waters.

Belugas are typically observed in groups, which typically range from 2-25 individuals although they have been observed in groups of hundreds and even up to a thousand animals. There may be some seasonal segregation of sexes, as at times males form distinct groups and females are often tightly associated with one or more generations, at other times the groupings are a mixed social unit (O'Corry-Crowe 2002). Beluga whales also have a wide variety of facial expressions, as they can alter the shape of the mouth and melon. The lateral flexibility allows them to exploit shallow habitats and likely enhances visual signaling between animals (like vocalization, visual acuity is highly developed).

Threats

Natural Threats. Natural threats to Cook Inlet beluga whales include stranding, predation, parasitism and disease, environmental change, and genetic risks associated with small populations (e.g., inbreeding, loss of genetic variability). Beluga whales may strand accidentally as they occupy shallow water areas or escape predators, or as a result of diseases, illness or injury (NMFS 2008). Given the extreme tidal fluctuations in Cook Inlet, beluga whale strandings are not uncommon. According to NMFS (2008) killer whales have been observed in Cook Inlet concurrent to beluga whale strandings, and evidence of killer whale attacks is apparent in some beluga whale strandings (see Table 26).

According to NMFS (2008) over 700 beluga whales have stranded in Cook Inlet since 1988, many of which occurred in Turnagain Arm and often coincided with extreme tidal fluctuations (see Table 26 for a complete record). Where stranding occurs from extreme tidal fluctuations, and animals are out of the water for extended periods the risk of mortality increases from cardiovascular collapse. Ten hours may be the upper limit for out of the water for beluga whales before serious injury or death occurs (NMFS 2008). Strandings may represent a significant threat to the conservation and recovery of the Cook Inlet beluga whale population.

Table 26. Cook Inlet beluga whale stranding records from 1988 through September 2008 (from Hobbs and Shelden 2008, and NMFS 2008).

Year	Month	Location	No. w/evidence of Killer whale predation	Number of Whales	Known Associated Deaths	Total Mortalities* (live + dead stranded)
1988	October	Turnagain Arm		27	0	0
1989	-	-		-	-	4
1988	-	-		-	-	2
1991	August	Turnagain Arm	1	70-80	0	2
1992	October	Kenai River	2	2	2	5
1993	July	Turnagain Arm	1	10+	0	3
1994	June	Susitna River		186	0	7
1995	-	-		-	-	2
1996	June	Susitna River		63	0	12
	August	Turnagain Arm		60	4	
	September	Turnagain Arm		20-30	1	
	September	Knik Arm		1	0	

Year	Month	Location	No. w/evidence of Killer whale predation	Number of Whales	Known Associated Deaths	Total Mortalities* (live + dead stranded)
	October	Turnagain Arm		10-20	0	
1997	-	-		-	-	3
1998	May	Turnagain Arm		30	0	10
	September	Turnagain Arm		5	0	
1999	August	Turnagain Arm	5	58	5	12
	September	Turnagain Arm		12-13	0	
2000	August	Turnagain Arm	2	8	0	13
	September	Turnagain Arm		15-20	0	
	October	Turnagain Arm		1-2	0	
2001	-	-		-	-	10
2002	-	-		-	-	13
2003	April	Turnagain Arm	1	2	0	20
	August	Turnagain Arm		46+	5	
	September	Turnagain Arm		58	0	
	October	Turnagain Arm		9		
2004	-	-		-	-	13
2005	August	Knik Arm		6	1	6
2006	September	Knik Arm		12	0	8
2007	-	-		-	-	15
2008	August	Knik Arm	1	28-30	2	11

*Known subsistence harvested beluga whales are not included in these numbers.

Gaydos et al. (2004) identified 16 infectious agents in free-ranging and captive southern resident killer whales, but concluded that none of these pathogens were known to have high potential to cause epizootics. Many of these same infectious agents could pose a problem for Cook Inlet beluga whales. At this time little information is available to date to suggest bacterial or viral agents are actively contributing to the decline in the Cook Inlet population. About 80% of Cook Inlet beluga whales examined, however, have evidence of the parasite *Crassicauda giliakiana* in the kidneys, although it is presently unclear whether the parasite is affecting the status of the population (NMFS 2008). Necropsies have also revealed infestations of the common nematode anasakids, or whaleworm in the stomach of adult Cook Inlet beluga whales. While the parasite tends to favor the stomach and can cause gastritis or ulcerations, the infestations in beluga whales has not been considered severe enough to have caused clinical responses (NMFS 2008). Liver trematodes have also been identified in at least one beluga whale. At present, NMFS has no information to suggest that parasites are having a measureable impact on the survival and health of the Cook Inlet whale population (NMFS 2008).

Anthropogenic Threats. Human induced threats to Cook Inlet beluga whales include subsistence harvest, poaching and illegal harvest, incidental take during commercial fishing and reduction of prey through fishing harvests, pollution, oil and gas development, urban development, vessel traffic including from tourism and whale watching, noise, as well as research activities directed at beluga whales. Subsistence harvest of beluga whales by Alaskan natives has occurred since prehistoric times, but the effect of recent harvests has been significant. Although harvest levels have only recently been recorded, population declines in the mid 1990s are largely attributed to subsistence harvests during that period. In part, improved efficiencies of harvest techniques has allowed natives and others to increase catch of beluga whales. During the early 1900s there was a short-lived commercial whaling company, The Beluga Whaling Company, which operated at the Beluga River in upper Cook Inlet. The Company during its 5 years of operation harvest 151 belugas from 1917-1921 (Mahoney and Shelden 2000). Another commercial hunt of beluga whales in 1930s is recollected by residents, but no record of the hunt exists in Alaska fishery and fur seal documents (Bower, 1931-41 as cited in Mahoney and Shelden 2000). In 1999 and 2000 there was a voluntary moratorium on subsistence harvest, and since substance harvest have been conducted under co-management agreements. Since 2000, no more than 2 beluga whales have been taken in subsistence harvests in any one year (NMFS 2008).

Commercial fisheries likely have varying levels of interactions with Cook Inlet beluga whales, according to the timing, gear types, targeted species, and location of activities (NMFS 2008). Reports of fatal interactions with commercial fisheries have been noted in the literature (Murray and Fay 1979 cited in Hobbs et al. 2008; Burns and Seaman 1986). Direct interactions with fishing vessels and nets are considered unusual, based on observer data, and unlikely to inhibit the recovery of Cook Inlet beluga whales. The reduction of prey species, however, is of more concern for the species. In 2000 NMFS recommended the closing of the eulachon fishery due to a lack of understanding of how this fishery interfered with beluga whale feeding, but in 2005 this fishery was reopened with a harvest limited at 100 tons of eulachon. Currently, it is unclear if fishery harvest of beluga whale prey species is having a significant impact on the population. Impacts from recreational fisheries, which are very popular in the region, likely include the reduction of fish prey species particularly salmonid species, and also the harassment from noise and risk of injury from vessel strikes from the operation of small watercraft in the estuarine/river mouths may (NMFS 2008).

Contaminants in beluga whales are of concern, both for whale health and the health of subsistence users. Tissue samples are regularly collected from subsistence harvested and stranded beluga whales and archived. Tissues and organs commonly collected include blubber, liver and kidneys, as well as muscle, heart, bone, skin and brain. Blubber is the most commonly collected; due to the lipid content it typically contains the most lipophilic substances (Becker 2000). The kidney and liver are used to analyze heavy metal compounds. Relatively high levels of PCBs, chlorinated pesticides and mercury are evident in beluga whales, although the more contaminated belugas are from the St. Lawrence River, Canada (Becker 2000). Concentrations of chlorinated hydrocarbons in Cook Inlet beluga whales range from 0.1-2.4 µg/g, w.w. DDT, 0.6-4.7 µg/g, w.w. PCB, 0.1-0.6 µg/g, w.w. chlordane, <0.1-4.3 µg/g, w.w. toxaphene. The higher levels of these compounds found in beluga whales in comparison to bowhead whales is probably reflective of the trophic levels of the species, as bowhead are baleen whales that feed on copepods while belugas are primarily fish eaters (Becker 2000). Studies indicate that PCBs and chlorinated pesticide concentrations are higher in male beluga whales than females, reflecting the transference of body loads to the offspring that occurs during gestation and lactation (Becker et al. 2000). Other contaminant detected in Cook Inlet beluga whales include heavy metals such as cadmium, mercury, selenium, copper, and zinc to name a few. Comparative studies suggest that Cook Inlet beluga whales generally carry less body burdens than beluga whales from other areas. An exception is copper, which is two to three times higher in Cook Inlet beluga whales than beluga whales from the eastern Beaufort Sea and the eastern Chukchi Sea, but is similar concentrations found in Hudson Bay beluga whales (Becker et al. 2000). To date, the health implications of high copper levels in Cook Inlet beluga whales is not clear.

Critical Habitat

NMFS proposed critical habitat for the Cook Inlet beluga whale on December 2, 2009 (74FR 63080). Two areas specific areas are proposed comprising 7,809 square kilometers of marine habitat. Area 1 encompasses 1,918 square kilometers (741 sq. mi.) of Cook Inlet northeast of a line from the mouth of Threemile Creek (61º 08.5'N., 151 º 04.4' W.) to Point Possession (61º 02.1'N., 150 º 24.3' W.). This area is bounded by Anchorage, the Matansuska-Susitna Borough, and the Kenai Peninsula Borough. This area contains shallow tidal flats, river mouths or estuarine areas and is important as foraging and calving habitats. Area 1 also has the highest concentrations of beluga whales in the spring through fall as well as the greatest potential for adverse impact from anthropogenic threats. Area 1 contains many rivers with large eulachon and salmon runs, including 2 rivers in Turnagain Arm (Twenty-mile River and Placer River) which are visited by beluga whales in the early spring. Use declines in the summer and increases again in August through the fall, coinciding with coho salmon returns. Also included in Area 1 is Knik Arm and the Susitna delta. Area 2 consists of 5,891 square kilometers (2,275 sq. mi.) of Cook Inlet, located south of Area 1, north of a line at 60º 25.0'N., and includes nearshore areas south of 60º 25.0'N. along the west side of the Inlet and Kachemak Bay on the east side of the lower inlet. Area 2 is used by Cook Inlet beluga whales in a dispersed fashion for fall and winter feeding and as transit waters. Area 2 includes near and offshore areas of the mid and upper Inlet, and nearshore areas of the lower Inlet. Area 2 includes Tuxedni, Chinitna, and Kamishak Bays on the west coast and a portion of Kachemak Bay of the east coast. Dive studies indicate that beluga whales in this area dive to deeper depths and are at the surface less frequently than they are when they inhabit Area 1. The primary constituent elements essential to the conservation of Cook Inlet beluga whales are: (1) intertidal and subtidal waters of Cook Inlet with depths <30 ft. (MLLW) and within 5 miles of high and medium flow accumulation anadromous fish streams; (2) primary prey species consisting of four species of Pacific salmon (Chinook, coho, sockeye, and chum salmon), Pacific eulachon, Pacific cod, walleye pollock, saffron cod, and yellowfin sole; (3) the absence of toxins or other agents of a type or amount harmful to beluga whales; (4) Unrestricted passage within or between the critical habitat areas; and (5) absence of in-water noise at levels result in the abandonment of habitat by Cook Inlet beluga whales. The comment period on this proposed rule closed on February 1, 2010.

Southern Resident Killer Whale

Distribution and Description of the Listed Species

Three kinds of killer whales occur along the Pacific Coast of the United States: Eastern North Pacific (ENP) southern resident killer whales, ENP Offshore killer whales, and ENP transient killer whales. Of these only the southern resident killer whales are listed as endangered or threatened under the ESA. Southern resident killer whales primarily occur in the inland waters of Washington State and southern Vancouver Island, although individuals from this population have been observed off the Queen Charlotte Islands (north of their traditional range) and off coastal California in Monterey Bay, near the Farallon Islands, and off Point Reyes (NMFS 2005; BOR 2008).

Southern resident killer whales spend a significant portion of the year in the inland waterways of the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound, particularly during the spring, summer, and fall, when all three pods regularly occur in the Georgia Strait, San Juan Islands, and Strait of Juan de Fuca (Heimlich-Boran 1988; Felleman et al. 1991; Olson 1998; Osborne 1999). The K and L pods typically arrive in May or June and remain in this core area until October or November, although both pods make frequent trips lasting a few days to the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000). The J pod will occur intermittently in the Georgia Basin and Puget Sound during late fall, winter and early spring. During the warmer months, all of the pods concentrate their activities in Haro Strait, Boundary Passage, the southern Gulf Islands, the eastern end of the Strait of Juan de Fuca, and several localities in the southern Georgia Strait (Heimlich-Boran 1988; Felleman et al. 1991; Olson 1998; Ford et al. 2000).

Southern resident killer whales are fish eaters, and predominantly prey upon salmonids, particularly Chinook salmon, but are also known to consume more than 20 other species of fish and squid (Scheffer and Slipp 1948; Ford et al. 1998; Ford et al. 2000; Saulitis et al. 2000; Ford and Ellis 2005; Ford and Ellis 2006;). Throughout inland waters from May to September, southern resident killer whale diet is approximately 88% Chinook salmon, with a shift to chum salmon in fall. Chum salmon are also taken in significant amounts (11%), especially in autumn (Hanson et al. 2005; Ford and Ellis 2006; Hanson et al. 2007b). Chinook salmon are preferred despite much lower abundance in comparison to other salmonids (such as sockeye) presumably because of the species' large size, high fat and energy content, and year-round occurrence in the area. Killer whales also capture older (i.e., larger) than average Chinook salmon (Ford and Ellis 2006). Little is known about the winter and early spring diet of southern residents. Early results from genetic analysis of fecal and prey samples indicate that Southern Residents consume Fraser River-origin Chinook salmon, as well as salmon from Puget Sound, Washington and Oregon coasts, the Columbia River, and Central Valley of California (Hanson et al. 2007a). However, recent studies suggest that members of L pod have undergone dietary shifts from Chinook salmon during fall months over the past decade (Krahn et al. 2009).

The local movements of southern resident killer whales usually follow the distribution of salmon (Heimlich-Boran 1986a, 1988, Nichol and Shackleton 1996). Areas that are major corridors for migrating salmon, and therefore, for southern resident killer whales, include Haro Strait and Boundary Passage, the southern tip of Vancouver Island, Swanson Channel off North Pender Island, and the mouth of the Fraser River delta, which is visited by all three pods in September and October (Felleman et al. 1991, Ford et al. 2000, K.C. Balcomb, unpublished data).

Female southern resident killer whales give birth to their first surviving calf between the ages of 12 and 16 years (mean ~ 14.9 years) and produce an average of 5.4 surviving calves during a reproductive life span lasting about 25 years (Matkin et al. 2003; Olesiuk et al. 1990). Females reach a peak of reproduction around ages 20-22 and decline in calf production gradually until reproductive senescence (Ward et al. 2009a). Older mothers tend to have greater calving success than do their younger, less-experienced counterparts (Ward et al. 2009b). Calving success also appears to be aided by the assistance of grandmothers (Ward et al. 2009b). The mean interval between viable calves is four years (Bain 1990). Males become sexually mature at body lengths ranging from 17 to 21 feet, which corresponds to between the ages of 10 to 17.5 years (mean ~ 15 years), and are presumed to remain sexually active throughout their adult lives (Christensen

1984; Duffield and Miller 1988; Olesiuk et al. 1990; Perrin and Reilly 1984). Most mating is believed to occur from May to October (Matkin et al. 1997; Nishiwaki 1972; Olesiuk et al. 1990). However, conception apparently occurs year-round because births of calves are reported in all months. Newborns measure seven to nine feet long and weigh about 200 kg (Clark et al. 2000; Ford 2002; Nishiwaki and Handa 1958; Olesiuk et al. 1990). Mothers and offspring maintain highly-stable, life-long social bonds and this natal relationship is the basis for a matrilineal social structure (Baird 2000; Bigg et al. 1990; Ford et al. 2000). Some females may reach 90 years of age (Olesiuk et al. 1990).

Southern resident killer whales spend a significant portion of the year in the inland waterways of the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound, particularly during the spring, summer, and fall, when all three pods are regularly present in the Georgia Basin (defined as the Georgia Strait, San Juan Islands, and Strait of Juan de Fuca) (Felleman et al. 1991; Heimlich-Boran 1988; Olson 1998; Osborne 1999). Typically, K and L pods arrive in May or June and primarily occur in this core area until October or November. During this stay, both pods also make frequent trips lasting a few days to the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000); however, J pod's movements differ considerably and are present only intermittently in the Georgia Basin and Puget Sound. Late spring and early fall movements of Southern Residents in the Georgia Basin have remained fairly consistent since the early 1970s, with strong site fidelity shown to the region as a whole (NMFS 2005b). During late fall, winter, and early spring, the ranges and movements of the southern residents are less well known. Offshore movements and distribution are largely unknown for the southern resident population.

While the southern residents are in inland waters during the warmer months, all of the pods concentrate their activities in Haro Strait, Boundary Passage, the southern Gulf Islands, the eastern end of the Strait of Juan de Fuca, and several localities in the southern Georgia Strait (Felleman et al. 1991; Ford et al. 2000; Heimlich-Boran 1988; Olson 1998). Individual pods are similar in their preferred areas of use, although there are some seasonal and temporal differences in certain areas visited (Olson 1998). For example, J pod is the only group to venture regularly inside the San Juan Islands. The movements of southern resident killer whales relate to those of their preferred prey, salmon. Pods commonly seek out and forage in areas where salmon occur, especially those associated with migrating salmon (Heimlich-Boran 1986; Heimlich-Boran 1988; Nichol and Shackleton 1996).

Members of different pods do interact, but members generally remain within their matrilinear group (Parsons et al. 2009). However, additional interaction between pods has occurred over the past two decades, possibly in association with the decline of the Southern Resident population as a whole (Parsons et al. 2009).

Population Structure

Southern resident killer whale DPS consists of three pods, or stable familial groups: the J pod, K pod, and L pod. The J pod is seen most frequently along the western shore of San Juan Island and is the only pod observed regularly in Puget Sound throughout winter (Heimlich-Boran 1988; Osborne 1999). The K pod is most frequently observed during May and June when they occur along the western shore of San Juan Island while searching for salmon. The L pod is the largest of the three pods (Ford et al. 1994) and frequently breaks off into separate subgroups.

Status

Southern resident killer whales were listed as endangered under the ESA in 2005 (70 FR 69903). In the mid- to late-1800s, southern resident killer whales were estimated to have numbered around 200 individuals. By the mid-1960s, they had declined to about 100 individuals. As discussed in the preceding section, between 1967 and 1973, 43 to 47 killer whales were removed from the population to provide animals for displays in oceanaria and the population declined by about 30 percent as a result of those removals. By 1971, the population had declined to about 67 individuals. Since then, the population has fluctuated between highs of about 90 individuals and lows of about 75 individuals.

At population sizes between 75 and 90 individuals, we would expect southern resident killer whales to have higher probabilities of becoming extinct because of demographic stochasticity, demographic heterogeneity (Coulson et al. 2006; Fox et al. 2006) —including stochastic sex determination (Lande et al. 2003) — and the effects of phenomena interacting with environmental variability. Demographic stochasticity refers to the randomness in the birth or death of an individual in a population, which results in random variation on how many young that individuals produce during their lifetime and when they die. Demographic heterogeneity refers to variation in lifetime reproductive success of individuals in a population (generally, the number of reproductive adults an individual produces over their reproductive lifespan), such that the deaths of different individuals have different effects on the growth or decline of a population (Coulson et al. 2006). Stochastic sex determination refers to the randomness in the sex of offspring such that sexual ratios in population fluctuate over time (Melbourne and Hastings 2008). For example, the small number of adult male southern resident killer whales might represent a stable condition for this species or it might reflect the effects of stochastic sex determination. Regardless, a high mortality rates among adult males in a population with a smaller percentage of males would increase the imbalance of male-to-female gender ratios in this population and increase the importance of the few adult males that remain.

At these population sizes, population’s experience higher extinction probabilities because stochastic sexual determination leaves them with harmful imbalances between the number of male or female animals in the population (which occurred to the heath hen and dusky seaside sparrow just before they became extinct), or because the loss of individuals with high reproductive success has a disproportionate effect on the rate at which the population declines (Coulson et al. 2006). In general, an individual’s contribution to the growth (or decline) of the population it represents depends, in part, on the number of individuals in the population: the smaller the population, the more the performance of a single individual is likely to affect the population’s growth or decline (Coulson et al. 2006). Given the small size of the southern resident killer whale population, the performance (= “fitness,” measured as the longevity of individuals and their reproductive success over their lifespan) of individual whales would be expected to have appreciable consequences for the growth or decline of the southern resident killer whale population.

These phenomena would increase the extinction probability of southern resident killer whales and amplify the potential consequences of human-related activities on this species. Based on their population size and population ecology (that is, slow-growing mammals that give birth to single calves with several years between births), we assume that southern resident killer whales would have elevated extinction probabilities because of exogenous threats caused by anthropogenic activities that result in the death or injury of individual whales (for example, ship strikes or entanglement) and natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) *as well as* endogenous threats resulting from the small size of their population. Based on the number of other species in similar circumstances that have become extinct (and the small number of species that have avoided extinction in similar circumstances), the longer southern resident killer whales remain in these circumstances, the greater their extinction probability becomes.

Social Behavior

Killer whales are highly social animals that occur primarily in groups or pods of up to 40-50 animals (Dahlheim and Heyning 1999; Baird 2000). Mean pod size varies among populations, but often ranges from 2 to 15 animals (Kasuya 1971; Condy et al. 1978; Mikhalev et al. 1981; Braham and Dahlheim 1982; Dahlheim et al. 1982; Baird and Dill 1996). Larger aggregations of up to several hundred individuals occasionally form, but are usually considered temporary groupings of smaller social units that probably congregate near seasonal concentrations of prey, for social interaction, or breeding (Dahlheim and Heyning 1999; Baird 2000; Ford et al. 2000).

In terms of gender and age composition, southern and northern resident killer whales social groups consisted of 19 percent adult males, 31 percent adult females, and 50 percent immature whales of either sex in 1987 (Olesiuk et al. 1990a). This composition is comparable with the composition of southern Alaska resident killer whales and killer whale populations in the Southern Ocean (Matkin et al. 2003; Miyazaki 1989).

Threats

Natural Threats. Southern resident killer whales like many wild animal populations (Nettles, 1992), experience highest mortality in the first year age class (Olesiuk et al. 1990; Krahn et al. 2002), although the reasons for these mortalities are still uncertain. The causes could include poor mothering, infectious or non-infectious diseases, and infanticide (Gaydos et al. 2004).

Gaydos et al. (2004) identified 16 infectious agents in free-ranging and captive southern resident killer whales, but concluded that none of these pathogens were known to have high potential to cause epizootics. They did, however, identify pathogens in sympatric odontocete species that could threaten the long-term viability of the small southern resident population.

Anthropogenic Threats. Several human activities appeared to contribute to the decline of southern resident killer whales. Southern resident killer whales were once shot deliberately in Washington and British Columbia (Scheffer and Slipp 1948; Pike and MacAskie 1969; Olesiuk et al. 1990; Baird 2001). Until 1970, about 25 percent of the killer whales that were captured for aquaria had bullet scars (Hoyt 1990). The effect of these attacks on individual whales or the population itself remains unknown. However, between 1967 and 1973, 43 to 47 killer whales were removed from the population for displays in oceanaria; because of those removals, the southern resident killer whale population declined by about 30%. By 1971, the population had declined to about 67 individuals. Since then, the population has fluctuated between highs of about 90 individuals and lows of about 75 individuals.

Over the same time interval, southern resident killer whales have been exposed to changes in the distribution and abundance of their prey base (primarily Pacific salmon) which has reduced their potential forage base, potential competition with salmon fisheries, which reduces their realized forage base, disturbance from vessels, and persistent toxic chemicals in their environment. The primary prey of killer whales, salmon, has been severely reduced due to habitat loss and overfishing of salmon along the West Coast (NRC 1996;Slaney et al. 1996; Gregory and Bisson 1997; Lichatowich 1999; Lackey 2003; Pess et al. 2003; Schoonmaker et al. 2003;). Several salmon species are currently protected under the ESA, and are generally well below their former numbers. A 50% reduction in killer whale calving has been correlated with years of low Chinook salmon abundance (Ward et al. 2009a).

Puget Sound also serves as a major port and drainage for thousands of square kilometers of land. Contaminants entering Puget Sound and its surrounding waters accumulate in water, benthic sediments and organisms (Krahn et al. 2009). Exposure to contaminants may harm southern resident killer whales. The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame –retardants have been documented in southern resident killer whales (Ross et al. 2000; Ylitalo et al. 2001; Herman et al. 2005; Ross 2006). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Kakushke and Prange 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal’s tissues (Martineau 2007). Because of their long life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of contaminants.

Since the 1970s commercial shipping, whale watching, ferry operations, and recreational boat traffic have increased in Puget Sound and the coastal islands of southern British Columbia. This traffic exposes southern resident killer whales to several threats that have consequences for the species’ likelihood of avoiding extinction and recovering if it manages to avoid extinction. First, these vessels increase the risks of southern resident killer whales being struck, injured, or killed by ships. In 2005, a southern resident killer whale was injured in a collision with a commercial whale watch vessel although the whale subsequently recovered from those injuries. However, in 2006, an adult male southern resident killer whale, L98, was killed in a collision with a tug boat; given the gender imbalances in the southern resident killer whale population, we assume that the death of this adult male would have reduced the demographic health of this population (see further discussion below).

Second, the number and proximity of vessels, particularly whale-watch vessels in the areas occupied by southern resident killer whales, represents a source of chronic disturbance for this population. Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Goodwin and Green 2004; Lusseau 2006). However, several authors suggest that the noise generated during motion is probably an important factor (Blane and Jackson 1994; Evans et al. 1992, 1994). These studies suggest that the behavioral responses of marine mammals to surface vessels are similar to their behavioral responses to predators.

Several investigators have studied the effects of whale watch vessels on marine mammals (Watkins 1986; Cockeron 1995; Au and Green 2000; Erbe 2002; Félix 2001; Magalhães et al. 2002; Williams et al. 2002; Richter et al. 2003; Scheidat et al. 2004; Amaral and Carlson 2005; Simmonds 2005;). The whale's behavioral responses to whale watching vessels depended on the distance of the vessel from the whale, vessel speed, vessel direction, vessel noise, and the number of vessels. The whales' responses changed with these different variables and, in some circumstances, the whales did not respond to the vessels. In other circumstances, whales changed their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions.

In addition to the disturbance associated with the presence of vessels, the vessel traffic affects the acoustic ecology of southern resident killer whales, which would affect their social ecology. Foote et al. (2004) compared recordings of southern resident killer whales that were made in the presence or absence of boat noise in Puget Sound during three time periods between 1977 and 2003. They concluded that the duration of primary calls in the presence of boats increased by about 15% during the last of the three time periods (2001 to 2003). At the same time, Holt et al. (2007) reported that southern resident killer whales in Haro Strait off the San Juan Islands in Puget Sound, Washington, increased the amplitude of their social calls in the face of increased sounds levels of background noise. Although the costs of these vocal adjustments remains unknown, Foote *et al*. (2004) suggested that the amount of boat noise may have reached a threshold above which the killer whales needs to increase the duration of their vocalization to avoid masking by the boat noise.

Critical Habitat

NMFS designated critical habitat for the DPS of Southern Resident killer whales on November 29, 2006 (71 FR 69054). Three specific areas were designated; (1) the Summer Core Area in Haro Strait and waters around the San Juan Islands; (2) Puget Sound; and (3) the Strait of Juan de Fuca, which comprise approximately 6,630 square kilometers of marine habitat. Three primary constituent elements exist in these areas: water quality to support growth and development, prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth, and passage conditions to allow for migration, resting, and foraging. Water quality has declined in recent years due to agricultural run-off, urban development resulting in additional treated water discharge, industrial development, and oil spills. The primary prey of southern residents, salmon, has also declined due to overfishing and reproductive impairment associated with loss of spawning habitat. The constant presence of whale-watching vessels and growing anthropogenic noise background has raised concerns about the health of areas of growth and reproduction as well.

Environmental Baseline

By regulation, the environmental baseline for biological opinions include the past and present impacts of all state, Federal or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this biological opinion also includes a general description of the natural factors influencing the current status of the listed species, their habitats, and the environment within the action area. The baseline analysis "is not the proportional share of responsibility the federal agency bears for the decline in the species, but what jeopardy might result from the agency's proposed actions in the present and future human and natural contexts." Pacific Coast Federation, 426 F.3d at 1093.

Our summary of the environmental baseline complements the information provided in the status of the species section of this Opinion, provides information on the past and present ecological conditions of the action area that is necessary to understand the species' current risk of extinction, and provides the background necessary to understand information presented in the *Effects of the Action* and *Cumulative Effects* sections of this biological opinion. The "impact" of the activities we normally identify in the *Environmental Baseline* of our Opinons allows us to assess the prior experience and state (or condition) of the endangered and threatened individuals and areas of designated critical habitat that occur in an action area. This is important because, as noted in the *Approach to the Assessment* section of this Opinion, in some phenotypic states, listed individuals will commonly exhibit responses they would not exhibit in other phenotypic states. The same is true for populations of endangered and threatened species: the consequences of change in the performance of individual on a population depend on the prior state of the population. Designated critical habitat is not different: under some ecological conditions, the physical and biotic features of critical habitat will exhibit response that they would not exhibit in other conditions. When we "add" the effects of a new, continuing, or proposed action to the prior condition of endangered and threatened individuals and designated critical habitat, as our regulations require, our assessments are more likely to detect a proposed action's "true" consequences on endangered species, threatened species, and designated critical habitat.

Because this is a programmatic consultation on what is essentially a continuing action with a geographic scope that encompasses all waters of the United States and its territories, this environmental baseline serves a slightly different purpose. First, as both a programmatic and a national consultation this Opinion does not assess the consequences of the EPA's recommended aquatic life criteria for specific sites or the listed resources that occur those specific sites. Rather, the *Environmental Baseline* for this Opinion focuses on the status and trend of the aquatic ecosystems in the United States and the consequences of that status for listed resources. Since our action area and the environmental baseline encompass a very broad spatial scale with many distinct ecosystems, wherever possible we have focused on common indicators of the biological, chemical, and physical health of the nation's aquatic environments. The *Environmental Baseline* for this consultation provides the background information and context that is necessary for our assessment of the *Effects of the Action.*

We divided the environmental baseline for this consultation into five broad geographic regions of the United State: the Atlantic Northeast Region, the Atlantic Southeast Region, the Gulf Coast Region, the Southwest Region, and the Pacific Northwest Region. In some instances regions were further subdivided according to ecoregions, importance to NMFS' trust resources or other natural features. In each section we describe the biological and ecological characteristics of the region such as the climate, geology, and predominant vegetation to provide landscape context and highlight some of the dominant processes that influence the biological and ecological diversity of the region where threatened and endangered species reside. We then described the predominant land and water uses within a region to illustrate how the physical and chemical health of regional waters and the impact of human activities have contributed to current status of listed resources.

Atlantic Northeast Region

This region encompasses Maine, New Hampshire, Massachusetts, Rhode Island, Vermont, Connecticut, New York, New Jersey, Delaware, Pennsylvania, Maryland, and Virginia. Major rivers in this region are the Penobscot, Connecticut, Hudson, Delaware, and Susquehanna rivers. Important estuarine areas include the Chesapeake Bay, Long Island Sound, Cape Cod Bay, and Massachusetts Bay.

The region is ecologically diverse, encompassing several broad ecoregions. According to Bailey's (1995) *Description of the Ecoregions of the United States*, this region encompasses the warm continental, the hot continental and the hot continental mountains divisions, and northern portions of the subtropical division – these ecoregions can be further subdivided into provinces based on vegetation. Climate is defined by hot humid summers and cold winters. Mean annual precipitation varies from about 35 to 45 inches per year. Vegetation in this region is characterized by tall broadleaf trees that provide a continuous dense canopy in summer, but shed their leaves completely in winter. Lower layers of small trees and shrubs are weakly developed. In spring, a luxuriant ground cover of herbs quickly develops, but is greatly reduced after trees reach full foliage and shade the ground. Needleleaf trees grow in colder, northern parts of the region and in mountain areas. Soils are generally rich in humus and strongly to moderately leached, although in the southern portions of this region, soils tend to be sandier and support second-growth forests of longleaf, loblolly, and slash pines (Bailey 1995).

Gulf of Maine

Natural History

This region encompasses drainages entering the Gulf of Maine, and is one of the most productive marine ecosystems in the world. Several significant rivers that drain into the gulf include the Merrimac, Kennebec, Androscoggin, Penobscot, and St. John Rivers (Table 24), and the significant estuaries that compose the larger Gulf of Maine include the Bay of Fundy, Massachusetts Bay, Merrymeeting Bay, and Cape Cod Bay. The Gulf of Maine is semi-enclosed, bounded to the south by Georges Banks and to the north by Brown's Bank. The area is strongly influenced by the Labrador Current, which makes the waters significantly colder and more nutrient rich than waters to the south, which are more strongly influenced by the Gulf Stream. The Gulf of Maine is characterized by salt marshes, kelp and seagrass beds, tidal mudflats, and underwater rocky outcrops, which form the foundation of a complex ecosystem and provide habitat for Atlantic herring, American lobster, Atlantic salmon, and several whale species. Merrymeeting Bay is the largest freshwater tidal estuary that enters the Gulf of Maine and has the largest freshwater outflow to the gulf (Kistner and Pettigrew 2001; Jackson et al. 2005). The

Kennebec and Androscoggin Rivers, along with four smaller tributaries, converge to form Merrymeeting Bay with the two larger rivers accounting for 98% of the inflow.

Table 27. Select rivers of the northeast United States that drain to the Gulf of Maine

Watershed	Approx. Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	Number of Fish Species	Number of Endangered Species
Penobscot	275	8,592	NE	42	14,196	45	1 fish
Kennebec	230	5,383	NE	43	9,076	48	1 fish
Androscoggin	164	3,263	NE	44	6,180	33	1 fish
Merrimack	180	5,014	NE	36	8,299	50	1 fish

Data from Jackson et al 2005; Maine Rivers 2007a, b
*Physiographic Provinces: NE = New England, AD = Adirondack Mountains, VR = Valley Ridge, AP = Appalachian Plateau, PP = Piedmont Plateau, CP = Coastal Plain, BR = Blue Ridge.

Human Activities and Their Impacts

Land Use. Most of the watersheds within this region are heavily forested with relatively small areas of highly urbanized lands (Table 25). While there is not much urban development in the Penobscot watershed except in and around Bangor, Doggett and Sowles (1989) report that tanneries, metal finishing, pulp and paper mills, textile plants, chemical products, and municipal sewage contribute chromium, mercury, zinc, copper, lead, arsenic, hydrocarbons, dioxins, PAHs, pesticides, and other contaminants to the river. The only major town in the Kennebec River watershed is Augusta, Maine (Jackson et al. 2005). The heaviest population density occurs in the watershed of the Merrimack River, which flows through industrial centers Manchester and Concord, New Hampshire, and Lowell and Lawrence, Massachusetts.

Textile mills, as well as paper and pulp mills, have long influenced water quality in the Penobscot, Kennebec, and Androscoggin rivers. The Kennebec River exceeds recommended levels of dioxins, arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, zinc, and PAHs in the sediments and surface water (MDEP 1999, Harding Lawson Associates 1999, Harding Lawson Associates 2000). Since 1990, the levels of dioxins in other Maine rivers have been decreasing, but the levels in the Kennebec have remained constant (Kahl 2001). At one time, the Androscoggin River was considered one of the ten most polluted rivers in the country. The river has become much cleaner since the CWA was passed, but pesticides, mercury, lead, sedimentation, total suspended solids, PCBs, and dioxins are still considered too high (Chamberland et al. 2002).

The Merrimack River watershed is one of the most heavily urbanized watersheds in the region, and some of the biggest sources of pollution facing the river are from industrial and urban sources, such as combined sewage overflows, industrial discharge, and stormwater run-off (USACE 2003). The upper mainstem of the Merrimack River has problems with bacteria, *E. coli*, and acidity, while the lower mainstem has problems with bacteria, metals, nutrients, dioxins, turbidity and suspended solids, and un-ionized ammonia. In all, over 125 miles of mostly lower watershed areas do not support their designated uses (USACE 2003).

Toxins draining from river systems have produced significant toxin levels in regional estuarine systems, particularly from New Hampshire south throughout the Cape Cod region. Casco Bay still harbors residual sediment contamination and organic carbon levels from industries of a century ago, including heavy metals, PCBs, pesticides, TBT, dioxins and furans, and PAHs (EPA 2006). Low dissolved oxygen and red tide from nutrient loading also remain issues in the area. Habitats here remain relatively coalesced, although fragmentation is on the rise, and eelgrass beds have undergone local reductions.

Toxic sediments have been identified in Merrymeeting Bay, although some pollutants like metals declined in the bay between 1980 and 1991, although copper levels have increased (Hayden 1998). Sediments associated with the Androscoggin River exhibit higher levels of PAHs and mercury, while sediments from the Kennebec River had higher levels of chromium, arsenic, and selenium (Hayden 1998). Merrymeeting Bay has more moderate levels of these toxins than the rivers themselves. Chilcote and Waterfield (1995) found that levels of arsenic are higher than levels identified by EPA as likely to have adverse effects. At one station, PAHs from the Androscoggin also exceeded EPA-identified levels of minimal effects. Commercially important fish also have elevated metal concentrations in their livers, which is thought to be from their time spent in Merrymeeting Bay (Kistner and Pettigrew 2001).

Human activities have impact the coasts of New Hampshire and Massachusetts. New Hampshire estuaries suffer from habitat fragmentation and degradation, bacterial and nutrient contamination, salt marsh degradation, and declines in the commercially valuable oyster and clam populations resulting from sewage and industrial pollution (EPA 2006). Several areas experience elevated nitrogen and phosphorus in water, high total organic carbon, and sediment contaminant levels in the benthos, as well as above average contaminants (PAHs, DDT, and PCBs) in fish and shellfish. A massive decline in eelgrass habitats occurred in 1989 and meadows have been relatively constant since.

Estuarine and bay systems of Massachusetts experience pressures from the major metropolitan region around Boston Harbor. The increased sewage and stormwater outflow results in a loss of roughly 1,000 acres of wetland habitat per year and cause closings in shellfish harvests due to bacterial contamination. Local wetland restoration projects have improved over 450 acres of wetland in the region. Over 26 invasive species have been identified in Massachusetts Bay, including the Asian shore crab and Pacific tunicate, and have contributed to a reduction in the industrial scallop fishery.

Table 28. Land uses and population density of several watersheds that drain to the Gulf of Maine

Watershed	Land Use Categories (%)				Density
	Agriculture	Forested	Urban	Other	(people/mi^2)
Penobscot	5	95			21
Kennebec	6	82	2	10	39
Androscoggin	5	86	2	7	65
Merrimack	6	75	13	6	404

Data from Jackson et al. 2005

Hydromodification Projects. There are five major hydroelectric dams along the mainstem of the Penobscot River as well as 111 other licensed dams located along the river and its tributaries.

Atlantic salmon historically migrated as far as 143 miles upstream of the mouth, but due to development along the river in the 1960s, Atlantic salmon were extirpated (Jackson et al. 2005). The population has since been re-established and runs of 2,000 to 4,000 occur with natural spawning as far upstream as 62 miles. Because 6,000 to 10,000 salmon are required for a sustainable population, the Penobscot run depends on fish from a local hatchery (Moore and Platt 1996).

The Kennebec River mainstem has eight large hydroelectric dams, which restrict fish passage both up and downstream. In 1999, the Edwards Dam was removed, opening 17 additional miles of habitat for fish and macroinvertebrates in the river. Removal of Edwards Dam restored full access to historical spawning habitat for species like Atlantic sturgeon, shortnose sturgeon, and rainbow smelt, but not for species like alewife, American shad, and Atlantic salmon that migrated much further up the river. Since the removal of Edwards Dam, dissolved oxygen levels and macroinvertebrate density have improved. Additionally, in 2007, the fish passage facilities on the lowest dam on the Kennebec River, as well as those on the Sebasticook River's second and third lowest dams, became operational.

The Androscoggin River has 14 hydroelectric dams on the mainstem of the river and 18 in the watershed. Fish ladders have been installed on the lower dams allowing anadromous fish passage to Lewiston Falls (Brown et al. 2006). The dams play a considerable role in the poor water quality of the river, causing reduced dissolved oxygen throughout the summer. During the 1960s, most of the river had oxygen levels of 0 ppm, resulting in massive fish kills. There is still a 14-mile stretch of river that requires aerators to provide dissolved oxygen to the river.

The Merrimack River watershed has over 500 dams, including three in Massachusetts and three in New Hampshire, that essentially make the mainstem into a series of ponds (Dunn 2002; Jackson et al. 2005). Flow alteration is considered a problem on the upper mainstem of the river and has resulted in the river not meeting EPA's flow requirements (USACE 2003).

Mining. Mining in watersheds of the Atlantic Northeast Region began before the Civil War. Since then, mining has been conducted for granite, peat, roofing slate, iron ore, sulfur, magnetite, manganese, copper, zinc, mica, and other materials. Currently, exploration for precious metals and basic metals is ongoing, but to a lesser extant than during the 1980s. Recent mining activities were conducted in this region by The Penobscot Nation, Champion Paper Company, Oquossoc Minerals, Boliden Resources, Inc., Black Hawk Mining, and BHP-Utah. There are several abandoned mines in the northeast watersheds that have become Superfund sites due to excessive pollutants being leached into groundwater, such as Elizabeth, Pike Hill, and Calhoun Mines, and others. Common pollutants leaked by mining operations in this area are lead, mercury, arsenic, and selenium (Ayuso et al. 2006; Piatak et al. 2006). Many of the abandoned mines are scheduled for cleanup; however, the impacts of their former use could persist for years after decommissioning.

*Commercial and Recreational Fishing****.*** The primary commercial fisheries along the Northeast coast by harvest weight exist for herring (39%), lobster (26%), blue mussel (6%), hatchery-origin sea-run Atlantic salmon (4%), groundfish (4%), quahog (4%), soft clam (3%), sea cucumber (3%), seaweed (3%), crabs (2%), and various other species (6%). Directed harvest of shortnose sturgeon and wild Atlantic salmon is prohibited by the ESA; however, both are taken incidentally in other fisheries along the east coast and are probably targeted by poachers throughout their range (Dadswell 1979; Dovel et al. 1992; Collins et al. 1996).

Long Island and the Connecticut River

Natural History

South of the Gulf of Maine is the Long Island Sound watershed, which includes portions of Connecticut, New York, Massachusetts, New Hampshire, Rhode Island, and Vermont. Long Island Sound was designated a national estuary in 1987, due to its significance as an area where fresh water from the Connecticut, Thames, and Housatonic rivers (90% of the freshwater input) mixes with the Atlantic Ocean. The sound ranges in salinity from 23 ppt in the western end to 35 ppt on the eastern side. The surface area of Long Island Sound is 1,320 square miles, draining an area of over 16,000 square miles. Long Island Sound connects to the Atlantic Ocean on both the eastern and western side, called “The Race” and the East River, respectively. The sound substrate is primarily mud, sand, silt, and clay, with very small areas of exposed bedrock. The sound is home to more 120 species of fish and at least 50 species use Long Island Sound as spawning grounds.

The Connecticut River drains a watershed of 11,259 square miles and flows approximately 410 miles to Long Island Sound. The river flows from the highlands of New Hampshire and Quebec, and is bordered by the Green and White Mountains. The Connecticut River bed is composed of glacial deposits and granitic bedrock. The average annual precipitation is approximately 43 inches. At the mouth, the average discharge is 10.2 billion gallons per day, or 15,715 cubic feet per second, which accounts for approximately 70% of the freshwater inflow to Long Island Sound (Jackson et al. 2005). The final 56 miles of the river prior to Long Island Sound is a tidal estuary (Jackson et al. 2005). The river and estuary are also important for many fish species, with 64 fresh water and 44 estuarine species having been recorded in the river or estuary, but 20 of the fish are nonnative (Jackson et al. 2005).

Human Activities and Their Impacts

Land Use. More than eight million people live in the Long Island Sound watershed. With so many people in the watershed, both point and non-point source pollution is a major concern. Toxic substances often adsorb to the surface of sediments, which means sediments with high surface to volume ratios like sand, silt, and clay, can hold more pollutants than larger substrates. The sound has elevated levels of PCBs, PAHs, nitrogen, lead, mercury, cadmium, cesium, zinc, copper, and arsenic. Organic and metal contaminants in Long Island Sound are above national averages (Turgeon and O’Connor 1991). Much of the lead, copper, and zinc are likely deposited via the atmosphere (Cochran et al. 1998). Cadmium, chlordane, and lead appear to be decreasing while copper is increasing (Turgeon and O’Connor 1991). Studies on winter flounder showed PAHs and PCBs leading to alteration of DNA in the livers of those fish (Gronlund et al. 1991). One of the biggest problems facing the sound is dissolved oxygen depletion (Parker and O’Reilly 1991), resulting in dead zones. The governors of Connecticut and New York have signed agreements to reduce the total nitrogen input to Long Island Sound by 58.5% before 2015 in an effort to get dissolved oxygen levels above 5 ppm for surface water, above 3.5 ppm for deeper water, and at or above 2 ppm for all water.

Within the Connecticut River watershed the dominant land use is forest (80%), with 11% used for agriculture and the remaining 9% in mixed uses (Jackson et al. 2005). Major towns in the Connecticut River watershed are Holyoke and Springfield, Massachusetts and Hartford, Connecticut. The human population in the watershed is approximately 179 people per square mile (Jackson et al. 2005). Throughout the 20th century, power plants, defense contractors, municipalities, and corporations such as General Electric, Union Carbide, and Pfizer contributed large quantities of pollutants to the river. Still to this day, approximately one billion gallons of raw sewage enters the river as a result of combined sewer overflow from Hartford, Connecticut alone (CRWC 2006). The river has become much cleaner since the CWA was passed, but chromium, copper, nickel, lead, mercury, and zinc, chlordane, DDT, DDE, PCBs, and PAHs are found in quantities above the EPA-recommended levels in sediments and fish tissue throughout the watershed (Jackson et al. 2005). Acid rain also affects rivers in the northeast, as it reduces the pH of rivers and causes metals to leach from bedrock at a faster rate (USFWS 2007).

Estuaries within Long Island Sound have historically been plagued by low dissolved oxygen, pathogens, habitat degradation and species decline, and sediment contamination (EPA 2006). These issues remain relevant today, with increasing human populations increasing contaminant loads and decreasing wetland habitat. Almost all measures of quality have been affected, including phosphorus load, low dissolved oxygen, and chlorophyll *a* concentrations, high sediment contaminants (DDT and metals) and total organic carbon, as well as excessive levels of PCBs in nearly all fishes sampled. Riverine and wetland restoration has been ongoing for several years and provided an additional 2,000 acres of wetland and over 50 miles of stream passage for migratory fishes. This may help curtail the decline of estuarine bird populations and oysters in recent years. Oyster harvest closures resulting from pathogen concentrations have been common for two decades and additional regulation of vessel discharges, illegal sewage connections to Long Island Sound, high volume of storm water effluent, and malfunctioning septic systems are identified as point sources for this.

Hydromodification Projects. The Connecticut River has 16 hydroelectric dams on the mainstem of the river and as many as an estimated 900 have been built in the watershed. Fish ladders have been installed at Vernon, Turner Falls, and Holyoke Dams allowing fish passage to areas above Holyoke Dam in Massachusetts since 1981 (USGS 2004). For some species, the ladders are not efficient, so fish passage continues to be compromised. For instance, overall passage efficiency at Turner Falls fish ladder is 17%, and has historically been inefficient at passing shad. Shortnose sturgeon are not able to migrate to spawning habitat above Holyoke Dam, which was recently re-licensed through 2039, so the only spawning shortnose sturgeon in the river are the fish that reside above the dam. The dams also affect the river's water quality, causing reduced dissolved oxygen and elevated water temperatures throughout the summer.

Mining. Dating back thousands of years, there is evidence of native people mining and extracting natural resources from the headwaters of the Connecticut River. Towns such as Plymouth, Vermont were famous for mining gold, iron, talc, soapstone, marble, asbestos, and granite (Ewald 2003). Other towns throughout New Hampshire and Vermont also mined gold, silver, soapstone, talc, granite, slate, and copper (Ewald 2003). There are many mines along the

Connecticut River, which currently degrade the river's water quality, including the country's first chartered copper mine. In many locations, far downstream of the mines, accumulated heavy metals are in concentrations high enough to threaten aquatic life. In other cases, the mines are abandoned or failing and need to be cleaned. Such is the case with Elizabeth Mine, an old copper mine perched above the Connecticut River that leaches heavy metals into the river. As a result, Elizabeth Mine has been declared a Superfund site. There is little to no mining in Long Island Sound although there has been and continues to be discussions about mining for sand and gravel.

Commercial and Recreational Fishing. Few commercial fisheries exist in the Connecticut River. Shad is the primary commercial fishery, although shellfish, bluefish, striped bass, and flounder can be caught in the tidal estuary near the mouth. There are many recreationally angled fish, such as shad, striped bass, bluefish, northern pike, largemouth and smallmouth bass, perch, catfish, and others.

Long Island Sound fisheries provide an estimated 5.5 million dollars to the Connecticut economy. The primary fisheries target oysters, lobsters, scallops, blue crabs, flounder, striped bass, and bluefish. Recently, due to dissolved oxygen deficiencies, the western portion of Long Island Sound has seen major declines in fish and shellfish populations. Despite these declines, the sound houses the largest oyster fishery in the US, providing 95% of the nation's oysters. At this same time, lobsters have been suffering from an unknown disease and their population has been declining. Simultaneously, menhaden have made a dramatic recovery over the past 10 years, which has resulted in much better fishing for larger predatory fish, such as striped bass.

Directed harvest of shortnose sturgeon is prohibited by the ESA. However, shortnose sturgeon are likely taken incidentally in fisheries in the Connecticut River and Long Island Sound. Moser and Ross (1993) found that captures of shortnose sturgeon in commercial shad nets disrupted spawning migrations in the Cape Fear River, North Carolina. Weber (1996) reported that these incidental captures caused abandonment of spawning migrations in the Ogeechee River, Georgia.

Hudson River

Natural History

The Hudson River flows approximately 315 miles to the ocean, with a watershed of 13,365 square miles. The river flows from the Adirondack Mountains, draining most of eastern New York State, to the Atlantic Ocean where the Hudson River Canyon continues onto the continental shelf, marking where the original mouth of the Hudson was covered by rising sea levels after the last ice age. The Hudson River bed is composed of metamorphosed plutonic rock in the Adirondack Mountains, then transitions to sedimentary rock, such as shale and limestone in the middle portion of the watershed. The lower portion of the watershed is a mixture of sedimentary, metamorphic, and igneous rocks. Average annual precipitation is approximately 36 inches per year. At the mouth, the average discharge is 13.5 billion gallons per day, or 20,906 cubic feet per second (Jackson et al. 2005). The Hudson River is a freshwater tidal estuary between Troy, New York at river mile 154 to Newburgh Bay at river mile 62, and then it is a tidal brackish estuary for the lower 62 miles to the Atlantic Ocean (Jackson et al. 2005). The river and estuary are home to over 200 fish species, with approximately 70 native freshwater fish species and 95 estuarine species having been recorded (Jackson et al. 2005).

Human Activities and Their Impacts

Land Use. The Hudson River watershed usage is 25% agriculture, 65% forested, 8% urban, and 5% other (Jackson et al. 2005). Major towns in the Hudson River watershed are New York City, Albany, Poughkeepsie, and Hudson, New York as well as Jersey City, New Jersey. The average population density in the watershed is approximately 350 people per square mile, but the actual density within a reach of the watershed varies widely. For instance, according to Jackson et al. (2005) population density at the headwaters at Lake Tear of the Clouds is 0 people per square mile, while the population density in Manhattan is over 25,907 people per square mile.

Throughout the 20th century, power plants, municipalities, pulp and paper mills, and corporations such as IBM, General Motors, and General Electric in particular, whom the EPA estimates dumped between 209,000 and 1.3 million pounds of PCBs into the river, contributed large quantities of pollutants to the Hudson River. The PCB levels in the Hudson River are among the highest nationwide. The upper basin is mostly unaffected by humans, with clear, soft water with low nutrients. The middle Hudson River is more polluted, with 30 to 50% of the land in this region being used for agriculture and several cities such as Corinth, Glens Falls, Hudson Falls, and Fort Edward contributing industrial waste to the river. The tidal freshwater portion of the river is nutrient rich with exceptionally low gradient. High tide in this stretch causes the river to flow backwards due to the low gradient and this prevents stratification. The brackish tidal estuary portion of the Hudson River is nutrient rich with hard water. Two hundred miles of the Hudson River, from Hudson Falls to New York City, were designated as a Superfund site due to the amount of pollution. There are still elevated amounts of cadmium, copper, nickel, chromium, lead, mercury, zinc, DDT, PCBs, and PAHs in quantities above the EPA recommended levels in sediments and fish tissue throughout the watershed (Wall et al. 1998).

Estuarine conditions surrounding the Hudson River are extremely poor (EPA 2006). Most issues stem from the extremely dense human population center around New York City. Fish consumption warnings are commonplace due to high mercury levels, over 200 million gallons of untreated sewage enter the bay daily, and only 20% of the former wetland area remains. Nitrogen and phosphorus are generally very high, sediments are highly contaminated (PCB), and total organic carbon is generally elevated. Only about 20,000 acres of wetland remain in the region. Although these poor conditions persist, wading birds formerly absent are present today and osprey (indicator bird species) are showing resurgence.

Hydromodification Projects. The mainstem Hudson River has 14 dams and dams exist near the mouths of many tributaries, but the lower 154 miles of tidally influenced river is undammed. Several flood control dams on tributaries such as the Indian and Sacandaga rivers have drastically altered the flow of the mainstem Hudson River. The Hudson is an important river for anadromous fishes because it has a low number of physical obstructions, and contains one of the largest populations of endangered shortnose sturgeon in the United States. Prior to the Clean Water Act, the middle stretch of the Hudson River and much of the lower reaches had low dissolved oxygen as a result of reduced flow behind the dams, high nutrients, and the collection of waste with high biological oxygen demand (Jackson et al. 2005).

Mining. The Hudson River has been periodically important as a source of metals and mined resources. The Adirondack Mountains, in the headwaters, have been mined for silver, iron, titanium, coal, talc, vanadium, graphite, garnet, and zinc at various times over the past 300 years. McIntyre Mine is an example of a mine that has produced different minerals during different generations. Initially bought as an iron mine, McIntyre sat dormant for 75 years before titanium was discovered there and mined until 1982, when it was abandoned.

Commercial and Recreational Fishing. The Hudson River commercial fishery historically caught fish, blue crabs, and oysters. Now, the only fish that is caught commercially in the Hudson is American shad. Historically, Atlantic sturgeon, striped bass, American eel, and white perch were productive commercial fisheries. The striped bass fishery closed in 1976 due to PCBs in the river and fish tissue. Atlantic sturgeon were fished until the mid 1990s. Blue crabs are still fished in the estuary all the way to Troy, New York with recent catches over 88,185 pounds per year. There is no commercial fishery for oysters but they used to be taken commercially in the brackish tidal section of the Hudson River.

Delaware River

Natural History

The Delaware River flows approximately 329 miles to the ocean, with a watershed of 12,757 square miles. The river originates in the Catskill Mountains with over half of the river flowing through Pennsylvania and the rest of the watershed occupying parts of New Jersey, New York, and Delaware. The Delaware River’s geology is sandstone with shale conglomerate in the upper watershed transitioning to sandstone, shale, and limestone in the middle watershed and igneous and metamorphic rock in the lower watershed. The average annual precipitation is approximately 43 inches. At the mouth, the average discharge is 9.6 billion gallons per day, or 14,903 cubic feet per second. Although it is only the 42nd largest river in the United States by discharge, Philadelphia is home to the largest freshwater port in the country (Jackson et al. 2005). The Delaware River estuary begins in Trenton, New Jersey and extends downstream for 144 miles (Jackson et al. 2005). The river and estuary are home to 105 species of fish, with approximately eight nonnative fish (Jackson et al. 2005).

Human Activities and Their Impacts

Land Use. The Delaware River watershed usage is 24% agriculture, 60% forested, 9% urban, and 7% surface water or other (Jackson et al. 2005). Major towns in the Delaware River watershed are Easton, Allentown, Reading, and Philadelphia, Pennsylvania; Trenton and Camden, New Jersey; and Wilmington, Delaware. The human population in the watershed is approximately 555 people per square mile with most distributed around the estuary (Fischer et al. 2004; Jackson et al. 2005). As the area’s population grew, water quality significantly degraded around Philadelphia with serious water quality problems observed as early as 1799. By the 1960s, the average dissolved oxygen in the lower river was approximately 0.2 ppm. A survey in the 1970s of organochlorine frequency in rivers ranked the Delaware at Trenton and the Schuylkill, the largest tributary to the Delaware River, as the 8th and 1st worst, respectively in the nation (Jackson et al. 2005). Urban and agricultural activities continue to affect the basin water quality today. Organochlorines, pesticides, nutrients, organics, and trace elements were widely detected in small tributaries and mainstream reaches (Fischer et al. 2004). In the Delaware River Basin, commonly detected organochlorines include DDTs, PCBs, chlordanes, and dieldrin. Fisher et al. (2004) found that 84% of the fish tissue sampled contained PCBs, while 21% of the sediment samples contained PCBs despite that the manufacture and use of these compounds ceased in the 1970s or earlier. These compounds are bioaccumulating in the food chain, and occasionally exceed wildlife protection guidelines (52% of the sites exceeded wildlife guidelines for PCBs, whereas 16% of the sites exceeded guidelines for dieldrin, and 12% exceeded wildlife guidelines for DDT [Fischer et al. 2004]).

Trace metal contamination is also a significant concern within the basin, and may be a particular concern for benthos including endangered shortnose sturgeon. Trace metals detected at high levels included arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc (Fischer et al. 2004). Most trace metal contamination was associated with former or on-going industrial sites such as mines and metal smelters.

The Delaware Estuary is considered to be in poor condition primarily from historical and modern toxin contributions from population centers such as Philadelphia along the Delaware River (EPA 2006). Overfishing and habitat loss also play a role in and along the estuary system itself. Estuarine waters contain high nitrogen and phosphorus levels with low chlorophyll *a* concentrations. Water clarity is variable, but fish tissues contain unsatisfactory levels of PCBs, DDT, and PAHs. Invasive plant species, including the common reed and purple loosestrife, have displaced native marsh species. Disease and low recruitment in oyster populations have had significant effects in commercial and ecological parameters of Delaware Bay. Shad populations declined due to poor water quality and have not recovered, which may indicate environmental stress in several other finfish populations.

Hydromodification Projects. The Delaware River has 16 dams in its headwaters, but the middle and lower rivers are the longest undammed stretch of river east of the Mississippi. This stretch of free-flowing river is beneficial to anadromous and catadromous species, such as American shad, striped bass, and American eels.

Mining. The Delaware River watershed, particularly the eastern section, was home to the majority of the nation’s anthracite coal. As a result, many mining towns were established in the watershed to exploit the abundant resources. By 1914, over 181,000 people were employed as miners in the region. Apart from the coal mining, other minerals such as sulfur, talc, mica, aluminum, titanium, and magnesium were mined. Mines were also established for sand and gravel.

Commercial and Recreational Fishing. In the Delaware River, commercial fisheries exist for American shad, weakfish, striped bass, Atlantic croaker, Atlantic silversides, bay anchovy, black drum, hogchoker, northern kingfish, and American eel. Commercial fishermen use gillnets and trawls as the primary means of capturing fish. Bycatch is a concern for the recovery of endangered shortnose sturgeon, where the highest mortality rates are recorded in gillnet fisheries. Recreational fishermen target weakfish, striped bass, croaker, drum, kingfish, and eel.

Chesapeake Bay Drainages

Natural History

Chesapeake Bay, the largest estuary in the United States, was formed by glacial activity more than 18,000 years ago. The bay stretches some 200 miles from Havre de Grace, Maryland to Norfolk, Virginia, with more than 11,000 miles of shoreline. At its widest point, Chesapeake Bay is about 35 miles wide (near the Potomac River). Despite its massive size, the bay is relatively shallow – average depth is only 21 feet – making it susceptible to significant fluctuations in temperature.

Chesapeake Bay lies totally within the Atlantic Coastal Plain but the watershed includes parts of the Piedmont Province and the Appalachian Province. Tributaries provide a mixture of waters with a broad geochemical range to the bay with its own mixture of minerals, nutrients, and sediments depending on the geology of the place where the waters originate. In turn, the nature of the bay itself depends on the characteristics and relative volumes of these contributing waters. More than 100 rivers and streams deliver fresh water to Chesapeake Bay, and major rivers include the Susquehanna and Potomac (see Table 26).

The Susquehanna River, rated as the 18th largest river in the United States based on average discharge at the mouth, flows approximately 448 miles to Chesapeake Bay (Kammerer 1990; Jackson et al. 2005). The river flows north to south from New York, through Pennsylvania, and reaches the Chesapeake Bay at Havre de Grace, Maryland. The Susquehanna River's bed is rocky throughout, being described as a mile wide and a foot deep with distinct pool/riffle formations even near the mouth. The Susquehanna drains into the most northern portion of Chesapeake Bay and is not tidally influenced. The Susquehanna River provides about 50% of the freshwater inflow into the Chesapeake Bay.

Table 29. Select rivers of the northeast United States that drain to Chesapeake Bay

Watershed	Approx. Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species	No. Endangered Species
Susquehanna	448	27,580	AP, VR, PP, BR	39	40,718	103	2 birds
Potomac	383	14,700	AP, VR, PP BR, CP	39	11,301	95	1 fish, 1 mussel
James	340	10,102	VR, BR, PP, CP	43	8,016	109	3 fish, 1 amphibian, 1 reptile, 6 mussels

Data from Jackson et al. 2005; Smock et al. 2005
*Physiographic Provinces: NE = New England, AD = Adirondack Mountains, VR = Valley Ridge, AP = Appalachian Plateau, PP = Piedmont Plateau, CP = Coastal Plain, BR = Blue Ridge.

The Potomac and James Rivers, on the other hand, are located south of the Susquehanna River basin and are tidally influenced. The Potomac River estuary begins two miles below the Washington, D.C.-Maryland border, just below the Little Falls of the Potomac River. The river's headwaters begin in the Allegheny Mountains of West Virginia and the Potomac and flows through Washington, D.C., to the western side of the Chesapeake Bay. The substrate of the

Potomac and its tributaries is mostly schist, phyllite, and metavolcanic rock. Ninety-five fish species live in the Potomac, but only 65 of those are native to the area (Jackson et al. 2005). The James River is one of the longest bodies of water in entirely one state, beginning in the Allegheny Mountains of western Virginia and flowing across the state to the Chesapeake Bay.

Human Activities and Their Impacts

Land Use. The Chesapeake Bay area, while dominated by forested lands, is more heavily influenced by agriculture than many other areas in the northeast and middle Atlantic (see Table 27 for land uses by watershed). Urbanized areas are scattered but dominate the landscape in certain areas (e.g., Washington D.C. metro area, in the Potomac River watershed, and Scranton and Harrisburg, Pennsylvania in the Susquehanna River watershed). Most of the bay's waters are degraded, and algal blooms are a chronic problem. Nutrient pollution and heavy sediment loads have lead to large anoxic areas within the bay, and fish kills in some areas. Agricultural practices, urban development, as well as natural sources of sediment influence water quality within the bay. Past logging practices in the basins draining to Chesapeake Bay also influenced sediment loads within several rivers. In the Susquehanna River watershed, sediment transport in the early 1900s was nine times higher than it was 200 years earlier, due to logging and agriculture.

Overall, in 2006, less than one-third of Bay water quality goals were met (Chesapeake Bay Program 2007). Direct discharges of sewage and industrial wastewater into the Susquehanna River watershed and contributes to degraded water quality in the basin. The number of outfalls totals over 400 in the watershed, generally with the number of outfalls being proportional to the size of the city (Binghamton, New York: 10, Harrisburg, Pennsylvania: 65, Scranton, Pennsylvania: 70). As a result, the Susquehanna River contributes 44% of the nitrogen and 21% of the phosphorous to the Chesapeake Bay. This has led to large algal blooms in the bay and a resulting "dead zone" between Annapolis, Maryland and Newport News, Virginia. In 2005, the Susquehanna River was named America's most endangered river by American Rivers, who produces an annual list. Even 35 years after the Clean Water Act, there are still elevated levels of copper, sulfur, selenium, arsenic, cobalt, chromium, lead, mercury, zinc, and pesticides (Beyer and Day 2004) as well as depressed pH associated with abandoned mines in the watershed (Hoffman 2008). Excessive nutrient and sediment loads also significantly impair the Susquehanna, stemming from urban and agricultural runoff and sewage treatment discharge (Hoffman 2008). Although water quality has significantly improved in the Potomac River over the past 50 years, the river remains threatened by elevated amounts of cadmium, chromium, copper, lead, dioxin, PCBs, and chlordane, which may have resulted in recent highly publicized reports of male fish producing eggs.

Similarly, the James River has elevated levels of zinc, copper, cadmium, nickel, chromium, lead, arsenic, dioxin, PCBs, and pesticides. The James River was also the site of one of the nation's most publicized pollution events when a manufacturing plant discharged, for nearly ten years, the chlorinated insecticide Kepone in the lower river (Smock et al. 2005). The insecticide is bioaccumulated, and resulted in a ban on commercial fishing in the lower river. Although the ban has been lifted, accumulations of Kepone in the riverine sediments are still cause for concern (Smock et al. 2005).

Table 30. Land uses and population density in several watersheds that drain to Chesapeake Bay

Watershed	Land Use Categories (%)				Density
	Agriculture	Forested	Urban	Other	(people/mi^2)
Susquehanna	27	63	9	---	145
Potomac	32	58	5	6	358
James	23	71	6	---	249

Data from Jackson et al. 2005; Smock et al. 2005

Hydromodification Projects. The Chesapeake Bay is home to several moderate to small sized dam projects. While only a few impoundments in the Potomac River and its tributaries are larger than 1.5 square miles, the Susquehanna River has over 100 dams along the mainstem and the first major dam is located just 10 miles upstream of the mouth. The Anacostia River, a major tributary to the Potomac River is scheduled to have some 60 dams removed or altered to improve water quality and fish passage. Dams in other basins have also been upgraded or are planned for upgrades. For instance, between 1928 and 1972, no shad passed Conowingo Dam on the Susquehanna River, 10 miles upstream of the mouth, but since passage facilities were installed fish abundance has increased from approximately 100 to more than 100,000 individuals. Similarly, many dams have been improved or removed in the James River. In 1999, a fish ladder added to Boscher Dam, which had prevented upstream fish runs since 1823 provided access to 137 additional miles of the James River and 168 miles of its tributaries.

Mining. In the Chesapeake Bay watershed, coal mining has likely had the most significant impact on water quality. Coal waste and acid mine drainage damaged much of the river and its tributaries. There was so much coal silt in the Susquehanna at one point that a fleet of over 200 vessels began harvesting the silt from the river's bed. From 1920 to 1950, over 3 million tons of coal was harvested from behind one dam. Later, between 1951 and 1973, over 10 million tons were harvested from behind another dam. Mining in this watershed was so extensive that while many mines have been reclaimed and others are currently being reclaimed, at the current level of funding, it will take decades to completely reclaim all of the old mines in the watershed. Abandoned coal mines leach sulfuric acid as a result of natural reactions with the chemicals found in coal mines. Much of the Appalachian Mountain chain that was mined for coal is now leaching sulfuric acid into tributaries of the Chesapeake Bay and requires some sort of treatment to improve the water quality of the region. Many of these abandoned coal mines must be treated with doses of limestone to balance the pH of the water draining from the mines. Coal is abundant through the watershed, amounting to nearly 30 billion tons of coal mined. Coal is no longer a primary industry in the watershed, but the impacts of the acid mine drainage are still prominent.

Commercial and Recreational Fishing. The Chesapeake Bay supports fisheries for American eel, croaker, blue crab, black sea bass, bluefish, oyster, red drum, spot, striped bass, summer flounder, weakfish, menhaden, and white perch (CFEPTAP 2004). Populations of striped bass got so low in the mid 1980s that a moratorium started in 1985, but they recovered so well that well-regulated harvests are now permitted. Since the mid 1990s, levels of blue crab and menhaden have dropped to the lowest levels in history. Species such as catfish and white perch are year round residents and managed by individual states around the bay. Species like Spanish mackerel, king mackerel, red drum, and summer flounder have ranges that extend beyond the bay and are managed under multiple regional management plans. Some species such as American shad are allowed to be fished by some states (Virginia and Maryland) within the Chesapeake Bay, but not by other states (Delaware and Pennsylvania).

Atlantic Southeast Region

This region covers all drainages that ultimately drain to the Atlantic Ocean (South Carolina and parts of Georgia, North Carolina, Florida, and Virginia). The region encompasses three ecoregions—the hot continental division, subtropical division, and savanna division (southern most tip of Florida's panhandle). The hot continental division is characterized by its winter deciduous forests dominated by tall broadleaf trees, soils rich in humus and moderately leached (Inceptisols, Ultisols, and Alfisols), and rainfall totals that decrease with distance from the Atlantic Ocean (Bailey 1995).

Most of the Atlantic Southeast Region is contained within the subtropical ecoregion and is characterized by a humid subtropical climate with particularly high humidity during summer months, and warm mild winters. Soils are strongly leached and rich in oxides of iron and aluminum (Bailey 1995). The subtropical ecoregion is forested, largely by second growth forests of longleaf, loblolly and slash pines, with inland areas dominated by deciduous trees. Rainfall is moderate to heavy with annual averages of about 40 inches in the north, decreasing slightly in the central portion of the region, and increasing to 64 inches in southern Florida. The savanna ecoregion has a tropical wet-dry climate, controlled by moist warm topical air masses and supports flora and fauna that is adapted to fluctuating water levels (Bailey 1995).

In the sections that follow we describe several basins and estuaries to characterize the general ecology and natural history of the area, and past and current human activities and their impacts on the area. The region contains more than 22 river systems that generally flow in a southeasterly direction to the Atlantic Ocean. The diverse geology and climate ensures variability in biological productivity and hydrology. Major basins include the Albemarle-Pamlico watershed and its tributaries, the Cape Fear River, Winyah Bay and the Santee-Cooper Systems, the Savannah, Ogeechee, and the St. Johns Rivers. The more northern river, the Roanoke, which is part of the Albemarle-Pamlico watershed, is cooler and has a higher gradient and a streambed largely characterized by cobble, gravel and bedrock.

The southern rivers are characterized by larger portions of low gradient reaches, and streambeds that are composed of greater amounts of sand and fine sediments—are often high in suspended solids, and have neutral to slightly acidic waters with high concentrations of dissolved organic carbon. Rivers emanating entirely within the Coastal Plain are acidic, low alkalinity, blackwater systems with dissolved organic carbon concentrations often up to 50 mg/L (Smock et al. 2005). We describe several river basins in detail to provide additional context for evaluating the influence of the environmental baseline on listed species under NMFS' jurisdiction and the health of their environment.

Albemarle-Pamlico Sound Complex

Natural History

The Albemarle-Pamlico Sound Estuarine Complex, the largest lagoonal estuarine system in the United States, includes seven sounds including Currituck Sound, Albemarle Sound, Pamlico Sound and others (EPA 2006). The Estuarine Complex is separated from the Atlantic Ocean by the Outer Banks, a long barrier peninsula, and is characterized by shallow waters and wind-driven tides that result in variable patterns of water circulation and salinity. Estuarine habitats include salt marshes, hardwood swamp forests, and bald cypress swamps.

The Albemarle-Pamlico watershed encompasses four physiographic regions—the Valley and Ridge, Blue Ridge, Piedmont and Costal Plain Provinces. Basin geology strongly influences the water quality and quantity within the basin. The headwaters of the basin tributaries are generally steep and surface water flowing downstream has less opportunity to pick up dissolved minerals. As the surface water flows reaches the Piedmont and Coastal Plain, water velocity slows due to the low gradient and streams generally pick up two to three times the mineral content of surface waters in the mountains (Spruill et al. 1998). At the same time, much of the upper watershed is composed of fractured rock overlain by unconsolidated and partially consolidated sands. As a result more than half of the water flowing in streams discharging to the Albemarle-Pamlico Estuarine Complex comes from ground water.

Primary freshwater inputs to the estuary complex include the Pasquotank, Chowan and Roanoke rivers that flow into Albemarle Sound, and the Tar-Pamlico and Neuse rivers that flow into Pamlico Sound. The Roanoke River is approximately 410 miles long and drains a watershed of 9,580 square miles. The Roanoke River begins in the mountains of western Virginia and flows across the North Carolina border before entering Albemarle Sound. The upper Roanoke River's geology is primarily a high gradient boulder-rubble bedrock system. The middle Roanoke River is primarily course sand and gravel. The lower section of the river is almost entirely organic-rich mud. The average annual precipitation is approximately 43 inches. At the mouth, the average discharge is 5.3 billion gallons per day, or 8,193 cubic feet per second (Smock et al. 2005). The Roanoke River is home to 119 fish species, and only seven of those are not native to the area (Smock et al. 2005). The Roanoke is also home to nine endangered fish species, two amphibians, and seven mussels, including several important anadromous fish species.

The Neuse River is 248 miles long and has a watershed of 6,235 square miles (Smock et al. 2005). The Neuse River watershed is also located entirely within the state of North Carolina, flowing through the same habitat as the Cape Fear River, but ultimately entering Pamlico Sound. The river originates in weathered crystalline rocks of the Piedmont and crosses sandstone, shale, and limestone before entering Pamlico Sound (Turekian et al. 1967). The average annual precipitation is approximately 48 inches. At the mouth, the average discharge is 3.4 billion gallons per day, or 5,297 cubic feet per second (USGS 2005).

Human Activities and Their Impacts

Land Use. Land use in the Roanoke River is dominated by forest (68%) and the basin contains some of the largest intact, least disturbed bottomland forest floodplains along the eastern coast.

Three percent of the basin qualifies as urban land uses and 25% is used for agriculture (Smock et al. 2005). The only major town in the Roanoke watershed is Roanoke, Virginia and population in the watershed is approximately 80 people per square mile (Smock et al. 2005). In contrast, the Neuse River watershed is described as 35% agriculture, 34% forested, 20% wetlands, 5% urban, and 6% other, with a basin-wide density of approximately 186 people per square mile (Smock et al. 2005). While the population has increased in the Albemarle-Pamlico Complex more than 70% during the last 40 years, the rate of growth is relatively low for many coastal counties in the Southeast (EPA 2006). Much of the estuarine complex is protected by large tracts of state and federally protected lands, which may reduce development pressures.

Coal is mined from the mountainous headwaters of the Roanoke River in southwestern Virginia. Mining through the Piedmont and coastal areas of North Carolina was conducted for limestone, lead, zinc, titanium, apatite, phosphate, crushed stone, sand, and fossils. Many active mines in these watersheds are still in operation today. These mines contribute to increased erosion, reduced pH, and leached heavy metals.

Agricultural activities are major source of nutrients to the estuary and a contributor to the harmful algal blooms (HABs) in summer, although according to (McMahon and Woodside 1997 as cited in EPA 2006) nearly one-third of the total nitrogen inputs and one-fourth of the total phosphorus input to the estuary are from atmospheric sources. Primary agricultural activities within the watershed include corn, soybean, cotton, peanut, tobacco, grain, potato, and the production of chicken, hog, turkey, and cattle.

The Neuse River entered the national spotlight during the early 1990s due to massive and frequent fish kills within the basin. Over one billion American shad have died in the Neuse River since 1991. The problem is persistent but the cause of the kills differs among events; in 2004 more than 700,000 estuarine fish died and more than 5,000 freshwater fish died within the basin. Freshwater species most commonly identified during investigations included sunfishes, shad, and carp, while estuarine species most commonly reported included menhaden, perch, and croaker. Atlantic menhaden have historically been involved in a majority of estuarine kill events and have exhibited stress and disease in conjunction with fish kills. Fish kill events may often have different causative agents, and in many cases the precise cause is not clear, but high levels of nutrients, HABs, toxic spills, outbreaks of a marine organism, *Pfiesteria pescicida*, low dissolved oxygen concentrations and sudden wind changes that mix hypoxic waters, are some of contributing factors or causes to the basins persistent fish kills (NCDWQ 2004).

Both the Roanoke River and the Neuse River are fragmented by dams. The reservoirs are used for flood control and recreation, but the amount of agricultural and urban runoff that collects behind the dams has caused sanitation problems in the recent past. Three dams were removed recently in an effort to improve environmental conditions and fish passage. Widespread stream modification and bank erosion were rated high within the greater watershed relative to other sites nationally (Spruill et al. 1998).

Conditions within the Albemarle-Pamlico estuary system are relative good compared to other northeastern estuaries. Agricultural and urban runoff provide the majority of toxins to the region that can impair water and habitat quality, as well as degrade fishery resource quality and quantity, including Atlantic sturgeon and numerous sport and commercially important fish species (EPA 2006). Chlorophyll *a* is the most significant detractor to water quality and total organic carbon has the greatest impact on sediment quality. Benthic quality and fish tissue contamination (PAHs and PCBs) also have suffered from human-introduced toxins. Losses of 25 to 50% of wetlands surrounding tributaries have lead to significant reduction in habitat as a result of human development.

Commercial and Recreational Fishing. The Albemarle and Pamlico Sounds and associated rivers support a dockside commercial fishery valued at over $54 million annually. The commercial harvest includes blue crabs, southern flounder, striped bass, striped mullet, white perch, croaker, and spot, among others. Roughly 100 species are fished commercially or recreationally in the region. The Neuse River supports many of the same species as the Roanoke River. Commercial and recreational fisheries exist for oyster, crab, clam, American shad, American eel, shrimp, and many other species. Shellfish can be collected by dredging, which has adverse effects to benthic organisms, including shortnose sturgeon that use estuarine areas for feeding. Commercial fisheries along the South Carolina coast use channel nets, fyke nets, gillnets, seines, and trawls. All of these methods can accidentally capture a shortnose sturgeon.

Major Southeast Coastal Plains Basins

Natural History

More than five major river basins flow through the Coastal Plains of the Southeast and directly enter the Atlantic Ocean, including the Cape Fear, Great Pee-Dee, Altamaha, and the St. Johns rivers (see Table 28 for a description of several basins within this region). Rainfall is abundant in the region and temperatures are generally warm throughout the year. Northern rivers originate in the Blue Ridge Mountains or the Piedmont Plateau, but all the rivers described in this section have sizeable reaches of slack water as they flow through the flat Coastal Plain. Two rivers, the Satilla in Georgia and the St. Johns in Florida, are located entirely within the Coastal Plain. The highest elevation of the St. Johns River is 26 feet above sea level, so the change in elevation is essentially one inch every mile, making it one of the most gradually flowing rivers in the country.

Smock et al. (2005) described the mountains and plateau as heavily dissected and highly metamorphosed rock of Paleozoic age, with occasional areas of igneous and sedimentary rock. Underlying rock is varied with bands of limestone, dolomite, shale, sandstone, cherts, and marble, with a number of springs and caves scattered throughout the area. At the fall line, where the Piedmont Plateau meets the sedimentary deposits of the Coastal Plain, steep changes in elevation result in rapids or falls before the rivers level off in their Coastal Plain reaches. Here, soils are acidic with a low cation exchange capacity and a sandy or loamy surface horizon, and a loamy or clay subsurface. The acidic characteristics, slow flowing water with poor flushing and high organic and mineral inputs gives these waters their characteristic blackwater (or brownwater, for those rivers that originate in the Piemont Plateau) appearance. The Satilla River is a blackwater river that has a naturally low pH (between four and six) and white sandbars. Due to the low pH, it also has naturally lower productivity than other rivers that originate within the mountains or the plateau.

Table 31. Rivers of the Southeast United States

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precip (in)	Mean Discharge (cfs)	No. Fish Species	Number of Endangered Species
Cape Fear	320	9,324	PP, CP	47	7,663	95	8 fish, 1 mammal, 15 mussels
Great Pee Dee	430	10,641	BR, PP, CP	44	13,102	>100	6 fish, 1 reptile
Santee-Cooper	440	15,251	BR, PP, CP	50	15,327	>100	5 fish, 2 reptiles
Savannah	300	10,585	BR, PP, CP	45	11,265	>100	7 fish, 4 amphibians, 2 reptiles, 8 mussels, 3 crayfish
Ogeechee	250	5,212	PP, CP	44	4,061	>80	6 fish, 2 amphibians, 2 reptiles, 1 mussel
Altamaha	140	14,517	PP, CP	51	13,879	93	1 mammal, 12 fish, 2 amphibians, 2 reptiles, 7 mussels, 1 crayfish
Satilla	200	3,530	CP	50	2,295	52	2 fish, 1 amphibian, 2 reptiles, 1 mussel
St. Johns	311	8,702	CP	52	7,840	>150	1 mammal, 4 fish, 2 reptiles, 2 birds

Data from NCDENR 1999; Smock et al. 2005
*Physiographic Provinces: BR = Blue Ridge, PP = Piedmont Plateau, CP = Coastal Plain

Human Activities and Their Impacts

Land Use. Across this region, land use is dominated by agriculture and industry, and to a lesser extent timber and paper production, although more than half of most basins remain forested. Basin population density is highly variable throughout the region with the greatest density in the St. Johns River watershed with about 200 people per square mile of catchment, most of whom are located near Jacksonville, Florida. In contrast, there are only 29 people per square mile in the Saltilla River watershed in Georgia (Smock et al. 2005). See Table 29 for a summary of land uses and population densities in several area basins across the region (data from Smock et al. 2005).

The largest population centers in the region include Miami and Jacksonville, Florida and Savannah, Georgia. Major towns include Greensboro, Chapel Hill, Fayetteville, and Wilmington, North Carolina in the Cape Fear River watershed; Winston-Salem, North Carolina and Georgetown, Florence, and Sumter, South Carolina in the Great Pee-Dee River Watershed; Charlotte, Hickory, and Gastonia, North Carolina and Greenville and Columbia, South Carolina in the Santee-Cooper River watershed; Savannah and Augusta, Georgia, in the Savannah River watershed; Louisville, Statesboro, and Savannah, Georgia, in the Ogeechee River watershed; Athens and Atlanta, Georgia, in the Altamaha River watershed; and Jacksonville, Florida in the St. Johns River watershed.

Several of the rivers in the region have elevated levels of metals including mercury, fecal coliform, ammonia, turbidity, and low dissolved oxygen. These impairments are caused by municipal sewage overflows, mining, non-point source pollution, waterfowl, urban runoff, marinas, agriculture, and industries including textile manufacturing, power plant operations, paper mills, and chemical plants (Mehta 2008; Harned and Meyer 1983; Berndt et al. 1998; NCDENR 1998; Smock et al. 2005).

Several watersheds exhibit high nitrogen loads including the Cape Fear River, Winyah Bay, Charleston Harbor, St. Helena Sound, Savannah River, Ossabaw Sound, Altamaha River, and St. Mary's River and Cumberland Sound (Bricker et al. 2007). Nitrate concentrations (as nitrogen) tend to be higher in stream draining basins with agricultural and mixed land uses (Berndt et al. 1998). Based on studies in Georgia, however, nitrate loads did not vary with growing season of crops (periods of heaviest fertilizer application), but were influenced by high stream flow, which could be related to downstream transport by subsurface flows (Berndt et al. 1998).

Table 32. Land uses and population density in several Atlantic southeast basins

Watershed	**Land Use Categories (%)**				**Density (people/mi^2)**
	Agriculture	**Forested**	**Urban**	**Other**	
Cape Fear River	24	56	9	11	80
The Great Pee-Dee	28	58	8	6	127
Santee-Cooper River	26	64	6	4	168
Savannah River	22	65	4	9	91
Ogeechee River	18	54	1	17 (wetlands)	78
Altamaha River	--	64	3	7	73
Satilla River	26	72	1	1	29
St. Johns River	25	45	6	24 (wetlands & water)	202

Data from Smock et al. 2005

Sediment is the most serious pollutant in the Yadkin (Pee-Dee) River and has historically been blamed on agricultural runoff. In the mid 1990s, farmers in the region began using soil conservation techniques that have reduced sediment inputs by 77%. The reduction in sediment inputs from farms did not translate to a reduction in sediment in the river, and during this period there was a 25% reduction in agricultural land and a 38% increase in urban development.

Where data are available, estuaries throughout the region contain generally moderate to severe nitrogen loads from river systems (Bricker et al. 2007). This has lead to toxic blooms of algae in some areas. Eutrophication has been noted particularly in the St. Johns River region. Low dissolved oxygen levels have also been found in the area around the Savannah River.

Mining. Mining occurs throughout the region. South Carolina is ranked 25th in terms of mineral value and 13th among the eastern 26 states, and produces 1% of the total nonfuel mineral production value in the United States. There are currently 13 minerals being extracted from 485 active mines in South Carolina alone. Portland and masonry cement and crushed stone were South Carolina's leading nonfuel minerals in 2004 (NMA 2007). In contrast, Georgia accounts for 4%, Florida accounts for 5%, and North Carolina accounts for about 2% of the total non-fuel mineral production value in the United States. North Carolina's leading nonfuel minerals in 2004 were crushed stone, phosphate rock, and construction sand and gravel. Georgia produces 24% of the clay in the nation; other leading nonfuel minerals include crushed stone and Portland cement. Florida is the top phosphate rock mining state in the United States and produces about six times more than any other state in the nation. Peat and zirconium concentrates are also produced in Florida.

The first gold mine operated in the United States is outside Charlotte, North Carolina in the Pee Dee watershed. Mines through Georgia are also major producers of barite and crude mica, iron oxide, and feldspar. There is a proposed titanium mine near the mouth of the Satilla River. Mines release toxic materials that negatively affect fish, as fish living around dredge tailings have elevated levels of mercury and selenium.

Hydromodification Projects. Several area rivers have been modified by dams and impoundments. In contrast to rivers along the Pacific Coast, considerable less information is available on other types of hydromodification projects in this area, such as levees and channelization projects. There are three locks and dams along the mainstem Cape Fear River and a large impoundment on the Haw River. The lower river and its tributaries are relatively undisturbed. The lower reach is naturally a blackwater river with naturally low dissolved oxygen, which is compounded by the reduced flow and stratification caused by upstream reservoirs and dams. The Yadkin (Pee Dee) River is heavily utilized for hydroelectric power. There are numerous dams on Santee-Cooper River System. The Santee River Dam forms Lake Marion and diverts the Santee River to the Cooper River, where another dam, St. Stephen Dam, regulates the outflow of the Santee River. Lake Moultrie is formed by both St. Stephen Dam and Pinopolis Dam, which regulates the flow of the Cooper River to the Atlantic Ocean. Below the fall line, the Savannah River is free-flowing with a meandering course, but above the fall line, there are three large dams that turn the Piedmont section of the river into a 100-mile long reservoir. Although the Altamaha River is undammed, hydropower dams are located on its tributaries, the Oconee and Ocmulgee rivers, above the fall lines. There are no dams along the entire mainstem Satilla River. There are no major dams on the mainstem St. Johns River either, but one of the largest tributaries has a dam on it. The St. Johns River’s flow is altered by water diversions for drinking water and agriculture.

Commercial and Recreational Fishing. The region is home to many commercial fisheries targeting shrimp, blue crab, clams, American and hickory shad, oysters, whelks, scallops, channel catfish, flathead catfish, snapper, and grouper. Shortnose sturgeon can be caught in gillnets, but gillnets and purse seines account for less than 2% of the annual bycatch. Shrimpers are responsible for 50% of all bycatch in Georgia waters. There are approximately 1.15 million recreational anglers in the state as well.

Southwest Coast Region

The basins described in this section are encompassed by the State of California and parts of Oregon. Select watersheds described herein characterize the general ecology and natural history of the area, and the past, present and future human activities and their impacts on the area. Essentially, this region encompasses all Pacific Coast rivers south of Cape Blanco, California through southern California. The Cape Blanco area marks a major biogeographic boundary and has been identified by NMFS as a DPS/ESU boundary for Chinook and coho salmon, and steelhead on the basis of strong genetic, life history, ecological and habitat differences north and south of this landmark. Major rivers contained in this grouping of watersheds are the

Sacramento, San Joaquin, Salinas, Klamath, Russian, Santa Ana and Santa Margarita Rivers (see Table 30).

Table 33. Select rivers in the southwest coast region

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Rogue River	211	5,154	CS, PB	38	10,065	23 (14)	11
Klamath River	287	15,679	PB, B/R, CS	33	17,693	48 (30)	41
Eel River	200	3,651	PB	52	7,416	25 (15)	12
Russian River	110	1,439	PB	41	2,331	41 (20)	43
Sacramento River	400	27,850	PB, CS, B/R	35	23,202	69 (29)	>50 T & E spp.
San Joaquin River	348	83,409	PB, CS	49	4,662	63	>50 T & E spp.
Salinas River	179	4,241	PB	14	448	36 (16)	42 T & E spp.
Santa Ana River	110	2,438	PB	13	60	45 (9)	54
Santa Margarita River	27	1,896	LC, PB	49.5	42	17 (6)	52

Data from Carter and Resh 2005
*Physiographic Provinces: PB = Pacific Border, CS = Cascades-Sierra Nevada Range, B/R = Basin & Range.

Natural History

The physiographic regions covered by the basins discussed herein include: (a) the Cascade-Sierra Nevada Mountains province, which extends beyond this region as we have defined it and continue north into British Columbia, (b) the Pacific Border province, and (c) the Lower California province (Carter and Resh 2005). The broader ecoregions division, as defined by Bailey (1995) is the Mediterranean Division. Three major vegetation types are encompassed by this region: the temperate coniferous forest, the Mediterranean shrub and savannah, and the temperate grasslands/savannah/shrub. The area, once dominated by native grasses, is naturally prone to fires caused by lightening during the dry season (Bailey 1995).

This region is the most geologically young and tectonically active region in North America. The Coast Range Mountains are folded and faulted formations, with a variety of soil types and nutrients that influence the hydrology and biology of the individual basins (Carter and Resh 2005). The region also covers the Klamath Mountains and the Sierra Nevada Range.

The climate is defined by hot dry summers and wet, mild winters, with precipitation generally decreasing in southern latitudes although precipitation is strongly influences by topography and generally increases with elevation. Annual precipitation varies from less than 10 inches to more than 50 inches in the region. In the Sierra Nevada about 50% of the precipitation occurs as snow (Carter and Resh 2005), as a result snowmelt strongly influences hydrological patterns in the area. Severe seasonal patterns of flooding and drought and high interannual variation in total precipitation makes the general hydrological pattern highly unpredictable within a basin, but consistant across years (Carter and Resh 2005). According to Carter and Resh (2005) this likely increases the variability in the annual composition of the fish assemblies in the region.

The San Joaquin River, draining the largest basin in the region, originates within the Sierra

Nevada Range near central California and flows in a northwesterly direction through the southern portion of the Central Valley. The alluvial fan of the Kings River separates the San Joaquin River from the Tulare River basin.

Human Activities and Their Impacts

Land Use. Land use is dominated by forest (and vacant land) in northern basins, and grass, shrubland, and urban uses dominate in southern basins (see Table 31). Overall, the most developed watersheds are the Santa Ana, Russian, and Santa Margarita rivers. The Santa Ana watershed encompasses portions of San Bernardino, Los Angeles, Riverside, and Orange counties. About 50% of coastal subbasin of the Santa Ana watershed is dominated by urban land uses and the population density is about 1,500 people per square mile. When steep and unbuildable lands are excluded from this area, then the population density in the watershed is 3,000 people per square mile. The most densely populated portion of the basin is near the City of Santa Ana, where density reaches 20,000 people per square mile (Burton 1998; Belitz et al. 2004). The basin is home to nearly 5 million people and the population is projected to increase two-fold in the next 50 years (Burton 1998; Belitz et al. 2004).

Table 34. Land uses and population density in several basins of the southwest coast region

Watershed	Land Use Categories (%)				Density (people/mi^2)
	Agriculture	Forest	Urban	Other	
Rogue River	6	83	<1	9 grass & shrub	32
Klamath River	6	66	<1	24 grass, shrub, wetland	5
Eel River	2	65	<1	31 grass & shrub	9
Russian River	14	50	3	31 (23 grassland)	162
Sacramento River	15	49	2	30 grass & shrub	61
San Joaquin River	30	27	2	36 grass & shrub	76
Salinas River	13	17	1	65 (49 grassland)	26
Santa Ana River	11	57	32	---	865
Santa Margarita River	12	11	3	71 grass & shrub	135

Data from Carter and Resh 2005

Not only is the Santa Ana watershed the most heavily developed watershed in the region, the Santa Ana is the most heavily populated study site out of more than 50 assessment sites studied across the nation by the United States Geological Survey (USGS) under the National Water-Quality Assessment (NAWQA) Program. Water quality and quantity in the basin reflects the influence of the high level of urbanization. For instance, the primary source of baseflow to the river is the treated wastewater effluent; secondary sources that influence peak flows include stormwater runoff from urban, agricultural, and undeveloped lands (Belitz et al. 2004). Concentrations of nitrates and pesticides are elevated within the basin, and were more frequently detected than in other national NAWQA sites (Leenheer et al. 2008; Kent et al. 2005; Belitz et al. 2004). Belitz et al. (2004) found that total nitrogen concentrations commonly exceeded 3 mg/L in the Santa Ana basin. In other NAWQA basins with elevated total nitrogen concentrations across the country, the primary influencing factor was the level of agriculture and the application of manure and pesticides within the basin. In the Santa Ana basin the elevated nitrogen is attributed largely to the wastewater treatment plants, where downstream reaches consistently exceeding 3 mg/L total nitrogen. Samples of total nitrogen taken upstream of the wastewater treatment plants were commonly below 2 mg/L (Belitz et al. 2004). Other contaminants detected at high levels included volatile organic compounds (VOCs; including chloroform, which sometimes exceeded water quality standards), pesticides (diuron, diazinon, carbaryl, chlophyrifos, lindane, malathion, and chlorothalonil), and trace elements (lead, zinc, and arsenic). As a result of the changes, the biological community in the basin is heavily altered (Belitz et al. 2004).

In many basins, agriculture is the major water user and the major source of water pollution to surface waters. In 1990, nearly 95% of the water diverted from the San Joaquin River was diverted for agriculture, and 1.5% diverted for livestock (Carter and Resh 2005). During the same period, Fresno, Kern, Tulare, and Kings counties ranked top in the nation for nitrogen fertilizer use. Nitrogen fertilizer use increased 500% and phosphorus use increased 285% in the San Joaquin River basin over a 40-year period (Knatzer and Sheton 1998 *in* Carter and Resh 2005). A study conducted by USGS in the mid-1990s on water quality within San Joaquin River basin detected 49 pesticides in the mainstem and three subbasins; 22 pesticides were detected in 20% of the samples and concentrations of seven exceeded water quality standards (Dubrovsky et al. 1998). Water chemistry in the Salinas River is strongly influenced by intensive agriculture; water hardness, alkalinity, nutrients and conductivity are high in areas where agricultural uses predominate.

Estuary systems of the region are consistently exposed to anthropogenic pressures stemming from high human density sources. As an example, the largest west coast estuary, the San Francisco Estuary, provides drinking water to 23 million people, irrigates 4.5 million acres of farmland, and drains roughly 40% of California's land area. As a result of high use, many environmental measures of the estuary are poor. Water quality suffers from high phosphorus and nitrogen loads, primarily from agricultural, sewage, and storm water runoff. Water clarity is also compromised. Sediments contain high levels of the contaminants PCB, pesticides, mercury, copper, and silver from urban runoff and historical activities. As these persist in the marine environment, the estuary system will likely carry loads for years to come, even with strict regulation or banning. Gold mining has reduced estuary depths in much of the region, causing drastic changes to habitat. Large urban centers are foci for contaminants and levels near San Francisco, Oakland, and San Jose are highest and are also where water clarity tends to be at its worst. These water and sediment quality characteristics biomagnify into the food chain; fish tissues contain high levels of particularly PCB and mercury, the former being concentrated 10 times more than human health guidelines for consumption. Birds, some of whom are endangered (clapper rail and least tern), have also concentrated these toxins.

Invasive species have become an increasingly recognized issue. Giant reeds have displaced native marsh species in many areas. Marine invasive species include the green crab, shimofuri goby, Asiatic clams, and zooplankton; these species are cited in reducing the abundance of local species. The Asian clam has become the dominant infaunal species and has likely reduced primary production in the estuary system (Nichols et al. 1990; Ray 2005).

Red tide significantly affects the California coastline. Here, poisoning and mortality of California sea lions, fish, and birds have been recorded, the most recent of which was in 2007 (Chea 2007). California red tide events are correlated with El Niño oscillations. In addition to the toxin produced by red tide diatoms, a pathogen associated with cholera has been identified in California red tide blooms (Mouriño-Pérez et al. 2003).

Hydromodification Projects. Several of the rivers within the area have been modified by dams, water diversions, and drainage systems for agriculture and drinking water, and some of the most drastic channelization projects in the nation. In all, there are about 1,400 dams within the State of California, more than 5,000 miles of levees, and more than 140 aqueducts (Mount 1995 *in* Carter and Resh 2005). While about 75% of the runoff occurs in basins in the northern half of California, 80% of the water demand is in the southern half. Two water diversion projects meet these demands—the Federal Central Valley Project and the California State Water Project. The Central Valley Project, one of the world's largest water storage and transport systems, has more than 20 reservoirs and delivers about 7 million acre-feet per year to southern California. The State Water Project has 20 major reservoirs and holds nearly 6 million acre-feet of water, delivering about 3 million acre feet. Together these diversions irrigate about 4 million acres of farmland and deliver drinking water to roughly 22 million residents and growing.

Both the Sacramento and San Joaquin rivers are heavily modified, each with hundreds of dams. In 2009, the Sacramento-San Joaquin river system was named America's most endangered river by American Rivers. In the prior year, the Rogue River was listed as the second most endangered river. The Rogue, Russian, and Santa Ana rivers each have more than 50 dams, and the Eel, Salinas and the Klamath rivers have between 14 and 24 dams each. The Santa Margarita, considered one the last free flowing rivers in coastal southern California has nine dams in its watershed. All major tributaries of the San Joaquin River are impounded at least once and most have multiple dams or diversions. The Stanislaus River, a tributary of the San Joaquin River, has over 40 dams. As a result, the hydrograph of the San Joaquin River is seriously altered from its natural state, and the temperature regime and sediment transport regime are altered. Such changes have had profound influences on the biological community within the basin. These modifications generally result in a reduction of suitable habitat for native species and frequent concomitant increases in suitable habitat for nonnative species. The Friant Dam on the San Joaquin River is attributed with the extirpation of spring-run Chinook salmon within the basin, a run once estimated as producing 300,000 to 500,000 fish (Carter and Resh 2005).

Mining. Famous for the gold rush of the mid 1800s, California has a long history of mining. In 2004, California ranked top in the nation for nonfuel mineral production with 8.23% of total production (NMA 2007). Today, gold, silver, and iron ore comprise only 1% of the production value. Primary minerals include construction sand and gravel, cement, boron and crushed stone. California is the only state to produce boron, rare-earth metals, and asbestos (NMA 2007).

California contains some 1,500 abandoned mines and roughly 1% are suspected of discharging metal-rich waters in the basins. The Iron Metal Mine in the Sacramento Basin releases more than 1,100 pounds of copper and more than 770 pounds of zinc to the Keswick Reservoir below Shasta Dam, as well as elevated levels of lead (Cain et al. 2000 *in* Carter and Resh 2005). Metal contamination seriously reduces the biological productivity within a basin and can result in fish kills at high levels or sublethal effects at low levels, including reduced feeding, overall activity levels, and growth. The Sacramento Basin and the San Francisco Bay watershed is one of the most heavily affected basins within the state from mining activities, largely because the basin drains some of the most productive mineral deposits in the region. Methylmercury contamination within San Francisco Bay, the result of 19th century mining practices using mercury to amalgamate gold in the Sierra Nevada Mountains, remains a persistent problem today. Based on sediment cores, we know that pre-mining concentrations were about five times lower than concentrations detected within San Francisco Bay today (Conaway et al. 2003 in EPA 2006).

Commercial and Recreational Fishing. The region is home to many commercial fisheries. The largest in terms of total landings in 2006 were northern anchovy, Pacific sardine, Chinook salmon, sablefish, Dover sole, Pacific whiting, squid, red sea urchin, and Dungeness crab (CDFG 2007). Red abalone are also harvested. The first salmon cannery established along the west coast was located in the Sacramento River watershed in 1864, but it only operated for about two years because the sediment from hydraulic mining decimated the runs in the basin (NRC 1996).

Pacific Northwest Region

This region encompasses Washington, Oregon, Idaho, and includes parts of Nevada, Montana, Wyoming, and British Columbia. The region is ecologically diverse, encompassing northern marine lowland forests, mountain forests, alpine meadows, and northern desert habitat. In this section we focus on three primary areas that characterize the region, the Columbia River Basin and its tributaries, the Puget Sound Region, and the coastal drainages north of the Columbia River. The broader ecoregion divisions, as defined by Bailey (1995) and encompassed within this region, are the Marine Division, Marine Division – Mountain Provinces, Temperate Steppe Division, Temperate Steppe Division – Mountain Provinces, and portions of the Temperate Desert Davison. Puget Sound and the coastal drainages are contained within the Marine Division, while the Columbia River watershed encompasses portions of all five ecoregions.

Columbia River Basin

Natural History

The most notable basin within the region is the Columbia River. The largest river in the Pacific Northwest and the fourth largest river in terms of average discharge in the United States, it drains over 258,000 square miles, making it the sixth largest in terms of drainage area. The Columbia River Basin includes parts of Washington, Oregon, Nevada, Utah, Idaho, Wyoming, Montana, and British Columbia and encompasses 13 terrestrial and three freshwater ecoregions, including arid shrub-steppes, high desert plateaus, temperate mountain forests, and deep gorges (Kammerer 1990; Hinck et al. 2004; Stanford et al. 2005).

Major tributaries include the Snake, Willamette, Salmon, Flathead, and Yakima Rivers; smaller rivers include the Owyhee, Grande Ronde, Clearwater, Spokane, Methow, Cowlitz, and the John Day Rivers (see Table 32 for a description of select Columbia River tributaries). The Snake River is the largest tributary at more than 1,000 miles long; its headwaters originating in Yellowstone National Park, Wyoming. The second largest tributary is the Willamette River in Oregon (Kammerer 1990; Hinck et al. 2004) and the 19th largest river in the nation in terms of average annual discharge (Kammerer 1990). The basins drain portions of the Rocky Mountains, Bitteroot Range, and the Cascade Range.

The average annual discharge at the mouth of the Columbia River is 265,000 cubic feet per second (Kammerer 1990). A saltwater wedge extends 23 miles upstream of the mouth with tidal influences extending up to 146 miles up river (Hinck et al. 2004). The climate within the basin is a mix of arid, dry summers, cold winters, and maritime air masses entering from the west. It is not uncommon for air temperatures in the Rocky Mountains to dip below zero in mid-winter, but summer air temperatures can reach more than 100° F in the middle basin.

Table 35. Select tributaries of the Columbia River

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precip. (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Snake/Salmon	870	108,495	CU, NR, MR, B/R	14	55,267	39 (19)	5 fish (4 T, 1 E), 6 (1 T, 5 E) snails, 1 plant (T)
Yakima	214	6,139	CS, CU	7	3,602	50	2 (T)
Willamette	143	11,478	CS, PB	60	32,384	61 (~31)	5 fish (4 T, 1 E),

Data from Carter and Resh 2005
*Physiographic Provinces: CU = Columbia-Snake River Plateaus, NR = Northern Rocky Mountains, MR = Middle Rocky Mountains, B/R = Basin & Range, CS = Cascade-Sierra Mountains, PB = Pacific Border

The river and estuary were once home to more than 200 distinct runs of Pacific salmon and steelhead with unique adaptations to local environments within a tributary (Stanford et al. 2005). Salmonids within the basin include Chinook, chum, coho, sockeye salmon, steelhead and redband trout, bull trout, and cutthroat trout. Other fish species within the basin include sturgeon, eulachon, lamprey, and sculpin (Wydoski and Whitney 1979). According to a review by Stanford et al. (2005), the basin formerly contained 65 native fish species and at least 53 nonnative fishes. The most abundant non-native fish is the American shad, which was introduced to the basin in the late 1800s (Wydoski and Whitney 1979).

Human Activities and Their Impacts

Land Use. More than 50% of the United States portion of the Columbia River Basin is in Federal ownership (most of which occurs in high desert and mountain areas), 39% is in private land ownership (most of which occurs in river valleys and plateaus), and the remainder is divided among tribes, state, and local governments (Hinck et al. 2004). See Table 33 for a summary of land uses and population densities in several subbasins within the Columbia River watershed (data from Stanford et al. 2005).

Table 36. Land uses and population density in select tributaries of the Columbia River

Watershed	Land Use Categories (%)				Density ($people/mi^2$)
	Agriculture	Forest	Urban	Other	
Snake/Salmon rivers	30	10-15	1	54 scrub/rangeland/barren	39
Yakima River	16	36	1	47 shrub	80
Willamette River	19	68	5	--	171

Data from Stanford et al. 2005

The interior Columbia Basin has been altered substantially by humans causing dramatic changes and declines in native fish populations. In general the basin supports a variety of mixed uses. Predominant human uses include logging, agriculture, ranching, hydroelectric power generation, mining, fishing, a variety of recreational activities, and urban uses. The decline of salmon runs in the Columbia River is attributed to loss of habitat, blocked migratory corridors, altered river flows, pollution, overharvest, and competition from hatchery fish. Critical ecological connectivity (mainstem to tributaries and riparian floodplains) has been disconnected by dams and associated activities such as floodplain deforestation and urbanization. The most productive floodplains of the watershed are either flooded by hydropower dams or dewatered by irrigation diversions. Portions of the basin are also subject to impacts from cattle grazing and irrigation withdrawals. In the Yakima River, 72 stream and river segments are listed as impaired by the Washington Department of Ecology and 83% exceed temperature standards. In the Willamette River, riparian vegetation was greatly reduced by land conversion. By 1990, only 37% of the riparian area within 120 m was forested, 30% was agricultural fields and 16% was urban or suburban lands. In the Flathead River, aquatic invasive plants such as pondweed, hornwort, water milfoil, waterweed, cattail, and duckweed grow in the floodplain wetlands and shallow lakes. In the Yakima River, non-native grasses and other plant are commonly found along the lower reaches of the river (Stanford et al. 2005).

Agriculture and Ranching. Roughly 6% of the annual flow from the Columbia River is diverted for the irrigation of 7.3 million acres of croplands within the basin. The vast majority of these agricultural lands are located along the lower Columbia River, the Willamette, Yakima, Hood, and Snake rivers, and the Columbia Plateau (Hinck et al. 2004). The Yakima River Basin is one of the most agriculturally productive areas in the United States (Fuhrer et al. 2004). Croplands within the Yakima Basin account for about 16% of the total basin area of which 77% is irrigated.

Agriculture and ranching increased steadily within the Columbia River basin from the mid to late 1800. By the early 1900s, agricultural opportunities began increasing at a much more rapid pace with the creation of more irrigation canals and the passage of the Reclamation Act of 1902 (NRC 2004). Today, agriculture represents the largest water user within the basin (>90%). Agriculture, ranching, and related services employ more than nine times the national average (19% of the households within the basin; NRC 2004).

Ranching practices have led to increased soil erosion and sediment loads within adjacent tributaries, the worst of these effects may have occurred in the late 1800s and early 1900s from deliberate burning to increase grass production (NRC 2004). Several measures are in use to reduce the impacts of grazing, including restricting grazing in degraded areas, reduced grazing allotments, and lower stocking rates. Today, agricultural impacts to water quality within the basin are second to large-scale influences of hydromodification projects for both power generation and irrigation. Water quality impacts from agricultural activities include alteration of the natural temperature regime, and insecticide and herbicide contamination, and increased suspended sediments.

The USGS has a number of fixed water quality sampling sites throughout various tributaries of the Columbia River, many of which have been in place for decades. Water volumes, crop rotation patterns, crop-type, and basin location are some of the variables that influence the distribution and frequency of pesticides within a tributary. Detection frequencies for a particular pesticide can vary widely. One study conducted by the USGS between May 1999 and January 2000 detected 25 pesticide compounds (Ebbert and Embrey 2001). Another study detected at least two pesticides or their breakdown products in 91% of the samples collected, with the median number of chemicals being eight, and a maximum of 26. The herbicide 2,4-D occurred most often in the mixtures, along with azinphos-methyl, the most heavily applied pesticide, and atrazine, one of the most aquatic mobile pesticides (Fuhrer et al. 2004). However, the most frequently detected pesticides in the Yakima River Basin are total DDT, as well as its breakdown products DDE and DDD, and dieldrin (Johnson and Newman 1983; Joy 2002; Joy and Madrone 2002; Furher et al. 2004). In addition to current-use chemicals, these legacy chemicals continue to pose a serious problem to water quality and fish communities despite their ban in the 1970s and 1980s (Hinck et al. 2004).

Fish and macroinvertebrate communities exhibit an almost linear decline in condition as the level of agriculture intensity increases within a basin (Cuffney et al. 1997; Fuhrer et al. 2004). A study conducted in the late 1990s examined 11 species of fish, including anadromous and resident fish collected throughout the Columbia River Basin for a suite of 132 contaminants, including 51 semi-volatile chemicals, 26 pesticides, 18 metals, seven PCBs, 20 dioxins, and 10 furans. The study revealed PCBs, metals, chlorinated dioxins and furans (products of wood pulp bleaching operations) and other contaminants within fish tissues; white sturgeon tissues contained the greatest concentrations of chlorinated dioxins and furans (Hinck et al. 2004).

Urban and Industrial Development. The largest urban area in the basin is the greater Portland metropolitan area, located at the mouth of the Columbia River. Portland’s population exceeds 500,000, and the next largest cities, Spokane, Salem, Eugene, and Boise, have over 100,000 people (Hinck et al. 2004). Overall, the basin’s population density is one-third the national average, and while the basin covers about 8% of United States land, only about 1.2% of the United States population lives within the basin (Hinck et al. 2004).

Discharges from sewage treatment plants, paper manufacturing, and chemical and metal production represent the top three permitted sources of contaminants within the lower basin according to discharge volumes and concentrations (Rosetta and Borys 1996 in Hinck et al. 2004). According to Rosetta and Borys (1996 in Hinck et al. 2004), based on their review of 1993 data, 52% of the point source waste water discharge volume is from sewage treatment plants, 39% from paper and allied products, 5% from chemical and allied products, and 3% from primary metals (Rosetta and Borys 1996 in Hinck et al. 2004). The paper and allied products industry is the primary source of the suspended sediment load (71%), while 26% comes from sewage treatment plants, and 1% is from the chemical and allied products industry. Non-point source discharges (urban stormwater runoff) account for significant pollutant loading to the lower basin, including most organics and over half of the metals. Although rural non-point sources contributions were not calculated, Rosetta and Borys (1996 in Hinck et al. 2004) surmised that in some areas and for some contaminants, rural areas may contribute a large portion of the load. This is particularly true for pesticide contamination in the upper river basin where agriculture is the predominant land use.

The Columbia River Estuary is under threat from several anthropogenic sources. Habitat loss has fragmented habitat and human density increase has created additional loads of pollutants and contaminants (EPA 2006). Water quality has been reduced by phosphorus loads and decreased water clarity, primarily along the lower and middle sections of the Columbia River Estuary. Although sediment quality is generally very good, benthic indices have not been established within the estuary, and fish tissue contaminant loads (PCBs, DDT, DDD, DDE, and mercury) are high, presenting a persistent and long lasting effect on estuary biology. Health advisories have been recently issued for people eating fish in the area that contain high levels of dioxins, PCBs, and pesticides. Habitat loss has been significant; 77% of swamps, 57% of marshes, and over 20% of tree cover has been lost to development and industry. Twenty-four threatened and endangered species occur in the estuary, some of whom are recovering and others (i.e., Chinook salmon) are not. Issues surrounding damming and environmental toxins have played key roles in original decline and subsequent recovery of several species and will be vital for future management. Invasive species in the estuary are pervasive; at least 81 have currently been identified, composing one-fifth of all species in some areas, and new species are being identified presently.

Hydromodification Projects. More than 400 dams exist in the basin, ranging from mega dams that store large amounts of water to small diversion dams for irrigation. Every major tributary of the Columbia River except the Salmon River is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects of which 18 dams are located on mainstem Columbia River and its major tributary, the Snake River. The Federal Columbia River Power System encompasses the operations of 14 major dams and reservoirs on the Columbia and Snake rivers, operated as a coordinated system. The Army Corps of Engineers operates nine of 10 major Federal projects on the Columbia and Snake rivers, and Dworshak, Libby and Albeni Falls dams. The Bureau of Reclamation operates Grand Coulee and Hungry Horse dams. These Federal projects are a major source of power in the region, and provide flood control, navigation, recreation, fish and wildlife, municipal and industrial water supply, and irrigation benefits.

The Bureau of Reclamation has operated irrigation projects within the basin since 1904. The irrigation system delivers water to about 2.9 million acres of agricultural lands; 1.1 million acres of land are irrigated using water delivered by two structures, the Columbia River Project (Grand Coulee Dam) and the Yakima Project. Grand Coulee Dam delivers water for the irrigation of over 670,000 acres of croplands and the Yakima Project delivers water to nearly 500,000 acres (BOR 2007).

The Bonneville Power Administration, an agency under the U.S. Department of Energy, wholesales electric power produced at 31 Federal dams (67% of its production) and non-hydropower facilities in the Columbia-Snake Basin, selling about half the electric power consumed in the Pacific Northwest. The Federal dams were developed over a 37-year period starting in 1938 with Bonneville Dam and Grand Coulee in 1941, and ending with construction of Libby Dam in 1973 and Lower Granite Dam in 1975.

Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on the ecosystems of the Columbia River Basin (ISG 1996). These effects have been especially adverse to the survival of anadromous salmonids. The construction of the Federal power system modified migratory habitat of adult and juvenile salmonids, and in many cases presented a complete barrier to habitat access. Both upstream and downstream migrating fish are impeded by the dams, and a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delays in passage, exposure to high concentrations of dissolved gases, warm water, and increased predation. Dams have also flooded historical spawning and rearing habitat with the creation of massive water storage reservoirs. More than 55% of the Columbia River Basin that was accessible to salmon and steelhead before 1939 has been blocked by large dams (NWPPC 1986). Construction of Grand Coulee Dam blocked 1,000 miles of habitat from migrating salmon and steelhead (Wydoski and Whitney 1979). The mainstem habitats of the lower Columbia and Willamette rivers have been reduced primarily to a single channel. As a result, floodplain area is reduced, off-channel habitat features have been eliminated or disconnected from the main channel, and the amount of large woody debris in the mainstem has been reduced. Remaining areas are affected by flow fluctuations associated with reservoir management for power generation, flood control and irrigation. Overbank flow events, important to habitat diversity, have become rare as a result of controlling peak flows and associated revetments. Consequently, estuary dynamics have changed substantially.

Artificial Propagation. There are several artificial propagation programs for salmon production within the Columbia River Basin, many of which were instituted under Federal law to ameliorate the effects of lost natural salmon production within the basin from the dams. The hatcheries are operated by Federal, state, and tribal managers. For more than 100 years, hatcheries in the Pacific Northwest have been used to produce fish for harvest and replace natural production lost to dam construction, and have only minimally been used to protect and rebuild naturally produced salmonid population (e.g., Redfish Lake sockeye salmon). In 1987, 95% of the coho salmon, 70% of the spring Chinook salmon, 80% of the summer Chinook salmon, 50% of the fall Chinook salmon, and 70% of the steelhead returning to the Columbia River Basin originated in hatcheries (CBFWA 1990). More recent estimates suggest that almost half of the total number of smolts produced in the basin come from hatcheries (Mann et al. 2005).

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive (Hard et al. 1992). Hatchery practices, among other factors, are a contributing factor to the 90% reduction in natural coho salmon runs in the lower Columbia River over the past 30 years (Flagg et al. 1995). Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from nonnative basins, and the impacts of these practices are largely unknown. Adverse effects of these practices likely included loss of genetic variability within and among populations (Busack 1990 *in* Hard et al. 1992; Riggs 1990; Reisenbichler 1997), disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and displacement of natural fish (Steward and Bjornn 1990; Fresh 1997). Species with extended freshwater residence are likely to face higher risk of domestication, predation, or altered migration than are species that spend only a brief time in fresh water (Hard et al. 1992). Nonetheless, artificial propagation also may contribute to the conservation of listed salmon and steelhead although it is unclear whether or how much artificial propagation during the recovery process will compromise the distinctiveness of natural population (Hard et al. 1992).

Currently, NMFS is working on a hatchery reform project in the Columbia River Basin, which will include a collaborative review of how harvest and hatcheries (particularly Federally funded hatcheries) are affecting the recovery of listed salmon and steelhead in the basin. This effort was mandated by Congress in 2005, and is in its early stages. Eventually, the project team would create a management approach that allows tribal, state and Federal managers to effectively manage Columbia River Basin hatcheries to meet conservation and harvest goals consistent with their respective legal responsibilities.

Mining. Most of the mining in the basin is focused on minerals such as phosphate, limestone, dolomite, perlite, or metals such as gold, silver, copper, iron, and zinc. Mining in the region is conducted in a variety of methods and places within the basin. Alluvial or glacial deposits are often mined for gold or aggregate, and ores are often excavated from the hard bedrocks of the Idaho batholiths. Eleven percent of the nation's output of gold has come from mining operations in Washington, Montana, and Idaho, and more than half of the nation's silver output has come from a few select silver deposits, with 30% coming from two deposits in the Columbia River Basin (the Clark Fork River and Coeur d'Alene deposits; Hinck et al. 2004, Butterman and Hilliard 2005). According to Wydoski and Whitney (1979), one of the largest mines in the region, located near Lake Chelan, once produced up to 2,000 tons of copper-zinc ore with gold and silver on a daily basis. Most of the phosphate mining within the basin occurs in the headwaters of the Snake River; the overall output from these deposits accounts for 12% of United States phosphate production (Hinck et al. 2004).

Many of the streams and river reaches in the basin are impaired from mining and several abandoned and former mining sites are designated as Superfund cleanup areas (Stanford et al. 2005; EPA 2007). According to the United States Bureau of Mines, there are about 14,000 inactive or abandoned mines within the Columbia River Basin of which nearly 200 pose a potential hazard to the environment (Quigley et al. 1997 *in* Hinck et al. 2004). Contaminants detected in the water include lead and other trace metals. Mining of copper, cadmium, lead, manganese, and zinc in the upper Clark Fork River have contributed wastes to this basin since 1880 (Woodward et al. 1994). Benthic macroinvertebrates and fish within the basin have bioaccumulated metals, which are suspected of reducing their survival and growth (Farag et al. 1994; Woodward et al. 1994). In the Clark River, several fish kills have occurred since 1984 and are attributed to contamination from trace metals such as cadmium, copper, lead, and zinc (Hinck et al. 2004).

Commercial, Recreational, and Subsistence Fishing. Archeological records indicate that indigenous people caught salmon in the Columbia River more than 7,000 years ago. One of the most well known tribal fishing sites within the basin was located near Celilo Falls, an area in the lower river that has been occupied by Dalles Dam since 1957. Salmon fishing increased with better fishing methods and preservation techniques, such as drying and smoking, such that harvest substantially increased in the mid-1800s with canning techniques. Harvest techniques also changed over time, from early use of hand-held spears and dip nets, to riverboats that used seines and gill-nets, eventually, transitioning to large ocean-going vessels with trolling gear and nets and the harvest of Columbia River salmon and steelhead from California to Alaska (Mann et al. 2005).

During the mid-1800s, an estimated 10 to 16 million adult salmon of all species entered the Columbia River each year. Large harvests of returning adult salmon during the late 1800s ranging from 20 million to 40 million pounds of salmon and steelhead annually significantly reduced population productivity (Mann et al. 2005). The largest known harvest of Chinook salmon occurred in 1883 when Columbia River canneries processed 43 million pounds of salmon (Lichatowich 1999). Commercial landings declined steadily from the 1920s to a low in 1993, when just over one million pounds were harvested (Mann et al. 2005).

Harvested and spawning adults reached 2.8 million in the early 2000s, of which almost half are hatchery produced (Mann et al. 2005). Most of the fish caught in the river are steelhead and spring/summer Chinook salmon, while ocean harvest consists largely of coho and fall Chinook salmon. Most ocean catches are made north of Cape Falcon, Oregon. Over the past five years, the number of spring and fall salmon commercially harvested in tribal fisheries has averaged between 25,000 and 110,000 fish (Mann 2004 in Mann et al. 2005). Recreational catch in both ocean and in-river fisheries varies from 140,000 to 150,000 individuals (Mann et al. 2005).

Puget Sound Region

Natural History

The Puget Sound watershed is defined by the crest lines of the Olympia Mountain Range (and the Olympic Peninsula) to the west and the Cascade Range to the east. The Olympic Mountains reach heights of about 8,000 feet, and are extremely rugged and steeply peaked with abrupt descents into the Puget Lowland. The Cascade Mountains range in heights of 4,000 to 8,000 feet with the highest peak, Mount Rainer, towering at 14,410 feet above sea level. As the second largest estuary in the United States, Puget Sound has about 1,330 miles of shoreline and extends from the mouth of the Strait of Juan de Fuca east, including the San Juan Islands and south to Olympia, and is fed by more than 10,000 rivers and streams.

Puget Sound is generally divided into four major geographic marine basins: Hood Canal, South Sound, Whidbey Basin, and the Main Basin. The Main Basin has been further subdivided into two sub-basins: Admiralty Inlet and Central Basin. Each of the above basins forms a depression on the sea floor in which a shallower ledge or sill separates the relatively deep water from the adjacent basin. The waters of Puget Sound function as a partially mixed, two-layer system, with relatively fresh water flowing seaward at the surface and salty oceanic water entering at depth.

The main ledge of Puget Sound is located at the north end of Admiralty Inlet where the water shoals to a depth of about 200 feet at its shallowest point (King County 2001). The deepest point in Puget Sound is in the Central Basin at over 920 feet in depth. Approximately 43% of the Puget Sound’s tideland is located in the Whidbey Island Basin. This reflects the large influence of the Skagit River, which is the largest river in the Puget Sound system and whose sediments are responsible for the extensive mudflats and tidelands of Skagit Bay.

Habitat types that occur within the nearshore environment include eelgrass meadows, kelp forest, mud flats, tidal marshes, subestuaries (tidally influenced portions of river and stream mouths), sand spits, beaches and backshore, banks and bluffs, and marine riparian vegetation. These habitats provide critical functions such as primary food production and support habitat for invertebrates, fish, birds, and other wildlife.

The Puget Sound ecoregion is a glaciated area consisting of glacial till, glacial outwash and lacustrine deposits with high quality limestone in the San Juan Islands (Wydoski and Whitney 1979). Relief in the valley is moderate, with elevation ranging from sea level to about 1,300 feet. Geology in the region consists of mostly Tertiary sedimentary bedrock formations.

The land and vegetation surrounding Puget Sound waters is classified as Puget Lowland Forest and occupies the depression or valley between the Olympic Peninsula on the west and the Cascade Range to the east (Franklin and Dyrness 1973). The alpine zone is expressly devoid of trees. Vegetation changes abruptly along the mountain slopes and across minimal horizontal distances as a result of steep topography, soil, and microclimate (sun exposure, temperature, and precipitation). Dominant vegetation types include the Puget lowland region – the lowland forest and the mid-montane forest of Pacific silver fir and Alaska yellow cedar; the subalpine forest of mountain hemlock with subalpine fir and Alaska yellow cedar; and the alpine tundra or meadow above the treeline (Kruckeberg 1991).

The Puget Sound region has a Mediterranean-like climate, with warm, dry summers, and mild wet winters (Franklin and Dyrness 1973). Annual precipitation varies from 28 to 35 inches, and falls predominantly as rain in lowland areas. Annual snowpack in the mountain ranges is often high; although the elevation of the Olympic Mountains is not as high as that of the Cascade Mountain Range, abundant accumulation occurs, such that it will sometimes persist throughout much of the summer. Average annual rainfall in the north Cascades at Mount Baker Lodge is about 110 inches, and at Paradise Station at Mount Rainer is about 105 inches, while average annual snowfall is 550 inches and 582 inches respectively, sometimes reaching more than 1,000 inches on Mount Rainer (Wydoski and Whitney 1979; Kruckeberg 1991).

Major rivers draining to Puget Sound from the Cascade Mountains include the Skagit, Snohomish, Nooksack, Puyallup, and Green rivers, as well as the Lake Washington/Cedar River watershed. Major rivers from the Olympic Mountains include the Hamma Hamma, the Duckabush, the Quilcene, and the Skokomish rivers. Numerous other smaller rivers drain to the Sound, many of which provide important salmonid habitats despite their small size.

The Puget Sound basin is home to more than 200 fish and 140 mammalian species. Salmonids within the region include coho, Chinook, sockeye, chum, and pink salmon, kokanee, steelhead, rainbow, cutthroat, and bull trout, as well as Dolly Varden (Wydoski and Whitney 1979; Kruckeberg 1991). Important commercial fishes include the five Pacific salmon and several rockfish species. A number of introduced species occur within the region, including brown and brook trout (*Salvelinus fontinali*s), Atlantic salmon, bass, tunicates (sea squirts), and a saltmarsh grass (*Spartina* spp.). Estimates suggest that more than 90 species have been intentionally or accidentally introduced in the region (Ruckelshaus and McClure 2007). At present over 40 species in the region are listed as threatened and endangered under the ESA.

Human Activities and Their Impacts

Land Use. Land use in the Puget Sound lowland is composed of agricultural areas (including forests for timber production), urban areas (industrial and residential use), and rural areas (low density residential with some agricultural activity). In the 1930s, all of western Washington contained about 15.5 million acres of "harvestable" forestland and by 2004 the total acreage was nearly half that originally surveyed (PSAT 2007). Forest cover in Puget Sound alone was about 5.4 million acres in the early 1990s and about a decade later the region had lost another 200,000 acres of forest cover with some watersheds losing more than half the total forested acreage. The most intensive loss of forest cover occurred in the Urban Growth Boundary, which encompasses specific parts of the Puget Lowland; in this area forest cover declined by 11% between 1991 and 1999 (Ruckelshaus and McClure 2007). Projected land cover changes (reviewed in Ruckelshaus and McClure 2007) indicate that trends are likely to continue over the next several decades with population changes; coniferous forests are projected to decline at an alarming rate as urban uses increase.

The Puget Sound Lowland contains the most densely populated area of Washington. The regional population in 2003 was an estimated 3.8 million people, with 86% residing in King, Pierce and Snohomish counties (Snohomish, Cedar-Sammamish Basin, Green-Duwamish, and Puyallup River watersheds), and the area is expected to attract four to six million new human residents in the next 20 years (Ruckelshaus and McClure 2007). According to the State of the Sound report (PSAT 2007) in 2001, impervious surfaces covered 3.3% of the region, with 7.3% of lowland areas (below 1,000 feet elevation) covered by impervious surfaces. In one decade, 1991 – 2001 impervious surfaces increased 10.4% region wide. The Snohomish River watershed, one of the fastest growing watersheds in the region, increased about 16% in the same period.

Much of the region's estuarine wetland losses have been heavily modified, primarily from agricultural land conversion and urban development (NRC 1996). Although most estuarine wetland losses result from conversions to agricultural land by ditching, draining, or diking, these wetlands are also experiencing increasing effects from industrial and urban causes. The most extreme case of river delta conversion is observed in the Duwamish Waterway in Seattle. As early as the mid-1800s, settlers in the region began discussing the need for a ship canal that linked Lake Washington directly with Puget Sound. After several private and smaller attempts, by the early 1900s locks were built achieving this engineering feat. The result was that the Black River, which formerly drained Lake Washington to the Green and White rivers (at their confluence, these rivers formed the Duwamish River), dried up. The lower White River, which historically migrated sporadically between the Puyallup and the Green/Duwamish basins, was permanently diverted into the Puyallup River basin in 1914 with the construction of a concrete diversion at river mile 8.5, resulting in a permanent increase of Puyallup River flow by about 50% and a doubling of the drainage area (Kerwin 1999). The Cedar River, on the other hand was permanently diverted to Lake Washington. The oxbow in the lower Duwamish River was lost with the lower river dredging in the early 1900s, reducing the lower nine miles of the river to 5 miles in length. Over time, the Duwamish Waterway has been heavily armored and diked, result in the loss of all tidal swamps, 98% of the tidal forests, marshes, shallows and flats and 80% of the riparian shoreline (Blomberg et al. 1988). By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at eleven deltas in Puget Sound (Bortleson et al. 1980). Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50-90%.

More than 100 years of industrial pollution and urban development have affected water quality and sediments in Puget Sound. Many different kinds of activities and substances release contamination into Puget Sound and the contributing waters. Positive changes in water quality in the region are also evident. One of the most notable improvements was the elimination of sewage effluent to Lake Washington in the mid 1960s, which significantly reduced problems within the lake from phosphorus pollution and triggered a concomitant reduction in cyanobacteria (Ruckelshaus and McClure 2007). Even so, as the population and industry has risen in the region a number of new and legacy pollutants are of concern. According to the State of the Sound Report (PSAT 2007) in 2004, more than 1,400 fresh and marine waters in the region were listed as “impaired.” Almost two-thirds of these water bodies were listed as impaired due to contaminants, such as toxics, pathogens, and low dissolved oxygen or high temperatures, and less than one-third had established cleanup plans. More than 5,000 acres of submerged lands (primarily in urban areas; 1% of the study area) are contaminated with high levels of toxic substances, including polybrominated diphenyl ethers (PBDEs; flame retardants), and roughly one-third (180,000 acres) of submerged lands within Puget Sound are considered moderately contaminated. Primary pollutants of concern in Puget Sound include heavy metals, organic compounds, PAHs, PCBs, dioxins, furans, DDT, phthalates, and PBDEs. Areas of highest concern in Puget Sound are Southern Hood Canal, Budd Inlet, Penn Cove, Commencement Bay, Elliott Bay, Possession Sound, Saratoga Passage, and Sinclair Inlet (PSAT 2007). Hypoxic or low dissolved oxygen concentration were found at a number of monitoring stations, including Saratoga Passage, Discovery Bay, Bellingham Bay, Elliott Bay, Budd Inlet, and Commencement Bay. Many of the contaminants in the Sound, including several that were banned years ago, continue to bioaccumulate in the food web to top level predators (NMFS 2008a).

Hydromodification Projects. More than 20 dams occur within the region’s rivers and overlap with the distribution of salmonids, and a number of basins contain water withdrawal projects or small impoundments that can impede migrating salmon. The impact of these and land use changes (forest cover loss and impervious surface increases) has been a significant modification in the seasonal flow patterns of area rivers and streams, and the volume and quality of water delivered to Puget Sound waters. Several rivers have been hydromodified by other means including levees and revetments, bank hardening for erosion control, and agriculture uses. Since the first dike on the Skagit River delta was built in 1863 for agricultural development (Ruckelshaus and McClure 2007), other basins like the Snohomish River are diked and have active drainage systems to drain water after high flows that top the dikes. Dams were also built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

In the next couple of years, a highly publicized and long discussed dam removal project is expected to begin in the Elwha River. The removal of two dams in the Elwha River, a short but formerly very productive salmon river, is expected to open up more than 70 miles of high quality salmon habitat (Wunderlich et al. 1994). Estimates suggest that nearly 400,000 salmon could begin using the basin within 30 years after the dams are removed (PSAT 2007).

About 800 miles of Puget Sound's shorelines are hardened or dredged (PSAT 2004). The area most intensely modified is the urban corridor – the eastern shores of Puget Sound from Mukilteo to Tacoma). Here, nearly 80% has been altered, mostly from shoreline armoring associated with the Burlington Northern Railroad tracks (Ruckelshaus and McClure 2007). Levee development within the rivers and their deltas has isolated significant portions of former floodplain habitat that was historically used by salmon and trout during rising flood waters.

Mining. Mining has a long history in the Washington, and in 2004 the state was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano et al. 1993; NMA 2007). Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 (Palmisano et al. 1993). Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Commercial and Recreational Fishing. Most of the commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by tribal fisheries and by charter and recreational anglers. Nets and trolling are used in commercial and tribal fisheries, whereas recreational anglers typically use hook and line, and may fish from boat, river bank, or docks. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Oregon-Washington-Northern California Coastal Drainages

Natural History

This region encompasses drainages originating in the Klamath Mountains, the Oregon Coast Mountains and the Olympic Mountains, all of which form the Coast Range ecoregion where elevations range from sea level to about 4,000 feet. More than 15 watersheds drain the region's steep slopes including the Umpqua, Alsea, Yaquima, Nehalem, Chehalis, Quillayute, Queets, and Hoh rivers. Numerous other small to moderately sized streams dot the coastline. Many of the basins in this region are relatively small; the Umpqua River drains a basin of 4,685 square miles and is a slightly over 110 miles long, and the Nehalem River drains a basin of 855 square miles and is almost 120 miles long. However, systems here represent some of the most biologically diverse basins in the Pacific Northwest (Johnson 1999; Carter and Resh 2005).

The region is part of a coastal, temperate rainforest system, and is characterized by a moderate maritime climate marked by long wet seasons with short dry seasons and mild to cool year-round temperatures. Average annual precipitation ranges from about 60 inches to more than 180 inches, much of which falls as rain, and supports a rich temperate forest. Vegetation is characterized by giant coniferous forests of Sitka spruce, western hemlock, Douglas fir, western red cedar, red alder, and black cottonwood

The Oregon Coast supports a unique coastal sand dune system. The sand dunes were largely created by the sand deposited from the coastal rivers, in particular the Umpqua and Columbia rivers. North, steep headlands and cliffs are separated by stretches of flat coastal plain and large estuaries. Significant estuaries in the region (outside of the Columbia River Estuary) include Coos Bay, Tillamook Bay, and the Nehalem River Estuary in Oregon, as well as Grays Harbor and Willapa Bay in Washington.

Human Activities and Their Impacts

Land Use. The rugged topography of the western Olympic Peninsula and the Oregon Coastal Range has limited the development of dense population centers. For instance, the Nehalem River and the Umpqua River basins consist of less than 1% urban land uses. Most basins in this region have long been exploited for timber production, and are still dominated by forestlands. In Washington State, roughly 90% of the coastal region is forested (Palmisano et al. 1993). Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural (Johnson 1999). Similarly, in the Umpqua River basin, about 86% is forested land, 5% agriculture, and 0.5% are considered urban lands. Roughly half the basin is under Federal management (Carter and Resh 2005).

Hydromodification Projects. Compared to other areas in the greater Northwest Region, the coastal region has fewer dams and several rivers remain free flowing (e.g., Clearwater River). The Umpqua River is fragmented by 64 dams, the fewest number of dams on any large river basin in Oregon (Carter and Resh 2005). According to Palmisano et al. (1993) dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat. In the past, temporary splash dams were constructed throughout the region to transport logs out of mountainous reaches. The general practice involved building a temporary dam in the creek adjacent to the area being logged, the pond was filled with logs and when the dam broke the floodwater would carry the logs to downstream reaches where they could be rafted and moved to market or downstream mills. Thousands of splash dams were constructed across the Northwest in the late 1800s and early 1900s. While the dams typically only temporarily blocked salmon habitat, in some cases they remained long enough to wipe out entire runs, since effects of the channel scouring and loss of channel complexity resulted in the long term loss of salmon habitat (NRC 1996).

Mining. Oregon is ranked 35th nationally in total nonfuel mineral production value in 2004, while Washington was ranked 13th nationally in total non-fuel mineral production value and 17th in coal production (Palmisano et al. 1993; NMA 2007). Metal mining for all metals (e.g., zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 (Palmisano et al. 1993). Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined in both Washington and Oregon. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Commercial and Recreational Fishing. Most commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by tribal fisheries, as well as by charter, and recreational anglers. Nets and trolling are used in commercial and tribal fisheries, whereas recreational anglers typically use hook and line, and may fish from boat, river bank, or docks.

Impact of the Environmental Baseline on Listed Resources

In 2007, the population of the United States increased to more than 300 million people for the first time in its history. That population growth and increase in population density was accompanied by dramatic changes in the landscapes of the United States. By 2000, half of the population in the United States lived in the suburbs (Hobbs and Stoops 2002). About 75% of all Americans now live in areas that are urban or suburban in character; that is, about 75% of the people in the lower 48 States live in less than 2% of the land area of the lower 48 states. Most modern metropolitan areas encompass a mosaic of different land covers and uses (Hart 1991). The mosaic or land uses associated with urban and suburban centers has been cited as the primary cause of declining environmental conditions in the United States (Flather et al. 1998) and other areas of the world (Houghton 1994).

The direct and indirect effects of these changes in land-use and land-cover have had lasting effects on the quantity, quality, and distribution of every major terrestrial, aquatic, and coastal ecosystem in the United States, its territories, and possessions. Many native ecosystems exist as small isolated fragments, surrounded by expanses of urban and suburban landscapes or by natural areas dominated by non-native species. As a result, many of the native plant and animal species that inhabited those native ecosystems over the past have become extinct, endangered, or threatened over the past 200 years. Even marine ecosystems, once deemed by many as the most resilient of ecosystems, a vast source of fish for harvest and a limitless sink for waste material, are threatened by human activities on a global scale. The most pervasive threats to marine ecosystems include ocean-based destructive demersal fishing practices, increasing sea temperatures, coastal development, increased sediment loading, point-source organic pollution, and hypoxia (Halpern et al. 2007).

The rapid growth of commercial fishing of what was once considered an endless food supply has resulted in drastic over-exploitation of fisheries resources and modification of the marine environment (Hall 1999). Increases in national and global populations have lead to a dramatic increase in demand for seafood, resulting in expansion of fishing fleets by orders of magnitude, development of new technology to capture resources more efficiently, and greater ability to exploit areas once considered out of reach. In particular, fishing practices have lead to pressures not only on target species, but changes to whole habitats and the protected species that are either caught directly, or whose habitat is degraded because of them. It has been estimated that global commercial fishing industries catch and discard 27 million metric tons of fish, sea turtles, marine mammals, and other organisms annually (Hall 1999). Gill nets set for several miles can entrap, drown, or disable any organisms larger than their mesh size, from salmon to large whales. Although gill nets may be set thousands of miles from domestic waters, individuals of protected species caught in these nets can be the same that nest, breed, or feed in United States waters.

Dredging and trawling gears clear bottom habitat of any sizeable material, eliminating habitat of small fishes and invertebrates on which other species feed (Hall 1999). This process also displaces large amounts of sediment into the water, dramatically altering water clarity and chemistry. There are likely additional factors that influence listed species directly or indirectly, which are thus far unknown.

The process of global warming is a developing concern to protected species management. Widespread habitat alteration or loss can also stem from even moderate, but prolonged, increases in temperature. Although many effects of climate change are unknown, the instability and environmental change that has been measured to occur thus far support the likelihood that global warming will have negative impacts on protected species and the habitats that they occupy.

Coastal development has more localized effects on marine environments, but is so extensive that most, if not all, nearshore environments are affected by it in some way. Development may be so detrimental as to extirpate populations or species in very short periods of time. Such is the case with several populations of salmon along the United States Pacific coast, where dam construction blocked fish movement to and from spawning and feeding habitats (Lichatowich 1999). As a result, entire populations are now considered extinct. In general, coastal development without environmental consideration has resulted in direct mortality to protected species, modification of habitat to displace individuals or populations from a region, or reduced reproductive success. In such cases, survivorship declines can be significant, resulting in protection of species not formerly listed, or moderate in species already listed that can ill-afford further impediments to recovery. As with fishing, coastal development in foreign countries can affect marine species protected in the United States by affecting habitat to which these species migrate for breeding or feeding. Environmental impacts, particularly to strategically important or listed species, of coastal development have received more global interest in recent years and changes, such as EIS statements, outreach and education, and environmentally friendly design have mitigated some impacts. However, many countries continue developing coastal regions without significant concern for protected or sensitive species or their habitats and these distant activities can have negative consequences for listed species in this country.

Additional activities on land have significant effects in ocean environments. This is particularly true for sedimentation as well as agricultural, industrial, and municipal pollution. Soils are normally covered by tracts of forest, grassland, marsh, or other vegetation preventing significant erosion. However, development activities tend to disturb these areas, or bring in large amounts of soil during construction, allowing for wind, rain, and other mechanisms to move the soil to local water bodies. Salmon nests become covered with sediment, or highly localized spaces for nests become covered, resulting in high hatching mortality or elimination of entire stretches of spawning habitat (NMFS and USFWS 2005).

Agricultural development and use has its own unique contribution to marine pollution. Fertilizers applied to tracts of land, from front lawns to large fields, can run-off in rainwater if not applied properly these fertilizers contain concentrated nutrients that dissolve in water and enter streams, rivers, lakes, estuaries, and the marine environment (Kennish 1992; Soares 1999). Along with nutrients contained in sediments, these elevated nutrient concentrations provide fodder for potentially exponential bacterial, algal, and plant growth. This rapid growth process can create algal "blooms" (red tide), which can make toxic metabolic byproducts in such concentrations that fish, seabirds, and marine mammals can become ill or die as a result. Such events happen continually along Gulf of Mexico states and instances are known for the west and east coasts. After nutrients have been used up, large numbers of small organisms die and the natural breakdown of their bodies results in areas of oxygen depleted water, sometimes hundreds of square miles in size, called "dead zones" in which organisms requiring oxygen in water to breathe cannot survive. Such an area occurs off the coast of Louisiana. This process of eutrophication can eliminate large areas of nearshore and oceanic habitat, resulting in direct mortality to or adverse modification of habitat utilized by listed species. Shortnose sturgeon are generally believed to be absent from numerous rivers feeding into and sections of the Chesapeake Bay itself because of eutrophication issues stemming from fertilizer use on lawns and fields. Unlike most other forms of pollution, eutrophication can eliminate or displace large sections of habitat and all animals within it. This issue has received more interest in recent years. Regulations are being installed to regulate fertilizer runoff and public outreach has been growing.

Although sedimentation and agricultural pollution comes from general areas, point-source pollution comes from specific effluents and can have additional effects. These drainages frequently come from municipal wastewater treatment plants, commercial and industrial discharges, as well as recreational and commercial vessels (Kennish 1992). Point-sources tend to contain specific chemical components that result from anthropogenic activity, as opposed to excessive sediments entering a waterbody. These components can be toxic and require regulation. However, the effects of components on species and their environment is generally unknown and it is only after several years of research that enough evidence is collected to initiate regulation. Such has been the case with pesticides, such as DDT and DDE, which caused severe fragility in bird eggs and led to the listing of several avian species, including bald eagles. Such is now the case with pharmaceuticals in wastewater. Hormones are currently released in wastewater from treatment plants. It is unknown what effects these chemicals have on endocrine disruption to species in habitats near wastewater discharges. It has been suggested that humans reconsuming these waters may have intra-sex children (Soares 1999), which indicates that these chemicals may affect other exposed organisms. What is known is that point-source discharges can introduce chemicals into fresh water, estuarine, and marine habitats whose effects can cause significant decline in a variety species, but the effects may not be known for years later.

Salmonids originally underwent dual pressures that led to their decline: dam construction and commercial fishing (Lichatowich 1999). Although fishing had occurred extensively through time, more widespread and technologically advanced methods were developed in the past two centuries to harvest salmon beyond the rate at which they could reproduce (Lichatowich 1999). More importantly and at the same time, dam construction occurred that cut the connection between two necessary salmon habitats: streams and ocean (Lichatowich 1999). This lead almost immediately to large-scale salmon declines or extinctions in several local areas. Now, dams have generally been modified or removed to re-establish communication between habitats for salmon in most areas. Commercial fishing is also closely monitored to prevent excessive pressures on populations. However, new threats in the forms of habitat loss, pollution, and genetic dilution of populations specialized for certain habitats impede recovery efforts (Reisenbichler 1997). As predators, salmon tend to bioaccumulate toxins as whales do, but generally accumulate more because they eat other fish instead of krill, which are lower on the food chain. Pollution is identified as a contributing factor for 38% of ESA listed species overall (Hoffman et al. 2003). Contaminants can cause reproductive disruption, immune dysfunction, and other physiological effects accumulate in vertebrates and can cause reduced reproductive fitness and subsequent population decline (Rand and Petrocelli 1985).

EFFECTS of the Action

The *Description of the Proposed Action* describes EPA's proposal to continue to recommend the 1985 304(a) aquatic life criteria for cyanide and approve state and tribal water quality standards, or federal water quality standards promulgated by EPA for the protection of aquatic life that are identical to or are more stringent than the section 304(a) cyanide aquatic life criteria. The *Status of the Species and Critical Habitat* section of this Opinion identified the endangered and threatened species, and designated critical habitat that may be affected by the proposed action, as well as those species and critical habitat that are currently proposed for listing under the ESA. The *Status* also summarized the status and trends of those species, and other ecological information relevant to our effect's analyses, while the *Environmental Baseline* summarized the consequences of a variety of human activities, including land and water uses that impact the listed species and critical habitat considered herein.

In this section, we identify specific stressors and subsidies associated with the proposed action, the likelihood endangered species, threatened species and designated critical habitat are exposed to those stressors and subsidies, the responses of listed species and critical habitat to their exposure, and the consequences of those responses to the different listed resources. Based on the results of these analyses, we assess the risks EPA's proposal to recommend and approve of water quality standards for cyanide poses to listed resources. For endangered and threatened species, our assessment focuses on the risk of increasing the extinction probability of these species, for designated critical habitat our assessment focuses on the risk of reducing the conservation value of the habitat designated for the endangered and threatened species.

As discussed in the *Approach to the Assessment* section of this Opinion, as part of this national consultation, our consultation examines the decision-making process that EPA uses to recommend and approve water quality criteria and the outcome of that decision making process. In particular, this consultation focuses on how EPA determines what constitutes a "safe and healthful level in waterbodies for a pollutant, which a regulatory authority can use to guide the control, reduction and eventual elimination of that pollutant (BE, page 15)" in the environment for the protection of fish and wildlife species consistent with the goals of the CWA, and threatened and endangered species in particular.

When EPA recommends 304(a) aquatic life criteria, that recommendation means that water quality standards identical to or more stringent than EPA's criteria will protect the designated uses of water that receive pollutants at levels consistent with the aquatic life criteria. As a default, EPA uses "fishable and swimmable" as the designated uses when it establishes their aquatic life criteria. That is, EPA has determined that the adoption of their criteria to be protective of aquatic life designated uses consistent with the objective and goals articulated in

CWA sections 101(a) and 101(a)(2) (EPA 2008b). Therefore, when EPA recommended 304(a) aquatic life criteria for cyanide, EPA also determined that cyanide at the recommended numeric value would protect designated uses consistent with the objective of the CWA to "restore and maintain the chemical, physical and biological integrity of the Nation's waters (CWA §101(a))," and the goal to provide "for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water…. (CWA §101(a)(2))."

If EPA recommends 304(a) aquatic life criteria, then fish and wildlife that might be exposed to pollutants at those criteria levels generally should not experience physical, physiological, behavioral, or ecolgocal consequences that would interfere with reproduction or reduce the long-term persistence of their populations resulting from that exposure. That is, EPA expects that the criteria would generally provide a "reasonable level of protection" of all but a small fraction of the "appropriate" taxa (0.05; Stephan et al 1985). Restated, there is 5% probability that an aquatic species would not be protected by EPA's national criteria. Provided EPA considers threatened and endangered species part of the taxa that would be protected by their national criteria, then we would expect that EPA's national criteria would generally protect endangered and threatened species and designated critical habitat. Specifically, we would expect that when EPA recommends the CMC of 22.36 μg/L and the CCC of 5.221 μg/L in fresh water, or at the CMC of 1.015 μg/L or the CCC of 1.015 μg/L in salt water as its 304(a) aquatic life criteria for cyanide, then endangered or threatened species or designated critical habitat exposed to cyanide at these concentrations should not experience physical, physiological, behavioral, or ecological consequences that would reduce the long-term persistence of their populations resulting from that exposure. Certainly this would be the case if (a) EPA considered aquatic listed species as an indicator of or part of the aquatic assemblage that defines the biological integrity of the Nation's waters, or part of the fish, shellfish and wildlife the CWA intends to protect; or (b) listed species are expressly or indirectly listed as a designated use by a state or tribe.

We begin our assessment of the *Effects of the Action* by evaluating the decision making process EPA uses to develop 304(a) aquatic life criteria and establish numeric values for the CMC and CCC for a particular pollutant, and EPA's 1985 published values for the cyanide CMC and CCC. These values represent EPA's recommended 304(a) aquatic life criteria for cyanide, upon which EPA intends to subsequently approve for use in state or tribal water quality standards. Our evaluation focuses on whether it is reasonable to expect that endangered species, threatened species, and designated critical habitat are exposed to cyanide at concentrations similar to national criteria values; and whether it is reasonable to expect that endangered species, threatened species, and designated critical habitat are not likely to respond to any exposures to cyanide at the CMC of 22.36 μg/L or the CCC of 5.221 μg/L in fresh water, or at the CMC of 1.015 μg/L or the CCC of 1.015 μg/L in salt water.

If listed resources are likely to respond to exposures to cyanide at the CMC of 22.36 μg/L or the CCC of 5.221 μg/L in fresh water, or at the CMC of 1.015 μg/L or the CCC of 1.015 μg/L in salt water, then we would evaluate the likelihood that endangered species, threatened species, and designated critical habitat would be exposed to: a) the one-hour average exposure concentrations of cyanide that would not exceed the CMC more than once every three years; and b) four-day average exposure concentrations of cyanide that would not exceed the CCC more frequently than once every three years on average. If we conclude that, endangered species, threatened species, and designated critical habitat would be exposed to cyanide at concentrations that deviate from the one-hour and four-day average, we would examine the variability in concentrations to which endangered species, threatened species, and designated critical habitat would be exposed. As part of this evaluation, we would examine whether endangered species, threatened species, and designated critical habitat "should not be affected unacceptably (EPA 1985)" if the four-day average concentration of cyanide does not exceed 5.2 μg/CN in fresh water or 1.015 μg/CN in salt water more than once every three years on average and if the one-hour average concentration does not exceed 22.36 μg/CN in fresh water or 1.015 μg/CN in salt water more than once every three years on average. Finally, we would evaluate the context for probable exposure events including whether environmental conditions in which listed species reside or the physiological state of the individual organism would influence the severity of probable responses.

EPA's Decision-Making Process

Derivation of Criteria

In order to evaluate whether the cyanide aquatic life criteria and any water quality standards that would be based on those criteria are not likely to jeopardize listed species or adversely modify critical habitat, we first examine how EPA derived the aquatic life criteria. The EPA document, *Guidelines for Deriving Numerical National Water Quality Criteria for* the *Protection* of *Aquatic Organisms and Their Uses* (the "Guidelines") outlines the process that EPA uses to derive water quality recommendations that intend to protect aquatic assemblages (Stephan et al. 1985; EPA 2008b). According to the guidelines, once a decision is made that a criterion is needed EPA collects and reviews all available information on the toxicity of the chemical is collected and reviewed for acceptability, and sorted.

The decision-making process for deriving aquatic life criteria involves a mix of quantified estimates of the effects a particular contaminant would have on a sample of test subjects, and professional judgement. That is, criterion development involves quantifying the sensitivity of a suite of species to toxic compounds in controlled studies; professional judgment comes into the process in several areas including the setting aside of data, determining whether a species is commercially or recreationally important and whether data on that species deserves additional attention in the final derivation of the criterion, determining whether particular data is useful or should be set aside (e.g., determining if water quality characteristics of a test are acceptable, or whether the degree of agreement between species is reasonable).

As a general matter, the *Guidelines* require the use of acute and chronic toxicity tests on a broad range of aquatic species to provide an indication of the sensitivities of untested species. These data are used by EPA to develop chronic and acute criteria for both salt and fresh water, the CCC and the CMC respectively. EPA's development of two values for fresh and salt water, the CMC and CCC, is premised on the assumption that doing so more accurately reflects toxicological and practical realities while not being as restrictive as a one-number criterion would have to be in order provide the same degree of protection (Stephan et al. 1985).

To derive an acute criterion for fresh water, the *Guidelines* suggest that toxicity data be available for at least one species of freshwater animal in at least eight different families. The families include:

1) Salmonidae (e.g., salmon or trout),
2) a second family in the class Osteichthyes, preferably a commercially or recreationally important warmwater species (e.g., bass, bluegill),
3) a third family in the phylum Chordata (e.g, salamander, frog),
4) a planktonic crustacean (e.g, daphnia),
5) a benthic crustacean (e.g, crayfish, amphipod),
6) an insect (e.g., dragonfly, mayfly),
7) a family in a phylum other than Arthropoda or Chordata (e.g., mussel, snail, worm), and
8) a family in any order of insect or any phylum not already represented.

For deriving a saltwater acute criterion the *Guidelines* suggest that acute tests with at least one species of saltwater animal in at least eight different families should be used. The represented families should include:

1) two families in the phylum Chordata,
2) a family other than Arthropoda or Chordata,
3) either the Mysidae or Penaeidae family,
4) three other families not in the phylum Chordata,
5) and any other family.

Additionally, at least one acceptable test is required for saltwater and freshwater plants, and at least one acceptable bioconcentration factor determined with an appropriate saltwater species. Data that is rejected from further consideration may include: data from studies that did not contain control treatment; too many organisms in the control treatment died or showed signs of stress or disease; data from tests using distilled or deionized water as the dilution water without adding appropriate salts; data from species that do not have reproducing wild populations in North America; data on organisms that were previously exposed to substantial concentrations of the test material or other contaminants (Stephan et al. 1985).

Studies used for determining the CMC are acute tests, which are performed with 48 or 96 hours of exposure, and measure the concentration at which the toxin causes death in 50% of the test population (LC_{50}). The LC_{50} values for each species are pooled and averaged to determine the species mean acute value (SMAV). If EPA has data on several species within a genus, then the data are pooled again to calculate a genus mean acute value (GMAV). If data are available from only one species, then that species mean value becomes the GMAV. Once calculated, the GMAV is ranked from high to low (least to most sensitive species) and the lowest four values are used in regression to estimate the concentration that would cause death for the fifth percentile of the most sensitive species. This fifth percentile value represents the final acute value or FAV. In the event a commercially or recreationally important species has a SMAV below the FAV, the SMAV can be substituted for the FAV to protect that important species. Once EPA has determined the FAV (or the lowest SMAV for an important species) then that value is divided by two, in an effort to avoid the death of exposed organisms. The resulting value is the criterion maximum concentration or CMC. EPA calculates the CMC under the assumption, that the CMC averaging period would be substantially less than the lengths of the acute tests upon which it is based (Stephan et al. 1985). As such, EPA recommends that the CMC be applied as a limit on the 1-hour average concentration in the environment to provide an addition level of protection.

Chronic toxicity values are calculated either in the same general manner as the acute values, or by dividing the FAV by the final acute-to-chronic (ACR). The ACR is a way of relating the acute toxicities to chronic toxicities and is more commonly employed because it allows EPA to make use of a smaller data set. Chronic toxicity test data must be available from at least three different families, so long as at least one is a fish, an invertebrate, and one is an acutely sensitive species, in order to derive a final chronic value. In contrast to acute studies, chronic tests may last weeks or more, at sublethal exposure concentrations and focus on the endpoints of growth and reproduction. Chronic studies focus on two levels of effect for a concentration: the NOEC and the LOEC that cause a statistically significant change in the endpoint of interest (growth or reproduction). Similar to the FAV, the CCC is derived by pooling values and calculating the geometric mean of the two effect levels.

EPA’s decision-making process was developed under the assumptions that:

1) Effects that occur on a species in laboratory tests generally occur on the same species in comparable field situations (Stephan et al. 1985);
2) Effect levels defined by chronic toxicity tests are conducted on the “most sensitive life stages” and therefore should protect all other (less sensitive) life stages (Stephan et al. 1985)
3) When the minimum data requirements are satisfied, but few data are available, then restrictive criteria values are derived (BE 2006).
4) The averaging recommendation is based in part on the assumption that most bodies of water could tolerate exceedences once every three years on the average provided the body of water is not subject to anthropogenic stress other than the exceedences of concern (Stephan et al. 1985).

Important caveats to the general approach in EPA’s decision-making process include:

1) The development of water quality standards may need to take into account additional factors such as hydrological considerations, environmental and analytical chemistry, extrapolation from lab to field situations, and relationships between species for which data are available and species in the water of concern (Stephan et al. 1985),
2) It may be necessary to derive site-specific criteria by modifying national criteria to reflect local conditions of water quality, temperature, or ecologically important species.
3) Some untested locally important species might be very sensitive to the contaminant of concern (Stephan et al. 1985),
4) Some aquatic organisms in the wild may be stressed by disease, parasites, predators, other pollutants, contaminated or insufficient food, and fluctuating and extreme conditions of flow, water quality and temperatures (Stephan et al. 1985),
5) The decision-making approach is meant to derive criterion that prevent unacceptable long-term and short-term effects, which is not the same as threshold of adverse effects. Some adverse effects (e.g., small reductions in growth, survival or reproduction) will probably occur at or below criterion values (Stephan et al. 1985),
6) The frequency, magnitude and duration of the exceedences should be based on the ability

of the aquatic ecosystem to recover, which will differ greatly according to the pollutant and the state or health of the ecosystem (Stephan et al. 1985)

Understanding the assumptions and the caveats inherent to EPA's decision-making process is important to understanding the uncertainty around the values EPA recommends to states and tribes for use as their water quality standards. For instance, according to EPA laboratory tests conducted at constant exposures simulates "worst case" field conditions. In limited circumstances this assertion is probably true, but in many cases it is not. In the wild, species will typically not be exposed to continuous concentrations of a particular chemical. Rather, concentrations typically vary temporally and spatially and would result in doses that are both higher and lower than the tested dose. This in itself does not make the laboratory exposure approach a reasonable simulation of a worst-case field (or natural) condition. Responses of organisms tested in controlled laboratory systems do not necessarily provide reasonable predictors of organisms' responses to similar chemicals in the wild, although admittedly in many cases this is the only type of data available to us from which to conduct an evaluation. In many cases, the conditions simulated in a laboratory test have little to do with the environment in which most species live in the wild, and as such are unlikely to resemble "worst case field conditions."

In laboratory tests, species are generally isolated from multiple stressors so that researchers are able to isolate the species responses to the chemical (or stressor) under study. In the wild, species are typically exposed to a wide range of stressors, from natural to human induced. For instance, lab studies do not replicate typical environmental conditions where intraspecific competition for food or shelter occurs. Instead, all the test organisms are about the same size, provided with abundant food, and minimal habitat complexity. Interspecific competition generally does not occur in lab tests either, as most lab environments isolate the species under study from typical predators. Physical conditions are maintained at optimal or constant levels (e.g., velocities, water temperature, and dissolved oxygen are not representative of fluctuating conditions in a natural aquatic environment) and generally, there are no other chemical stressors present. Typically, lab specimens are generally not exposed to other stressors such other chemicals, or environmental factors that can influence toxicity (e.g., some chemical or environmental changes in temperature or other parameters can increase or decrease toxicity, some times in a greater than additive fashion). Wild taxa are exposed to a myriad of factors that can influence their responses to a particular chemical at a particular concentration, and at best the laboratory tests are an indication of how species may respond to that chemical in nature. The actual physical and chemical conditions within a waterbody can, for some chemicals, alter the toxicity of the chemical evaluated in the laboratory under controlled conditions. Knowing this, the authors of EPA's decision-making process noted that it may be necessary to account for local conditions when setting water quality standards and permit limitations (see caveat 1 above).

Another important assertion is that EPA's decision-making process uses the most sensitive life stages for defining chronic toxicity. Unfortunately, chronic values, as is the case of acute values, are calculated on available data and generally, chronic studies are few in comparison to studies that examine mortality as the endpoint of concern. The species used for lab tests are also often not representative of the composition and sensitivities of species in a natural community or ecosystem. EPA's aquatic life criteria guidelines require species from eight different families be tested to determine acute toxicity values for both marine and fresh water. To derive chronic numeric criteria, however, only three chronic tests are necessary, despite the fact that chemical concentrations in the natural environment are likely to occur more often at chronic low levels. Use of such a small data set to make inferences to a much larger community in the wild is cause for concern. Further, it is unclear whether the assumption that the most sensitive life stage is tested, is regularly met. Certain life stages or the transition between life stages, which are inherently stressful as a result of the physiological changes the animal is undergoing (e.g., osmoregulation), are rarely tested.

Even if the tests are conducted on the most sensitive life stage to a particular toxicant, it does not necessarily follow that the critical concentration determined by these sensitive stages are correlated with the vulnerability of the species to the toxicant. For instance, Kammenga et al. 1996 demonstrated that the fitness implication of a toxicant was measureable on the least sensitive stage of the tested species, whereas the most sensitive trait did not have any effect on the fitness of organism. Equally important, however, is that the smaller the data set used to extrapolate responses, the lower the confidence can be in the outcome of the final value. As is the case for many compounds, for cyanide the most robust data set that EPA had available for making its decision was the acute data set. Only five studies were used for deriving the CCC for fresh water, and two from saltwater taxa ---none of which represent empirical evidence of how any of the species addressed in this Opinion would respond to low-level or prolonged exposures of cyanide.

According to EPA when the minimum data requirements are satisfied, however, then *restrictive* criteria values are derived. Unfortunately, extrapolating the stress responses of individuals in a limited number of lab tests to organisms exposed to similar chemical concentrations while in highly complex natural environment provides for weak gross scale predictions at best, particularly when few or none of the species of interest were evaluated by direct empirical evidence. The greatest utility in simple laboratory tests is that they facilitate faster (and cheaper) data on generalized responses of a range of taxa to a defined chemical exposure (Cairns and Niederlehner 2003). Models, mesocosm or field studies, transparent reasoning, and validation studies should temper the results of such lab studies in decision-making, particularly when extrapolating potential environmental outcomes to a complex environment and in situations, like the management of threatened and endangered species and their designated critical habitat, where a low tolerance for error is warranted. Stephan (2002) summed it up best when he said: “Unless species are selected from a field population using an appropriate procedure (e.g., using random or stratified random sampling), use of the resulting benchmark(s) to protect field populations requires a leap of faith that the distribution of the sensitivities of tested species is representative of the distribution of the sensitivities of field species.”

Consideration of Listed Resources in EPA’s Decision-Making Process

EPA’s decision-making process (a.k.a. the *Guidelines*) does not explicitly require EPA to consider toxicity data on endangered or threatened species, although one species in particular, *Oncorhynchus mykiss* (specifically the freshwater phenotype, rainbow trout) is a commonly tested fish species. How EPA incorporates threatened and endangered species into their approval of state and tribal water quality standards varies across regions. Certain regional offices of EPA have completed Section 7 consultations on their approval of state water quality standards for a subset of the numeric standards. This national consultation represents the first of a series of Section 7 consultations with EPA on their recommended criteria and EPA's subsequent approval of state and tribal water quality standards that are based on the recommended 304(a) aquatic life criteria. This enhanced coordination at the national level was envisioned under an MOA between the Services and EPA (66 FR 11202).

There is a critical difference between decision-making for the purpose of criteria setting and conducting a risk assessment on a particular species or group of species (Suter and Cormier 2008). The benchmark calculation used in EPA's decision to recommend a particular criterion "rests on an assumption that selecting a percentile [e.g. 95%] is an appropriate way of specifying a level of protection (Stephan 2002)." Whereas, the Section 7 consultation solves (or attempts to solve) the risk of exposing listed species to a particular federal action or set of actions, in this case the risk of exposing listed species to chemicals at particular concentrations. Unlike criteria development, Section 7 consultations begin by assessing the effect of the chemical to the individual of a listed species. This endpoint differs greatly from the population level response evaluated during criteria development.

To bridge the gap between the aquatic-life criteria decision-making process and information needed to conduct Section 7 consultation, EPA with the assistance of the Services, developed the *Draft Framework for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual*. The Methods Manual describes a process for evaluating whether the CMC protects acute mortality of listed species, and whether the CCC protects listed species under longer exposures. Additionally the Method introduces a process for evaluating the effects expected from a diet of aquatic organism contaminated with the chemical of interest to levels that would result from concentrations consistent with the criterion. The Method also addresses toxicity of the criterion chemical to the food items of listed species to determine if listed species are likely to be adversely affected by a loss of food. The basic goal of the Methods Manual was to produce robust decisions for determining when the aquatic life criteria for a specific chemical is likely to adversely affect (or not) a particular listed species, and whether formal consultation is required.

The Methods Manual Approach to Estimating Acute Responses. To evaluate whether a listed or proposed species would respond to a particular chemical when exposed at the criterion value, the Methods Manual uses a risk paradigm or risk ratio for conducting toxicity screening that is based on the numeric value represented by the CMC[9] as the "assessment exposure concentration" (represented by C_A), divided by the "assessment effects concentration" (EC_A).

$$R = C_A/EC_A$$

The EC_A is an estimate of the highest chemical concentration that EPA portends would cause an acceptable small adverse effect and for acute effects that estimate is derived when the mean acute

[9] The BE and the *Methods Manual* refer to this as the "maximum exposure concentrations allowed by the criteria", but this is deceptive as the approved standard allows that discharges may exceed the established value for the CMC under certain circumstances. See our discussion under *Concentrations of Cyanide in U.S. Waters*. Therefore, we note here that the C_A = CMC (or the CCC).

value divided by 2.27[10]. For acute toxicity, the small level of effect is EC_0 to EC_{10}. Under this simple paradigm when $C_A < EC_A$ then the chemical concentration established by the aquatic life criteria "is not likely to adversely affect" listed species. Conversely, when $C_A \geq EC_A$ then the chemical concentration established by the aquatic life criteria is considered "likely to adversely affect" listed species (see *Methods Manual*, page 9). This risk paradigm, defined by the risk ratio, forms the foundation of the each aquatic life criteria consultation.

For listed species for which acute data exist, the relationship is straightforward. Using the mean acute value calculated for rainbow trout or steelhead exposed to cyanide we illustrate the calculation. For example,

If the steelhead mean acute value = 44.73 µg CN/L,

Then $EC_A = 44.73/2.27 = 19.70$

And, $R = 22.36/19.70 = 1.14$[11]

Under this framework, a species with an $R < 1$ is not likely to suffer lethal consequences when exposed at the CMC, and a species with an $R \geq 1$ is more likely to suffer lethal consequences when exposed to the pollutant of concern at the CMC. Using this framework, the farther the species' R-value is away from 1, the more confidence there is in the determination that the species is (or is not) protected when exposed to cyanide at the CMC.

For listed and proposed species without species-specific data the EC_A is calculated using data from surrogate species. Since we do not have species-specific data for most listed species; most of the assessments will likely either estimate LC_{50}s for species using the Interspecies Correlation Estimations (ICE) model or Species Sensitivity Distributions (SSD). EPA developed the ICE model using taxonomic level information for endangered species. ICE models are based on regression analyses of LC_{50}s measured for a listed species to LC_{50}s measured for the same chemicals for commonly used surrogate species, preferably based on a minimum of five test chemicals. If surrogate species have been tested for the chemical of interest, but the listed species of interest have not, the relationships are used to estimate the LC_{50} for the chemical and species of interest. When an ICE model is not available for a listed species, then an ICE model for the genus or family is used. In this instance, each higher order ICE model must contain at least two species that represent the genus or family for it to be useful. Due to the uncertainty in the correlations, EPA stated in the *Methods Manual* that they intended to estimate the LC_{50} using the lower 95% confidence bound of the ICE. On the other hand, the SSD is calculated from several surrogate species within the same taxonomic unit as the species of interest, to define possible LC_{50}s for the species of interest. According the *Methods Manual*, to increase the confidence in protecting listed species the 5th percentile in this distribution will be used, such that

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[10] Note that we previously refer to this divisor as 2. The actual factor is 2.27, the inverse of 0.44, and "is based on 219 acute toxicity tests which showed that the mean concentration lethal to 0-10 percent of the test population was 0.44 times the LC50 (43 FR 21506, 18 May 1978)." In practice, such as the 1985 CN criteria document, EPA uses 2, but in the *Methods Manual* and this consultation EPA chose to not round to the nearest whole number but to use the fractional component as it was originally published in the *Federal Register*.

[11] If the CMC and the EC_A had been calculated with the same divisor, either 2 or 2.27, then in this example the rainbow trout R value would equal 1 because the CMC for cyanide was set using the rainbow trout SMAV.

the actual toxicity for the listed species should be higher than the chemical concentration estimated. When an ICE model was available for the listed species, or within the genus of the species of interest, the ICE model was given preference over the SSD. The *Methods Manual* lists a six-step approach for deriving EC_A estimates using surrogate data given the data that are available for closely related surrogates.

Given the lack of empirical information on the effects of many toxics on listed and proposed species, the Services and EPA will have to estimate to the best of their ability the potential effect using information from other species. Clearly, the validity and robustness of this risk ratio approach as a conceptual framework depends upon the value calculated for the EC_A. That is, the strength of the value (or range of values) represented by the EC_A depends ultimately on the identification, assimilation, and interpretation of evidence (i.e., the use best available scientific and commercial data) used in its calculation, which we expect will for most consultation predominantly come from surrogate species.

The Methods Manual Approach to Estimating Chronic Responses. To evaluate whether a listed or proposed species would respond to a particular chronic exposure to a particular chemical, the Methods Manual uses the same risk paradigm as described previously. For chronic toxicity, we used the numeric value represented by the CCC as the as the "assessment exposure concentration" (represented by C_A), divided by the EC_A.

As with the acute EC_A, the chronic EC_A represents an estimate of the highest chemical concentration in water or food that would cause an acceptable small adverse effect. For chronic toxicity, the acceptably small level of effect is the NOEC. Studies on the chronic effects of cyanide on listed species are few, and the literature search conducted by EPA was for a wide variety of species that have been tested. For chronic toxicity, the EC_A is based on the acute toxicity to the listed species, and the acute to chronic ratio (ACR) of surrogate species. The ACR is calculated as follows:

ACR = SS LC_{50}/SS NOEC

Where: SS LC_{50} is the LC_{50} for the surrogate species

SS NOEC is the No Observable Effects Concentration for the surrogate species

EC_As are estimated using the following equation:

EC_A = LS LC_{50}/ ACR

Where: LS LC_{50} is the LC_{50} for the listed species

So for example, if the fathead minnow SS LC_{50} = 138 µg CN/L,
And, the NOEC = 13 µg CN/L
Then, ACR = 10.6

The listed species LC_{50} is then divided by the ACR to derive an EC_A, which is compared to the CCC. Using this framework, when the $C_A < EC_A$ then the chemical concentration established by the aquatic life criteria "is not likely to adversely affect" listed species. Conversely, when the $C_A \geq EC_A$ then the chemical concentration established by the aquatic life criteria is considered "likely to adversely affect" listed species (see the *Methods Manual*, page 9).

Once the analysis produces a $C_A \geq EC_A$ for a particular listed species and contaminant combination, the *Methods Manual* provides little insight on the next step in EPA's evaluation. The *Methods Manual* merely states that when a particular chemical combination is classified as "likely to adversely affect" a particular listed species, these will require additional consideration and analysis to determine "under what circumstances risks are unacceptable (*Methods Manual*, page 9*)*." Unfortunately, the *Methods Manual* does not clarify for the reader what type or extent of "additional consideration and analysis" is necessary in such circumstances, nor does it provide a definition of when risks would be considered unacceptable (or acceptable). In contrast, the implementing regulations for Section 7 consultation state that "Each Federal agency shall review its actions at the earliest possible time to determine whether any action *may affect* listed species or critical habitat. If such a determination is made, formal consultation is required… (emphasis added; 50 CFR 402.14)."

Neither the implementing regulations for Section 7 consultation nor the ESA use the terminology "unacceptable" as a qualifier to describe effects to listed species. Therefore, it is unclear what EPA intended by this statement in the *Methods Manual* in terms of their Section 7(a)(2) consultations. An obvious unacceptable effect under Section 7 would be when an agency's action is likely to jeopardize the continued existence of listed species, or destroy or adversely modify critical habitat. Arguably, a reduction in the fitness of an individual of a listed species may also be considered unacceptable. The term's use is not defined under the ESA.

EPA's *Guidelines* may provide some insight into what EPA considers an unacceptable effect (Stephan et al. 1985). According to the *Guidelines*, the "protection of aquatic organisms and their uses should be defined as the prevention of *unacceptable* long-term and short-term effects on (1) commercially, recreationally, and other important species and (2)(a) fish and benthic invertebrate assemblages in rivers and streams, and (b) fish, benethic invertebrate, and zooplankton assemblages in lakes, reservoirs, estuaries, and oceans (emphasis added; Stephan et al. 1985)." According to Stephan (1986) his use of the term "unacceptable" in EPA's *Guidelines* was intentional because it allows for flexibility in determining the level of protection that a waterbody might receive and recognizes that such decisions are based on value judgements. When the validity of a criterion derived for a particular body of water is "based on an operational definition of 'protection of aquatic organisms and their uses' that take into account the practicalities of field monitoring and the concerns of the public" as suggested by EPA's *Guidelines*, then what drives the decision as to what constitutes an unacceptable risk is the level of protection (or conversely, adverse effect) that a particular criterion would have on a particular state or tribe's *designated uses* for their waters. The designated uses assigned to a particular waterbody by a state or tribe are explicit value-statements of what a particular state or tribe wants to protect their water resources for.

It follows that an unacceptable risk under EPA's decision-making process is one that fails to protect the designated uses for a waterbody. This is also consistent with EPA's review and approval of state standards. If EPA's line of inquiry as established through the *Methods Manual* leads them to a "may affect" or more specifically, a "may affect, likely to adversely affect" and assuming that the only risk that would be considered unacceptable is if the critierion under review fails to protect designated uses, the question that remains is whether EPA generally considers endangered and threatened aquatic species (and aquatic dependent as defined by the *Methods Manual*) a designated use. If listed aquatic species, however, are not specifically identified as a designated use by a particular state or tribe, we would ask whether EPA, states, and tribes would generally protect listed aquatic species, as part of the broader definition to protect species that are defined as "important", part of the aquatic "assemblage", or "fish and wildlife." That is, would listed species fall into any of the categories identified by the *Guidelines*:

1. commercially, recreationally, and other important species, and
2. (a) fish and benthic invertebrate assemblages in rivers and streams, and
   (b) fish, benethic invertebrate, and zooplankton assemblages in lakes, reservoirs, estuaries, and oceans?

The third category, "fish and wildlife" comes not specifically from the guidelines but from the language adopted by many states to describe their designated uses. The answer to the question "does EPA consider threatened and endangered fish or benthic invertebrates part of any of these categories?" is critical to understanding EPA's decision-making process both pursuant to the *Guidelines* and the *Methods Manual*, and this consultation.

Designated Uses

The *Approach to the Assessment* section of this Opinion identified that EPA's approval of a state water quality standard involves more than merely establishing a numeric value for a particular chemical pollutant, but also requires a positive finding from EPA that a state has adopted uses that are consistent with the requirements of the CWA and that their proposed criteria protect those designated uses. Thus, state designated uses are an action interrelated to EPA's approval of any state standards that rely on EPA's recommended CMC and CCC values. When a state modifies EPA's criteria or proposes their own water quality criteria, then EPA must evaluate and find that the criteria protect a state's designated uses. When EPA recommends a criterion or promulgates a federal water quality standard, EPA states that it would generally find its criterion support the designated uses and the goals of the CWA: to restore and maintain the chemical, physical, and biological integrity of the Nation's water (objective of the CWA); and provide for the protection and propagation of fish, shellfish, and wildlife (the interim goal). Thus a state or tribe that has identified acceptable designated uses under the CWA can expect that if they adopt EPA's recommended water quality criteria that EPA would approve the standard. When EPA approves state or tribal water quality standards, that approval implies that those standards protect the designated uses of the state's waters when state waters are exposed to chemical pollutants at levels consistent with the criteria.

Whether a state's water quality standards actually protect the designated uses is unclear, and likely varies by circumstance (e.g., pollutant, state, and use). A designated use is a goal statement for a water body that reflects the social and political value of the water. Like numeric criteria, each state has discretion to set their own designated uses. As a minimal standard, the

CWA requires states adopt use designations consistent with the provisions of sections 101(a)(2) and 303(c)(2) of the CWA. Thus, a state must adopt uses that provide for the protection and propagation of fish, shellfish, and wildlife, and other uses such recreation, agriculture and industry. If a state designates a use that does not address the "fishable and swimmable" goal, the state must complete a use attainability analysis (UAA) that justifies why such uses are not feasible, and that the state is establishing the highest attainable use, instead. A state has the discretion to make their uses as restrictive or loose as they desire, as long as they meet the "fishable and swimmable" goals of the CWA. While the designated use is a qualitative value statement for a waterbody, a criterion represents a scientific determination as to whether a particular water body can, given an ambient concentration of a pollutant, can still support the designated use (Gaba 1983). However, the designated use, while written in qualitative form, should be as specific as possible so as to be measurable or have meaningful and measureable surrogate indicators of goal (designated use) attainment (NRC 2001). According to the Government Accountability Office (2002), many states recognized that the linkage between their designated uses and their ability to measure attainment (or failure to reach attainment) was missing and acknowledged that they needed evaluation criteria to determine whether designated uses are being protected that are measured by reasonably obtainable monitoring data.

Accordingly, part of the problem that GAO (2002) and National Research Council (NRC 2001) noted was that many states' designated uses may be overly broad. Many states designated uses were established in the 1970s when they had only 180 days to do so. Consequently, many states adopted the very general goal of the CWA to provide for the protection and propagation of fish and wildlife (GAO 2002). According to the NRC (2001) the problem with such broadly defined designated uses is that broader the use designation and the weaker the linkage between the use and any measurable indicator of attainment, the greater uncertainty and higher likelihood of error in subsequent determinations of use attainment. We found that many of the coastal states and states that contain listed species under NMFS jurisdiction have updated their designated uses in the past ten years (Designated Use Table -see Appendix B). Currently, designated uses include such uses as fishing/harvest, propagation of fish, protection, natural state, viable populations, diversity, species richness, and species assemblages. In our review we found only a few specified that the use was for a native fish community, and a few that did not appear to have a designated use that included wildlife. We were also curious whether listed aquatic species are directly or indirectly protected as part of the designated uses coastal states had adopted.

We found only one state, California, and one territory, Puerto Rico that explicitly addressed threatened or endangered species as part of their designated use. California's designated uses include a broad statement that the waters must support the survival and maintenance of aquatic species that are protected, and Puerto Rico's designated uses note that endangered and threatened species are included as part of the broader category of desirable species (Table 34). Other states have revised their designated uses to incorporate the specific needs of certain threatened or endangered species (e.g., Oregon and Washington adopted designated uses for the protection of Pacific salmon). Washington's designated uses explicitly denote the following categories of aquatic life uses: char spawning and rearing; core summer salmonid habitat; salmonid spawning; rearing and migration; salmonid rearing and migration only and several others (WAC 173-201A-200). Washington's designated uses should provide additional protection for Washington's native char, bull trout and Dolly Varden, and several species of Pacific salmon that are listed as threatened or endangered, as well as others that are not listed. This is likely an improvement over the more generalized goals of “for the protection and propagation of fish, shellfish, and wildlife” or “fishable”.

Table 37. State designated uses that explicitly address threatened and endangered species.

State	Designated Use Name	Designated Use Description	EPA Effective Date
CA Regions 1, 2, 3, 4, 5, 6, 7, 8, 9	Rare, Threatened, Or Endangered Species	Uses of water that support aquatic habitats necessary, at least in part, for the survival and successful maintenance of plant or animal species established under state or federal law as rare, threatened or endangered.	8/18/1994
PR	Class SB, SC, SD	Coastal waters and estuarine waters intended for use in primary and secondary contact recreation, and for propagation and preservation of desirable species, including threatened or endangered species.	6/26/2003

Careful consideration of the relationship between the value statement of use and the manner of evaluating attainment of the use is essential. When the relationship between the endpoint and the indictor is weak particular life stages of regionally important species and regional biota may be under-protected. Portions of the native aquatic community may be left unprotected by omission and unique life histories may be overlooked. For instance, Washington’s designated uses may generally protect spawning salmon, but are under protective of early or summer migrating adult salmon for water temperature where warm water temperatures may interfere with gamete development during the migration and holding of the early migrating spawners (T. Hooper, pers. comm., October 28, 2008). Additionally, broadly defined designated uses are difficult to translate into meaningful and measurable criteria for determining whether uses have been attained. The closer a designated use is linked to its indicator, the chance of falsely concluding that the designated uses are being attained, when they are not, decreases.

To address this problem the NRC (2001) recommended greater stratification of designated uses at the state level to provide a logical link between designated uses and attainment of that use (NRC 2001). Considering that the designated use is the description of the desired endpoint for a waterbody and the criterion is the measurable indicator for determining attainment, using a stratified designated use framework could allow state’s to measure ranges of attainment, create stronger linkages between endpoint and indicator, decrease decision risk, etc. The further the criterion for determining attainment is apart from the desired condition (the designated use) the greater chance for introducing (or magnifying) error into the decision-making process.

Figure 4 illustrates some examples of water quality criteria as the measurable indicator for attainment of designated uses in relationship to the desired endpoint, attainment of uses (after NRC 2001). The unnumbered square represents the designated use for the water (depicted by a value statement such as “fishable” or “swimmable”). Square 1, the furthest from the designated use, represents measures of pollutants at their source (end of pipe measurements). Square 2 represents the chemical criterion as the measure of the ambient water quality condition, but may also include non-chemical measures (criteria) for physical attributes of ambient water quality such as dissolved oxygen and temperature. Square 3 represents criteria that are associated with physical or biological sources of pollution, and might include such measures as flow timing, pattern, non-indigenous taxa, channel sinuosity, etc. Square 4 represents biological measures of ambient water quality condition, such as those represented by indexes of biological community.

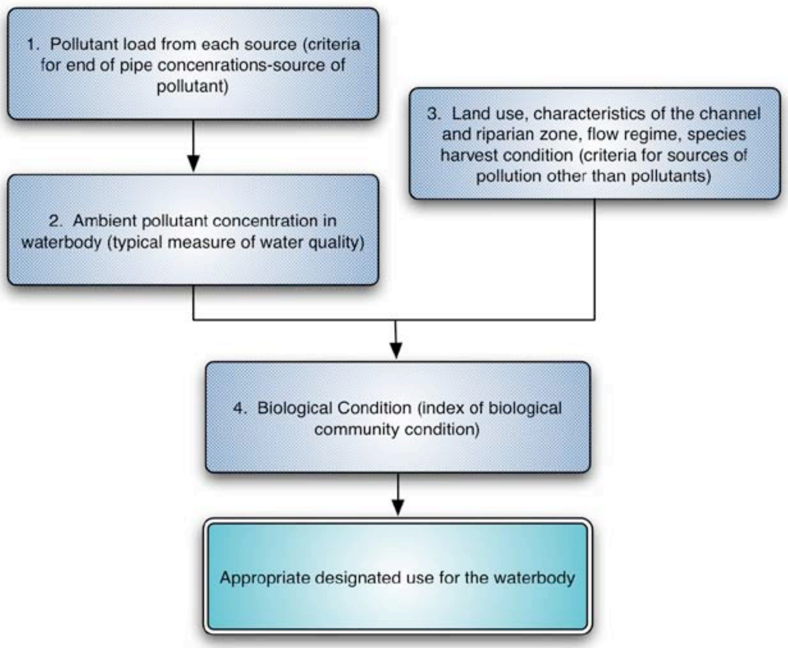

Figure 4. Types of water quality criteria and their position relative to designated uses (After NRC 2001).

A criterion, as described by NRC (2001) could be positioned at any point along the causal chain. However, if the desired endpoint is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters, the biological condition is closest indicator to the desired endpoint. Not only is the proximate position of the biological indicator closer to the designated uses that describe the desired biological community, the biological community reflects the interplay between the physical, biological and chemical conditions of its environment. Under the stratified designated uses framework as suggested by the NRC (2001), states would adopt biological indicators as an intermediate and measurable indicator of designated use attainment. An index of biological health that considers a balanced community of native species versus the abundance and viability of alien species, loss of sensitive species and long-lived species; hydrological regime shifts (alterations in peak flows versus low flows, timing, intensity and duration), and so on, would provide a more holistic view of water body health and it's ability to meet public goals.

If the outcome or desired state for a designated use is preserving the biological integrity of the native community, then more meaningful measures as to whether that designated use is being supported by the aquatic life criteria are necessary. One advantage of a more explicit biological framing of designated uses is that threatened and endangered species can be expressly incorporated into the designated uses. When the designated uses are explicit, and provided the criteria properly support such designated uses, the broader biological community should be protected. In turn, it would be reasonable to expect that enhanced aquatic conditions may prevent more aquatic species from becoming listed under the ESA, and promote the survival and recovery of currently listed threatened and endangered species suffering from poor water quality. In contrast, when the biological community is not a measured indicator of what EPA intends to protect through its chemical indicators, then EPA and the states are engaged in a water quality process, including designation of uses, to "merely to justify the specific numbers contained in pollutant criteria (Gaba 1983)." Absent robust indicators, Gaba (1983) notes that EPA, in reviewing the adequacy of state water quality standards is also engaged in an "ad hoc" assessment of whether the states are satisfying the minimum requirements of the CWA, and what kinds of fish or other wildlife are to be protected under a particular designation (Gaba 1983; Stephan 1985).

NMFS is particularly concerned about those instances where EPA finds that a criterion can adversely affect certain populations of listed species, while simultaneously protecting designated uses. Although individual listed species and the population they represent are part of the native aquatic assemblage within a waterbody and depend upon quality waters for protection and propagation, according to EPA it cannot disapprove a state's designated use solely on the basis that the designated use does not provide for the protection against "take" of listed species (EPA 2008b). Yet, where a listed fish species is failing to mate, rear, feed, migrate, or maintain viable populations for reasons attributable, in part, to water quality, it follows that the standard is not providing for the protection and propagation of at least some fish.

States and tribes that wish to avoid water quality related impacts to listed species could write their designated uses to include the protection of listed species, as a general category and, if necessary, include species specific designated uses. When states include the protection of the viability of listed species as a designated use, as a general matter, those states should be able to demonstrate that they would not be likely to increase a listed species risk of extinction due to chemical water quality impacts so long as they are meeting their designated uses. To demonstrate this level of protection would require a strong linkage between the designated use and the criteria states use for evaluating attainment. States that rely on chemical criteria without biological criteria to measure the attainment of designated uses, and fail to designate biologically meaningful indicators of use, may miss important changes in environmental health attributable to water quality impacts, including changes in the viability of listed species populations (see for instance, Karr et al. 2003). Currently, however, the approach used by most states in evaluating the effectiveness of the criteria (and other water pollution control efforts) at meeting the designated uses is unlikely to present a very complete or comprehensive picture of the biological health of their waters from chemical or physical stressors, and therefore cannot provide a very complete picture as to the successfulness of the water quality control program (GAO 2002; Karr et al. 2003). According to Gaba (1983) EPA has allowed states to trivialize designated uses as a scientifically credible endpoint by allowing designated uses to justify the specific numbers contained in pollutant criteria, which EPA has predetermined support any designated uses that would comply with the very general goal of the CWA.

Arguably it is even more important that EPA recognize that confidence in the ability of aquatic life criteria to protect the aquatic assemblage is increased when chemical and biological criteria are used in concert to evaluate environmental impacts. The traditional laboratory based studies used as the basis for recommending aquatic life criteria require validation using more definitive and biologically rigorous metrics of biological integrity of natural systems. According to Adler et al. (1993, citing CRS, 1972 Legislative History, 76-77), the definition of “biological integrity” includes a condition in which the natural structure and function of ecosystems is maintained, and natural levels of biological integrity are those “levels believed to have existed before irreversible perturbations caused by man’s activities.” While the Senate report instructed that integrity under the CWA ought to be determined by reference to historical records on species composition (Adler et al. 1993). Biological integrity as defined by Karr and Dudley (1985) is “the capability of supporting and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the region”. If it were EPA’s intent to design aquatic life criteria that protect the designated use for “fishable” waters, they would test the validity of *whether criteria are protecting the aquatic assemblage* in a waterbody, using rigorous biological indicators of aquatic ecosystem health.

The *native* aquatic assemblage is, arguably, the relevant endpoint envisioned by the Congress in establishing the CWA – when they stated the objective of the CWA is to restore and maintain the chemical, physical, and biological integrity of the Nation’s waters -- and is certainly is that envisioned by Congress in adopting the ESA. Regardless, EPA could engage states in better defining the objectives of their uses classifications, and identifying measureable indicators of attainment. More importantly, EPA should review their operational definition of protecting the aquatic assemblage in a waterbody and how the definition should be expanded beyond the limited indicators of species richness and species evenness to better reflect current science for a biological healthy aquatic community, and incorporate their affirmative duties under both the CWA and ESA (EPA 2008b). Species richness and species evenness are not necessarily indicators of the health of the native aquatic fauna. They can, however, be combined with other important variables for assessing the biological condition of a water body such as: species diversity, trophic composition, fish abundance, fish health metrics (e.g., body condition), presence (or absence) of non-native species, presence (or absence) of tolerant (or sensitive) species. EPA might also adopt a stratified designated use approach, rigorous and measurable indicators of the native aquatic assemblage in those states where EPA retains primacy for setting water quality standards, and engage in meaningful field studies for assessing the status of surface water integrity that integrates chemical criteria with indicators of biological and physical condition. While EPA has stated the guidelines for establishing aquatic life criteria are meant to protect aquatic assemblages, the laboratory studies used in setting the aquatic life criteria for cyanide do not represent species’ compositions from a natural community or ecosystem and consequently may fail to identify toxicant/population/community interrelationships. If field monitoring is not feasible, then mesocosm studies could provide EPA an opportunity to take a replicatable, laboratory-controlled approach to evaluate higher order effects in aquatic systems. Such studies may be useful in examining the indirect effects of reduced water quality and community response.

Stressors and Subsidies Associated with the Proposed Action

The primary stressor associated with the proposed action is aqueous cyanide. The following sections provide background on the characteristics of cyanide as a pollutant; including its uses and sources, observed concentrations, and other information that helps establish the exposure profile---the magnitude and spatial and temporal patterns of cyanide occurrence in the environment to which listed resources are exposed---for this analysis.

Exposure Analysis

Cyanide Sources and Production

We examined the typical sources of cyanide and the geographic distribution of those sources of cyanide to determine whether we would expect cyanide would co-occur with listed resources. This effort was based on the presumption that the fewer sources of cyanide there are across the United States, and the more limited their spatial distribution, the less likely that listed resources would be exposed to cyanide during their lifetime. If through this examination we would find that cyanide does not co-occur with listed resources, then we would conclude there is no exposure. The evidence leads us to conclude, however, that this is not the case. That is, based on the large number of sources of cyanide, their wide spatial distribution, and the increasing production of cyanide in the United States, we expect listed resources are more likely than not to be exposed to one or more sources of cyanide during their lifetime.

A common misconception is that cyanide is predominantly associated with gold mining or other mineral processing operations, which would tend to make this predominantly fresh water and perhaps rural pollutant. While cyanide is widely used in ore-extraction and cyanide related mine accidents have been widely publicized particularly when they have led to massive fish kills and human impacts, cyanide enters waterways from a wide variety of sources. Cyanide is ubiquitous in the environment, at least at low levels, as it is produced by a number of plants and microorganisms. However, cyanide is also produced synthetically to support industrial uses and is a byproduct of certain industrial processes (Leduc 1984; Eisler 1991; Dzombak et al. 2006).

Humans contribute the vast majority of cyanide to the environment. Cyanides are used widely in steel and heavy metal industries (e.g. electroplating), the manufacture of synthetic fabrics and plastics, as a pesticide and as an intermediate ingredient in herbicides, in road salts, and some fire retardants. Cyanide is also a byproduct of other activities such as municipal waste and sludge incineration and coking and gasification of coal (see Table 35). Of these sources metal industries and organic chemical industries are major contributors of cyanide into the freshwater aquatic environment, whereas, atmospheric cyanide, a by-product of forests fires, may be the primary source of oceanic cyanide except where cyanide enters coastal waters from fresh water sources (Leduc 1984; EPA 2005; Dzombak et al. 2006). Wastewater treatment plants across the United States can also be unexpected, but significant sources of cyanide to both fresh water and saltwater environments through several chemical processes, including dissociation of thiocyanide by chlorination or UV disinfection, chlorination in the presence of residual ammonia, nitrosation, and photolysis of ferrocyanate (Kavanaugh et al. 2003).

According to the 2002 United States Economic Census, there are 180 facilities engaged in gold-ore mining in 27 states across the nation, including Alaska, California, Idaho, Massachusetts, and Florida (U.S. Census Bureau 2002). The top four states, in terms of number of facilities, were Nevada, Colorado, California, and Alaska. In contrast, the manufacturing of photographic film, paper, plate, and associated chemicals occurs in more than 400 facilities and 24 states across the nation, and more than 3,000 establishments engage in electroplating and related activities in 41 states across the nation (U.S. Census Bureau 2002). The influx of cyanide to aquatic environments is likely as widely distributed across the landscape as the industries that use cyanide as part of their routine operations.

Cyanide is also synthetically produced in several states across the nation including Texas, Wyoming, West Virginia, Nevada, and Ohio (CMR 2008). In fact, the synthetic production of cyanide in the United States is a growing industry. The United States production of hydrogen cyanide (HCN) more than doubled in the past two decades from 330,000 tons in 1983 to 750,000 tons in 2001. Production growth between 1997 and 2000 increased about 1.7% per year (Dzombak et al. 2006; CMR 2008). The Chemical Market Reporter indicated that production demand in 2004 was estimated at nearly 2 million pounds. With demand exceeding current production of HCN, and price growth positive for the producers, HCN production and availability is expected to continue to increase in the United States. Incidentally, the United States does not export domestically produced HCN (CMR 2008).

The largest portion of the HCN produced in the United States is used in the textiles industry, for nylon production (47% is used for adiponitrile). Whereas, 27% is used in the production of acetone cyanohydrin for methyl methacrylate, the monomer for the transparent plastic polymethyl methacrylate also known as acrylic, 8% is for the production of sodium cyanide (NaCN), 6% is for methionine, 2% are chelating agents, 2% for cyanuric chloride, and 8% goes to miscellaneous uses including nitrilotriacetic acid and salts (CMR 2008). The demand for nylon remains high, with new growth and new applications still strong. According to CMR (2008) one such new application is in the automobile industry where metal components are being replaced by nylon parts. At the same time acrylic demands remain high, while the declining price of gold has reduced the demand for NaCN production, which had formerly been the primary driver for HCN production. With overall demand for HCN production growing in the United States, clearly cyanide is not a chemical that is being phased out of production or practical use but remains in prominent use. In fact, acrylonitrile (vinyl cyanide), a monomer in the synthesis of adiponitrile, is among the top 50 chemicals produced in the United States (Dzombak et al. 2006). While HCN facilities that support acrylonitrile production are in several states across the United States, several of the largest producers are in Texas (CMR 2008).

Table 38. Industrial Sources and Uses of Cyanide Compounds.

Source/Use	**Form**	**Reference**
Energy Production - Coal Gasification	Cyanide salts (potassium cyanide, sodium cyanide)	Way 1981; EPA 2008c
Steel manufacturing & heat-treating facilities, metal cleaning, electroplating		WHO 2004; Leduc 1984; EPA 2005
Ore-extraction (gold-mining, coke extraction)		WHO 2004; Leduc 1984; EPA 2005

Dyeing, printing of photographs		WHO 2004; EPA 2005
Production of resin monomers (acrylates)		WHO 2004
Pigments, paints	Ferrocyanides	Dzombak et al. 2006
Fire retardants		Little and Calfee 2002
Anti-caking agent for road salts		Dzombak et al. 2006
Detergents, dyeing of textiles		Dzombak et al. 2006
Pharmaceuticals (antibiotics, steroids, chemotherapy)		Dzombak et al. 2006
Fumigant/pesticide	Hydrogen cyanide, metallo-cyanide compounds	WHO 2004, Dzombak et al. 2006
Herbicides (dichlobenil, bromoxynil, bantrol)		EPA 2005, Dzombak et al. 2006
Road salts		EPA 2005
Production of other cyanides (e.g., sodium cyanide for gold mining)		EPA 2005, Dzombak et al. 2006
Pyrolysis of paper, wool, polyurethane		WHO 2004
Chelating agents for water and wastewater treatment		EPA 2005, Dzombak et al. 2006
Production of clear plastics		Dzombak et al. 2006
Methionine for animal food supplement		Dzombak et al. 2006
Wastewater Treatment Facilities (secondary treatment and/or disinfection w/ chlorine or UV)		Kavanaugh et al. 2003
Automobiles (with older or malfunctioning catalytic converters)		Voorhoeve et al. 1975; Karlsson 2004

With increasing uses and increasing production of cyanide we would expect that the amount of cyanide entering the environment would also be increasing (Way 1981). However, we have little data to ascertain if this is the case. According to the Toxics Release Inventory (TRI) data, total reported hydrogen cyanide releases have been increasing over the past 20 years (Figure 5). In 2008, some 4.5 billion pounds of HCN were released; over 432,000 pounds represent air emissions, while 58,000 pounds were discharged to surface waters (EPA 2008d). In comparison the long-term trend in releases to surface waters is declining, although this may not be a reflection of trends in actual ambient instream concentrations for several reasons. First, the data reflect one type of cyanide compound for which release data exists and does not include an assessment of the fate and transport of the released HCN including the ability of cyanide compounds to undergo transformation as under some environmental conditions that can increase or decrease its toxicological impact, and the TRI data does not include non-point sources of cyanide to the environment. Nonetheless, the TRI data, with its many caveats represents one of the only sources of data upon which trends in potential ambient cyanide can be discerned.

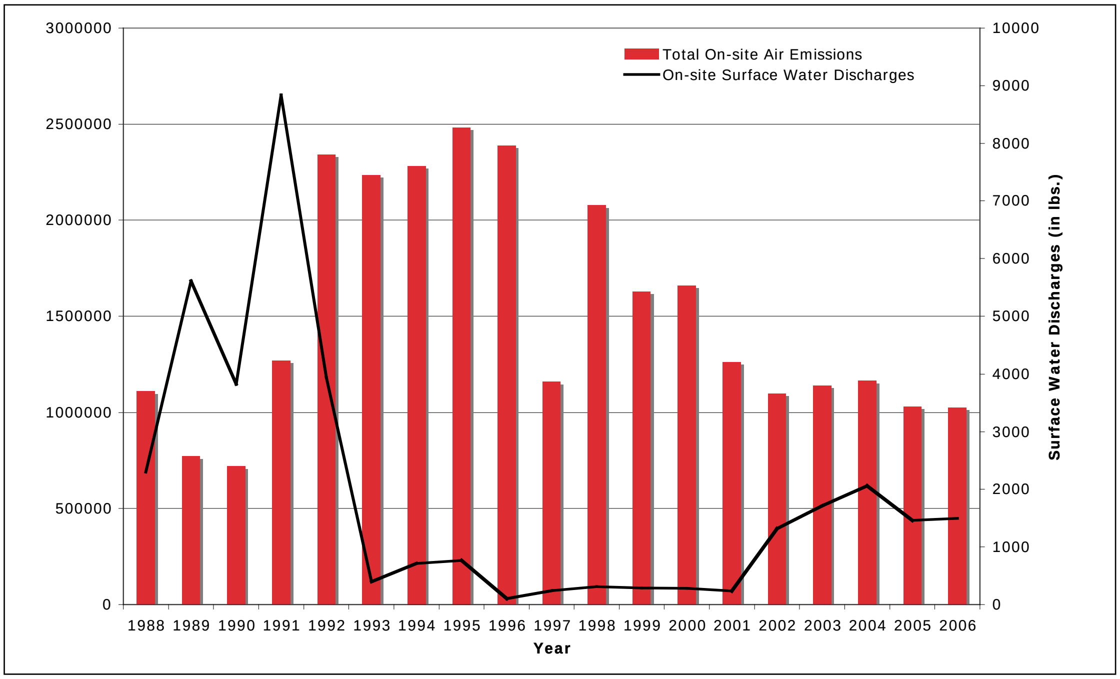

Figure 5. Toxics Release Inventory Data for HCN Releases in the United States to Air and Surface Waters, 1998 to 2006 (Source EPA 2008c).

The TRI data can also be used as indicator for understanding the geographic distribution of cyanide, in this case HCN, across the nation. The TRI data set, together with information on the distribution of manufacturers and user groups provide some insight into the distribution of cyanide sources and those areas where species might be at a higher risk of being exposed to ambient cyanide. While it is not clear that the volumes of cyanide discharged in these states typically resulted in aqueous concentrations that were problematic for listed species, the foregoing discussion illustrates that cyanide sources are widely distributed, and cyanide production and use is far from waning. On the contrary, cyanide production has increased in the past and is expected to increase in the future. As a result, we would expect listed aquatic resources are likely be exposed to one or more sources of cyanide during their lifetime. Due to the nature of the industrial sources, most exposure would occur in fresh water and marine coastal waters influenced by human activities. The predominant sources of cyanide to marine waters would be from direct discharges to marine waters (typically coastal outfalls), downstream transport from freshwater sources, and incidental releases from vessels (Dzombak 2006), which generally suggests that the further from shore a species or critical habitat occurs, the less likely it would be exposed to a wide variety of cyanide sources. However, with a large portion of cyanide entering the environment in gaseous form, we would expect some cyanide likely enters marine and fresh waters through atmospheric deposition.

Concentrations of Cyanide in U.S. Waters

As noted earlier, cyanide enters waterways through a variety of pathways and sources; however, the direct discharges (from point and nonpoint sources) pose the greatest concern for aquatic habitats because these sources are likely the dominant sources of cyanide loading to United States waters. To further characterize the exposure of listed resources in the aquatic environment, we asked whether and to what degree we would expect listed resources would be exposed to cyanide concentrations at or near EPA's recommended CCC or CMC for cyanide. We examined data in EPA's data base STORET (STORage and RETrieval data warehouse) for information on potential concentrations of cyanide in the environment, as well as individual studies of cyanide loading from various sources. Based on our evaluation, we expect listed resources will be exposed to a wide range of concentrations of cyanide, and a wide number of cyanide compounds with varying toxicity. We expect that most waters likely have some low-level background concentrations of cyanide at most times. When exacerbated by anthropogenic sources, in-water concentrations may exceed EPA's approved numeric criteria for cyanide and the averaging recommendations adopted in state standards.

Studies have detected low levels of cyanide as a natural condition in some waterways, likely resulting from plant and microbial input. There also appears to be a seasonal component to the cyanide loading in waterways, which presumably varies with cyanogenic plant production, atmospheric deposition and rainfall patterns. A study of the occurrence of cyanides (free and combined) in small streams in the North-West Germany, using a technique that allowed a detection limit of 0.1 µg/L, found annual values of total cyanide in rural watersheds was 3 µg/L, while mean annual values of total cyanide in industrial watersheds were 20 µg/L with values reaching over 200 µg/L (Krutz 1979 in Leduc 1981, Krutz 1981). Cyanide concentrations varied seasonally, with the lowest concentrations occurring in spring and late summer and highest concentrations occurred in winter. Krutz (1979 in Leduc 1981) calculated maximum winter loads at 6 g CN^-/day and summer loads at 0.2 g CN^-/day. Principal factors attributed to winter peak loading included increased potassium loads that induced cyanogenic microorganism activity and winter precipitation and runoff events that increased delivery of atmospheric cyanide and cyanide formed by plants and terrestrial microorganisms to the water. Seasonal peaks were more frequently observed in the small catchments, although seasonal peaks were also observed in medium to large sized catchments (Krutz 1979 and PPWB 1978 in Leduc 1981). On the other hand, Tarras-Walberg et al. (2001) found concentrations were highest when the river under study was in a low flow period. In many cases, the low flow period for a catchment would correspond with low-flows and peak vegetative growth within a basin. Consequently, small catchments tend to be more closely associated with streamside vegetation and allochthonous input of cyanogenic (and other) plants, which would explain the summer and low-flow peaks observed by Krutz (1981) and Tarras-Walberg et al. (2001).

Cyanide also enters waterways through the indirect pathway from airborne sources, such as burning waste biomass for energy conversion, crop burning, prescribed forest fires and wildfires, and through the atmospheric release of cyanide from industrial sources and the eventual transformation to aqueous cyanide. Barber et al. (2003) found that free cyanide concentrations in stormwater runoff collected after a wildfire in North Carolina averaged 49 µg/L, an order of magnitude higher than in samples from an adjacent unburned area (Barber et al. 2003). Atmospheric deposition of HCN may be one of the most significant sources of HCN to ocean waters, excluding coastal areas. However, according to Dzombak et al. (2006) the concentration of HCN in ocean waters is likely to be low (less than 1 µg/L than the criterion value for salt water).

Studies evaluating the direct discharge of cyanide to waterways indicate that the concentrations entering water are as variable as the sources themselves. Studies have shown that stormwater melting off roadside snow has a much greater capacity to accumulate and retain heavy metals and other pollutants than summer stormwater runoff. In a study of urban highway sites, concentrations of cyanide and metals were orders of magnitude higher than at the control sites and exceeded storm water (rain) runoff concentrations by one to two orders of magnitude. Cyanide concentrations, although demonstrating some variability, remained relatively constant at all sites (averaging 154 µg/L) or increased according increasing application rates of deicing salts that contained cyanide compounds as anti-caking agents (Glenn and Sansalone 2002). A study on the effect of cyanide on the anaerobic treatment of synthetic wastewater noted that cyanide is produced on an industrial scale of 2–3 million tons per year and, therefore is in many different industrial wastewaters. The concentrations encountered in industrial waste generally are in the range 0.01–10,000 mg/L, most of it in complexed forms of cyanide, which are less toxic than free cyanide but can transform to free cyanide or HCN. Cyanide contamination also occurs in the processing of agricultural crops containing high concentrations of this compound, such as cassava[12]. Systematic surveys of large wastewater effluents in Southern California suggest that free cyanide is routinely found in wastewaters, at low levels. In different years reported from 1992 – 2002, mean cyanide concentrations in effluents ranged from <2 to 30 µg/L (Steinberger and Stein 2003). Data from the US National Urban Runoff Program in 1982, revealed that 16% of urban runoff samples collected from four cities (Denver, Colorado; Long Island, New York; Austin Texas; and Bellevue, Washington) contained cyanide concentrations ranging from 2 to 33 µg/L (Cole et al. 1984 in ASTDR 2006). While demonstrating variability in the concentrations of cyanide found in some discharges, these studies also indicate that cyanide concentrations can be quite high at times.

The Difficulties of Measuring Cyanide in Water

Dzombak et al. (2006) refer to measuring cyanides as “a regulatory dilemma” because most analytical methods used in the field do not target specific cyanide compounds, rather the methods report various cyanide groups. EPA’s recommended aquatic life criteria are specified in terms of free cyanide, yet the conventional sampling methods provide for measurement of a group of cyanide compounds. Methods include total cyanide, weak-acid-dissociable cyanide (WAD), cyanide amenable to chlorination (CATC), available cyanide by ligand exchange, and free cyanide. Total cyanide, the most frequently conducted sampling method, measures free cyanide and metal-complexed forms of inorganic cyanide, while WAD measures weak metal-cyanide complexes plus free cyanide. Much of the older data available in such databases like STORET were measured and reported in terms of total cyanide, which although it could be used as a surrogate of the amount free cyanide in a sample, doing so would lead to an overestimate in the amount of free cyanide in the samples because total cyanide includes free cyanide, WAD cyanide plus the relatively non-toxic iron-cyanide complexes. When EPA published their recommended aquatic life criteria for cyanide in 1985, they recognized the incongruity between publishing numeric criteria for free cyanide, and the fact that no EPA approved sampling method was available at the time that would measure free cyanide (EPA 1985). Therefore, in 1985 EPA recommended that states apply the criteria to total cyanide, acknowledging that doing so may

[12] Although we are not aware whether the U.S. has any cassava processing plants, there are over 1,000 cyanogenic plants including many sorghum grains that may contribute to cyanide contamination when processed. Cassava is merely one of the best documented sources of cyanide contamination attributable to cyanogenic plant processing.

make the water quality standard over-protective. An approved method for measuring free cyanide is now available, but unfortunately a translator has not been developed to convert data on total cyanide to free cyanide (Kavanaugh et al. 2003).

At the same time, there is a concern over measurement precision with data found in sources such as STORET. Measurement precision varies among sampling methods and certain chemicals and procedures can interfere with measurements as well. Measurements are frequently conducted via colorimetric, titrimetric, or electrochemical finish techniques (Dzombak et al. 2006). Measurements of total cyanide are limited to detection in reagent water matrix of about 1 to 5 μg/L and do not measure: cyanates, thiocyanates, most organic-cyanide compounds, and most cobalt and platinum cyanide complexes (Dzombak et al. 2006). Problems with sample storage, regulatory criteria, and the methods for testing and their sensitivity are a concern (Eisler 1991; Dzombak et al. 2006). Eisler (1991) notes that due to the volatilization of cyanide, periodic monitoring is not informative (for example, monitoring once per quarter [for instance, see the permit requirements in EPA 2008e) except perhaps, where continuing or chronic conditions persist. Consequently, Eisler (1991) and others recommend that continuous monitoring systems are necessary, with particular emphasis on industrial dischargers, to understand the fate and transport, critical exposures, and relative contributions of human and natural sources of cyanide in the aquatic environment. The availability of data from case studies using continuous monitoring systems would significantly increase our understanding of cyanide in the aquatic environment, and provide us important exposure profiles for evaluating approved water quality standards. Unfortunately, we were not aware of any such data sets that we could examine as part of this analysis.

STORET – EPA's Main Repository for Water Quality Data

Since we do not have data on long-term studies using continuous monitoring systems to evaluate cyanide discharges, we conducted a query of EPA's STORET database to further characterize cyanide entering the action area for this consultation. STORET, EPA's main repository for water quality data, contains information on water quality collected from a variety of organizations across the United States, from small volunteer watershed groups to state and federal agencies (http://www.epa.gov/STORET/index.html). Our review of STORET data indicates that many dischargers reported no-detectable amount of cyanide in their samples, which in some case may have been a limitation of the sampling method and does not necessarily suggest that the water contained no cyanide or alternatively it may suggest that the discharges were free of cyanide – either way, we do not know. We searched the STORET database and found records spanning 1964 to 2008 (August), most of which were recorded as total cyanide. Some states of particular interest, like Washington, where NMFS has listed salmonids and where the TRI database suggests there have been large discharges of HCN to surface waters, were not represented in STORET. While data were available for several other states, data was often sparse for many of the coastal states where NMFS' listed resources occur. When we queried according to the data fields for "rivers, lakes, reservoirs, and canals" the database returned only one sample for Alaska and Oregon, 13 to 51 samples for states such as California, New Jersey and North Carolina. The largest number of samples in this category was from Florida. We compared the sample data to approved water quality standards for cyanide and found that 4 of 13 values reported for California (31%) exceeded the CCC and the CMC. Upon closer inspection it appears that most of the California data came from reservoirs and streams in the Mojave Desert that were presumably impacted by gold mining. The minimum value above the water quality standards was 300 μg/L and the highest reported value was 5,000 μg/L. For New Jersey, 17% of the reported values were above the CCC and the CMC. The highest reported concentration of cyanide above the approved water quality standard was 130,000 μg/L CN_T –there were two samples at this concentration in the data set, taken two weeks apart. Two months later, during the same year a concentration was sampled of 84,000 μg/L CN_T. All three of these samples were taken from the Ramapo River (near Mawhaw, New Jersey). Similarly, in Mississipi data show that state water quality standards were exceeded in 13% of the reported samples. When we queried STORET for data from marine waters we found only 5 reported values. All the samples were taken in Puerto Rico in about a 9-month span beginning late 2005. The mean concentration for this sampling station was 4.6 μg/L CN_T, four times higher than the approved water quality standard for marine waters, while the minimum reported concentration was 0 μg/L CN_T and the maximum concentration was 20 μg/L CN_T.

There are several very strong arguments that can be made questioning the utility of the STORET data for establishing an exposure profile for aquatic species. Not the least of which are the arguments that (a) the vast majority of information on cyanide in STORET is represented by total cyanide (CN_T) and a translator is yet to be developed that would allow us to determine the proportion of free cyanide (the most biologically toxic form) represented by the data on total cyanide, (b) the scarcity of data generally provides us little understanding of the spatial or temporal patterns of cyanide concentrations in United States waters, particularly since some states do not report their monitoring to STORET (or perhaps those states are not monitoring for cyanide), and (c) there are insufficient replicate data in STORET to provide any meaningful illustration of the trends in cyanide discharges within a particular locality. Despite the limitations of the data in STORET, it (with TRI data) represents some of the best available information we have on cyanide discharges across the United States. The STORET data however, does illustrate that listed resources may be exposed to a wide range of cyanide concentrations in receiving waters and that those concentrations may vary widely relative to EPA's approved (and recommended) national numeric criteria.[13]

Given typical monitoring schemes in many permits the probability that a particular facility would detect an exceedence event is quite low. A typical permit may require sampling once a week, once a month, or less frequently, and will often conduct their sampling using grab samples[14] (see for instance, permit requirements in EPA's 2008 Multisector General Permit). To determine whether or to what degree grab samples might detect events in which water quality criteria had been exceeded, we considered several scenarios. In the first scenario, we considered a facility that has 52 discharge events a year that result in elevated cyanide concentrations and assumed each discharge event lasted eight hours. In this scenario, there would be a 95% probability that the event would not be detected by a grab sample. Conversely, there would be a 5% probability that the event would be detected by a grab sample. If we increased the number of discharge events to 110 events per year with each event exceeding a particular criterion value and each

[13] An interesting question that merits exploration is whether EPA and/or the respective states considered many of the recorded peak discharges events that are included within the STORET database as violations of water quality standards.
[14] The grab sample technique is a rapid collection single point sampling method that does not integrate vertical or cross sectional variability, but captures point concentrations near the water's surface.

event lasts 8 hours, there would be a 90% probability that the event would not be detected by grab samples, and a 10% probability that the event would be detected by a grab sample [15].

A discharge containing a high concentration of cyanide would have to occur for more than 180 days a year (24 hours/day) to have a high probability of detection, which suggests that random grab samples generally are not likely to detect an exceedence event. Therefore, the sample data we found in our query of STORET may not have been produced by truly random samples, but instead, were produced by samples taken after known discharges containing high concentrations of cyanide. The fact that some samples data points reported high concentrations of cyanide could also be attributed to serendipity during the timing of sampling, or it could be that the discharged concentrations are high for frequently long intervals of time (e.g., more likely than low concentrations).

Allowable averaging schemes contained in many NPDES permits would further mask the true distribution of sample concentrations to which listed resources are exposed. That is, recall that the approved standards include a provision that allows for the average of the 1-hour concentration for the CMC, and the average of the 4-day concentration for the CCC. A facility that takes ten samples a year may have one sample exceeding 200 µg/L CN and nine samples with “non-detectable” concentrations and that facility would still fall within the recommended limit recommend by EPA. Figure 6 illustrates three hypothetical scenarios that demonstrate how individual discharges may exceed the approved numeric standards for the cyanide CMC, but still fall within allowable standard when averaged accordingly. The three alternatives presented illustrate three scenarios, all with the same central tendency despite widely different sample distributions. The result is that all three scenarios would be presumed equal in perceived risk under the recommended averaging scheme, despite the actual and widely disparate concentrations to which fish and wildlife would be exposed. As a result of the averaging and infrequent sampling schemes, the power of the data to detect problems is exceedingly low, and the fact that so many samples reported in STORET are unusually high is cause for concern and suggests that in some areas cyanide concentrations may exceed the numeric values defined by the cyanide CCC and CMC fairly often. Consequently, based on the best available data it appears that at least some listed resources would be exposed to cyanide at concentrations well above the approved CMC of 22.36 µg/L CN and the CCC of 5.221 µg/L CN, and at a frequency and duration which may result in demonstrable harm to aquatic life.

Factors That Influence Cyanide Toxicity

The risk to aquatic environments from cyanide releases depends on several factors including: the cyanide compound and concentrations released, pH, presence of iron and other metallic trace elements, solar radiation, air and water temperature, and dissolved oxygen levels to name a few (Doudoroff 1976; Smith et al. 1978; Dzombak et al. 2006). There are several compounds in the cyanide group, with varying degrees of complexity. Cyanide is formed by carbon and nitrogen, attached by three molecular bonds ($C \equiv N$). Complex cyanide compounds are formed when one or more CN compound forms with other atoms, such as hydrogen ($H - C \equiv N$). The resulting

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[15] The probabilities in this paragraph were derived using the equation: Probability = $1 - (1-p)^n$ where p is the probability of a water quality exceedance event in a sample, n is the number of samples, so $(1-p)^n$ is the probability of not detecting a water quality exceedance event, *and* then $1 - (1-p)^n$ is the probability of detecting a water quality exceedance event in a sample of size n. Adapted from McArdle 1990.

compound is HCN. Hydrogen cyanide (HCN(g)) is a gas that is miscible in water, and its water form, hydrocyanic acid (HCN(aq)), is weakly acidic and the most toxic cyanide compound. Other compounds include sodium cyanide (NaCN), potassium cyanide (KCN), adiponitrile ($C_6H_8N_2$), and copper cyanide (CuCN).

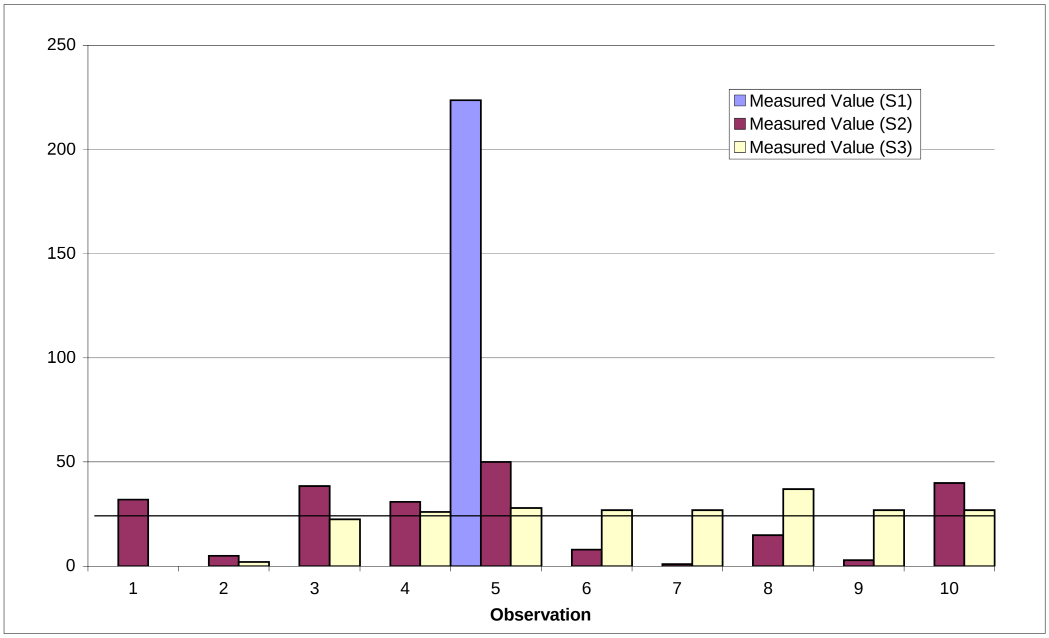

Figure 6. Three Hypothetical Discharge Scenarios that Comport with the Acute Water Quality Standard for Cyanide (Avg. CMC = 22.36 µg CN/L).

Free cyanide readily biodegrades, but the degradation is influenced by several factors including availability of oxygen, pH, carbon and nitrogen and the initial concentration of the cyanide compound released. Cyanide, through degradation, is converted to simple molecules like ammonia and carbon dioxide or it may it may be assimilated into the primary metabolism of bacteria, fungi, or plants (Dzombak 2006). Many forms of cyanide exist in the aquatic environment, including NaCN, KCN, metal-cyanide complexes, and organocyanides (e.g., acetonitrile), and metal-cyanide solids (e.g., ferric ferrocyanide). These forms have different chemical and toxicological properties. For instance, simple solids, like KCN and NaCN, are more soluble in solution and readily release free cyanide and HCN, which is the subject of this consultation. Solid forms of cyanide may exist in the soil of sites for years, and once exposed to water may result in dissolved cyanide reaching ground water and eventually surface waters (see Dzombak et al.'s [2006] discussion about the industrial legacy of cyanide box wastes at thousands of former manufactured gas plants in the United States). The chemical transformation of the cyanide compounds to HCN or CN^-, determines their toxicological significance to protected species, and their transport and fate in the environment.

The iron-cyanide complexes are the dominant form of cyanide found in soils and those most frequently encountered in dissolved form at concentrations in surface waters, making them the compound of most concern in managing water quality (Dzombak et al. 2006). The mobility of cyanide in soil and through groundwater depends upon precipitation, pH, the types of trace minerals present, and organic matter among other things. In water, cyanide transport, fate and toxicity vary according to volume dispersed, pH, temperature, mixing and turbulence, dissolved oxygen concentrations, form and abundance of alternative nitrogen sources, biological use, and incidental light as some cyanide complexes display photochemical reactivity (Leduc 1981; Kavanaugh et al. 2003; Dzombak et al. 2006). Regardless of what form of cyanide is introduced into a system, cyanide transformation mechanisms are variable according to environmental factors, and Kavanaugh et al. (2003) caution that managers need to acknowledge that multiple species of cyanide typically coexist, introconvert, and degrade in a system, and through its transformation the toxicological effect of cyanide may increase or decrease. Consequently, knowledge of cyanide compounds and their ability to undergo transformation is important to managing it in aquatic environments (Kavanaugh et al. 2003).

The Exposure Profile – Summarized

As noted earlier in this section (*Exposure – Cyanide Sources and Production*), we began our exposure assessment by examining general sources of cyanide across the United States, their spatial distribution and their production trends. We also examined available data to characterize CN concentrations in waters of the United States (the action area), and we compared the recommended (approved) numeric standards for cyanide to those values represented in data sets collected by EPA to determine if the numeric standards are representative of actual cyanide concentrations observed in United States waters. We also evaluated the ability of the data generally collected by EPA and authorized states to provide information that would help us make these comparisons.

Based on our analysis, we were unable to conclude that cyanide discharged in accordance with EPA's approved water quality standard has not co-occurred with listed and protected resources under NMFS' jurisdiction in the past, or that listed and protected resources would not be exposed to cyanide at some time in the future, such as over the course of the next 10 years[16]. As a result, we were unable to conclude that any particular listed or proposed resources should be excluded from our exposure analysis. The wide number of cyanide sources and uses and their broad geographic distribution suggests that some individuals of listed species, their designated critical habitat, and some individuals of species proposed for listing or their critical habitat proposed for listing, are all reasonably likely to be exposed to cyanide at some stage of their lives. Certainly, as the numbers of cyanide sources vary, the risk of exposure would also vary spatially and temporally across the action area. It appears that the potential for exposure may increase in urbanized areas, but rural areas are not free from potential sources of cyanide and some listed species would likely be exposed in these areas (e.g., gold mining and road maintenance activities are likely some of the sources in rural areas). In general anadromous fishes like salmonids and sturgeon, that traverse fresh and salt waters, would potentially be exposed to a greater number of cyanide sources throughout their life cycle, whereas listed marine species are more likely to be exposed to elevated concentrations of cyanide along the coasts than in deep or open ocean waters areas due to the combined effect of point and non-point sources from human activities. Both marine and fresh water species would likely be exposed to cyanide through deposition of

[16] We requested EPA identify a reasonable analysis period for their action, but did not receive assistance on this issue. Because the existing cyanide criteria were published in 1985 and have not been updated since, and we do not have any indication from EPA that they have plans to revisit their cyanide criteria, we used an analytical period of ten years. That is, our effects analysis considers the effects of continuing the approved cyanide criteria for an additional ten years into the future.

airborne releases. We did not find sufficient information to suggest that there were particular areas where listed species are not likely to be exposed to cyanide.

In a typical site specific assessment we would characterize the intensity of the listed resources and proposed resources exposure over time and space; however due to the inherent nature of this assessment, and the variability across sites and over time, such an estimate does not exist. Nor could we find data that we could use to assemble a case study of aquatic exposures in a particular space over a particular time. However, based on data collected by EPA (which is limited so the possibility for false positive errors (Type 1) and false negative errors (Type II) is high) it is clear that concentrations of cyanide have exceeded the approved standards in some locations and at some times. The data illustrate that the exceedances are sometimes orders of magnitude higher than the approved standards. Further, typical monitoring methods and the use of measures of central tendency on the collected data will often mask biologically important exposure scenarios. That is, we presented three alternative hypothetical sample data sets to illustrate that despite the distributions varied widely (i.e., even when individual events exceed the approved standards by 10 times) the perceived risk of the hypothetical sample sets would be presumed equal and in compliance with the approved water quality standard when using the central tendency as the measure of risk. In general, the monitoring and reporting practices routinely adopted in water quality standards severely reduce the utility of the data collected by EPA and states for characterizing typical exposure scenarios. Unfortunately, at the scale of this consultation and given the wide variability of the data available, it is not clear what might be a reasonable daily or longer-term potential dose for this analysis. Clearly, many factors influence the actual exposure of listed species in the wild and insufficient data are collected to evaluate the concentration, frequency and duration of allowable excursions, as well as the ambient concentrations to which authorized discharges are added. Simply, the criteria, as approved by EPA in state and tribal water quality standards, are the “protection level” to which the water quality based approach to pollution is applied. Absent better data to inform below and above criterion exposure events and other factors that influence exposure, we cannot confidently characterize the rarity or commonness of exposure scenarios that differ from the proposed criteria. Therefore, to anchor our response analysis for this consultation, we proceed with the core assumption that one or more life stages (all aquatic life stages) of all listed resources and resources proposed for listing would be exposed to cyanide at concentrations equivalent to EPA’s approved (and recommended) numeric water quality criteria. Since the CMC and the CCC represent the basis for administering water quality programs under the water quality-based approach to pollution control, including monitoring to determine whether waters are attaining designated uses, benchmarks for evaluating BMP performance in NPDES permits, evaluating whether waters should be listed as impaired, and as effluent limits for TMDL permits, we believe this is a reasonable core assumption for this analysis.

Response Analysis

As noted in our Approach to the Assessment, response analyses determine how listed resources are likely to respond after being exposed to an action’s effects on the environment or directly on listed species themselves. For the purposes of consultations on recommended or approved water quality standards, our assessments try to detect the probability of lethal responses, physiological responses, and behavioral responses that might result in reducing the fitness of listed individuals. Ideally, our response analyses consider and weigh evidence of adverse consequences, beneficial consequences, or the absence of such consequences.

It is important to begin these analyses by stating that, to the best of our knowledge, few data are available from the actual exposures of endangered or threatened species to cyanide in the laboratory or natural settings. We are aware of a few studies on rainbow trout, the resident form of *O. mykiss*; however, these studies are typically conducted on artificially propagated individuals that come from populations with a long history of artificial propagation such that their genetic make-up may be altered from their wild counterparts, and as a result there is some risk that their responses could differ from their wild counterparts. That said we have no information that would suggest this is the case and are assuming that there would be no difference in responses between artificially propagated individuals and wild individuals. Therefore, rainbow trout are the best surrogate available for predicting the response of wild steelhead, and many other species as well, because we lack species-specific data for several anadromous salmonids. We also have very little data for marine species as a group and no data on listed marine mammals. In fact, a recent reexamination of EPA's 1985 nationally recommended criteria for cyanide conducted by the Water Environment Research Foundation (Gensemer et al. 2007) concluded that "due to the lack of cyanide toxicity data for these species or reasonable surrogates", there was insufficient information available to evaluate the protectiveness of the saltwater cyanide criteria to threatened and endangered marine species. Instead, more research is needed on these species (Gensemer et al. 2007). Without empirical information on the actual responses of endangered and threatened species to cyanide, we reviewed the best scientific and commercial information available on the responses of fish and wildlife to cyanide. We also relied on estimates of sensitivity produced by EPA's Interspecies Correlation Estimations (ICE) model. We used this information to make inferences about the probable responses of endangered and threatened species when exposed to cyanide at the approved CCC and CMC.

Generalized Review of Responses

Individual aquatic organisms are exposed to cyanide by inhalation, ingestion, and absorption through epidural layers and mucus membranes. Cyanide is rapidly absorbed and distributed through the body. Once exposed, the primary manner of transport is via the bloodstream. In the bloodstream cyanide inhibits cellular respiration. Cyanide inhibits cytochrome c oxidase, an important hemeoprotein found in the mitochondria, by attaching to the iron in the protein it blocks the electron transfer to oxygen causing cellular respiration to cease. As a result many enzymes and biological systems are inhibited by cyanide, including succinic dehydrogenase, carbonic anhydrase and others (see Ballantyne 1987). Inhibition of cytochrome c oxidase activity, and the mitochondrial electron transport system will cause the cell to no longer aerobically produce ATP for energy, and the tissue then switches to anaerobic metabolism and the depletion of energy rich compounds (Eisler 1991; Dzombak et al. 2006). The result is rapid depression of central nervous system and the autonomic control of respiration. The heart is also a likely target of toxicity. Several species have shown consistently high concentrations within the myocardium, similar to brain concentrations, irrespective of the route exposure (Ballantyne 1987). Symptoms of acute poisoning in fish may include distress, increased ventilation – gill movement, surfacing, frantically swimming in circles at the surface, violently swimming against the bottom, convulsions, tremors, and finally death (Leduc 1981).

As a powerful and rapid asphyxiant, cyanide will cause death in a manner of minutes by hypoxic apoxia at lethal concentrations. Releases of cyanide at extreme lethal doses are likely rare based on known fish kills and STORET data, but they do occasionally occur. However, when they do occur, massive fish kills result. Some such events occurred in:

- Wissahickon Creek, Pennsylvania, where more than 1,000 fish were killed in 2006 due to the dumping of about 25 gallons of potassium thiocyanate, which is suspected of having interacted with chlorine in the nearby wastewater treatment plant (EPA 2006).
- Alamosa River, Colorado, where the Summitville Mine was responsible for contaminating 17 miles of the river and killing more than 15,000 trout in 1990 due to the escape of cyanide-laden pit waters. By the 1992, the site was abandoned by the mining company and was a notable superfund site, at high risk of additional leaks (Gavin 2004).
- Fall River, Oregon, where more than 22,000 trout died in 2002 when 1,000 to 2,000 gallons of fire retardant, which was released during fire fighting activities reached the waterway. The fire retardant was mixed with sodium ferrocyanide, which was used as corrosion inhibitor to protect the tanks the retardant was stored in (ODFW 2002).

Other events like these have occurred in the United States, and there have also been several events in other countries such as Ghana, Kyrgyzstan, and Canada, to name a few. Such events, while severe when they occur, tend to occur infrequently. Typically, we would find cyanide at much lower concentrations in the environment.

Cyanide although a potent asphyxiant, is also rapidly detoxified. The major determinant of the severity and rapidity of a response depends upon the rate of absorption versus the rate of detoxification, which are influenced by the rate and severity of exposure. Detoxification occurs primarily through the enzymatic transformation to thiocyanate, which is excreted by the kidney (Ballantyne 1987).

At sublethal doses, individuals may act stunned, which is why cyanide is widely used for the collection of tropical fish for aquariums. Sublethal doses can also inhibit reproduction, metabolic rate, egg production, spermatogenesis, oocyte development, lead to tissue necrosis, aggressiveness, impair food capture, and interrupt ion regulation and swimming ability (see Doudoroff 1976, Kimball et al. 1978, Leduc 1984, Eisler 1991). On the other hand, low-level exposure may also stimulate growth (Negliski 1973 in Dzombak et al. 2006; McCracken and Leduc 1980). Whether there are concomitant adverse effects to other physiological development process associated with growth stimulated by cyanide exposure is unclear. Rapid detoxification occurs at lower doses, as cyanide is metabolized to thiocyanate by two enzymes that are widely distributed in the body, and then excreted in urine over a period of days. Although thiocyanate (SCN^-) is the principle form of cyanide that is eliminated, it can also accumulate in tissues and is known to have antithyroidal properties. SCN^- inhibits iodine uptake by thyroid tissues and disrupts thyroid hormone homeostasis, which can result in the development of goiter. Cyanide does not bioaccumulate through the food web; however, the damage associated with prolonged exposure at low levels, recovery, and re-exposure may be cumulative. There is no evidence to suggest cyanide is mutagenic or carcinogenic (Ballantyne 1987).

Calculating a Response

Studies on the responses of listed resources and resources proposed for listing to cyanide and cyanide compounds are few. Directed studies on listed and proposed resources would generally rank highest for consideration, provided the studies were carefully designed, had large sample sizes (with small variances), and measured cyanide using a reliable test method. Such studies would generally provide the most reliable indicator of a listed species response, when exposed to cyanide in the wild. However, because data are not available for large number of fish and wildlife species EPA's *Guidelines* establish some minimum standards for deriving water quality criteria.

Generally, EPA would use the GMAV, which are calculated as the geometric means of the available SMAV to set the acute criterion, although this was not the case for their recommended aquatic life criterion for cyanide in fresh water. EPA calculated the acute freshwater value or CMC for cyanide (22.36 µg/L) to protect the recreationally and commercially important rainbow trout, the most sensitive of the species tested. Data were available for 15 different genera, and the most sensitive species of those tested was rainbow trout. At the time, rainbow trout was classified as *Salmo gairdneri*, and the other species in the same genera for which EPA had test data was the Atlantic salmon, which incidentally had a SMAV double that rainbow trout. Therefore, EPA chose to use the rainbow trout SMAV to set the acute criterion for cyanide. The acute criterion for saltwater was calculated using the GMAV from eight different genera, with *Cancer irratus* representing the lowest ranked GMAV. EPA then divided the FAV by 2[17] to derive the CMC. There was however, insufficient chronic toxicity data available to meet the minimum standards established by the *Guidelines* therefore EPA applied the ACR to the FAV to estimate the final chronic value. Unless there are other data to suggest the FCV is not sufficiently protective, the CCC is set to the FCV. For cyanide, once the ACR for four species was calculated, EPA took the geometric mean of the four freshwater species to derive the final ACR. Next the FAV was divided by the final ACR, to derive the final CCC. For saltwater, the CCC was set equal to the CMC because it was assumed that the acute sensitivity of the rock crab was a better indicator of the chronic sensitivity of the species than would be obtained otherwise. Table 36 contains a summary of the cyanide water quality standards and the top-ranked values used to calculate the CMC and the CCC.

Table 39. Summary of cyanide test results and subsequent water quality criteria[1].

GMAV	Fresh water		Saltwater	
Rank	Genus	GMAV (µg CN/L)	Genus	GMAV (µg CN/L)
4	Lepomis	99.28	Mysidopsis	118.4
3	Perca	92.64	Menidia	59
2	Salvelinus	85.80	Acartia	30
1	Salmo	63.45	Cancer	4.893
FAV (calculated from GMAVs)		62.68		2.030
FAV (SMAV for rainbow trout)		44.73		
CMC		22.36		1.015
ACR		8.568		2

[17] By dividing the FAV by 2, EPA believes that they have derived a CMC concentration "that will not severely adversely affect too many of the organisms (EPA 1985)".

CCC	5.221	1.015

[1]Table adapted from Gensemer et al. 2006; data from EPA 1985.

Acute Toxicity

Knowledge of the acute lethal effects of cyanide on fish has been gained through observations following accidental spills, intentional field application for lake/stream management and controlled laboratory studies. Cyanide is highly toxic with a relatively short half-life. At high levels of exposure, acute toxicity occurs rapidly (Leduc 1984). During intentional field applications exposed fish were observed exhibiting symptoms that include increased ventilation, surfacing, gulping for air, frantic swimming in circles, convulsions and tremors prior to death (Leduc 1984). Laboratory tests under controlled situations have revealed that not all life stages of fish are equally sensitive to acute cyanide exposure, that cyanide toxicity can be modulated by abiotic factors, and that there is a wide range in sensitivity among aquatic organism. For instance, Smith et al. (1978) demonstrated that bluegill, yellow perch, and brook trout juveniles were more sensitive than newly-hatched fry, where, as swim-up fry were the most sensitive fathead minnow life stages.

EPA and the Services conducted an extensive search for data for the consultation, which included a review of studies that had been used in the derivation of the cyanide criteria in 1985 and any new studies that had been conducted since 1984. EPA compiled toxicity data for 83 species of aquatic animals and plants (61 freshwater species and 22 saltwater species) as part of their BE for the cyanide consultation (EPA 2007). Based on this compilation, there appears to be a large range in sensitivity between the most sensitive (rock crab LC_{50} 4.89 µg CN/L) and the least sensitive species tested (river snail LC_{50} 760,000 µg CN/L). Freshwater species represented 9 phyla, 15 classes, 29 orders, 36 families, and 52 genera. Fishes were among the most sensitive freshwater taxa although there was substantial variability in sensitivity. Among the 24 freshwater fish species included in the list there was a 33 fold difference in sensitivity between the most sensitive (rainbow trout, *Oncorhynchus mykiss*, LC_{50} 59 µg CN/L) and the least sensitive (bata, *Labeo bata*, LC_{50} 1970 µg CN/L). The 8 most sensitive fish species belong to 3 different families, Salmonidae (3 species, 3 genera), Percidae (2 species, 1 genera), and Centrarchidae (3 species, 3 genera). Because of the relatively low number of species that have been tested within these families it is difficult to get a sense of the amount of intra-family variability in species sensitivity on the low end of the species sensitivity distribution. By contrast, the family cyprinidae was well represented with 10 different species representing 8 genera. Among those 10 species there is an 18-fold difference in sensitivity between the most sensitive (roach LC_{50} 108 µg CN/L) and the least sensitive (bata, *Labeo bata*, LC_{50} 1970 µg CN/L) species. Because of pronounced intra-family variation it is unlikely that the 8 species within the 3 most sensitive families represent the most sensitive species within those families.

Within the compiled data set, empirical data on the acute effects of cyanide was available for only two biological species under NMFS' jurisdiction—steelhead (representing 11 listed species (DPSs) of *O. mykiss*) and Atlantic salmon (*Salmo salar*)[18]. Consequently, EPA estimated EC_As

[18] Atlantic salmon are jointly managed with FWS. This species is addressed in the FWS' biological opinion on this action.

were calculated using ICE or SSD for six biological fish species under NMFS' jurisdiction, representing 19 listed species (ESUs and DPSs).

Steelhead/Rainbow Trout. Previous work by EPA and others suggest that rainbow trout are the most sensitive freshwater test species to cyanide. That is, the concentration of cyanide that induces mortality is lower than it is for many other species, with warmwater species generally exhibited greater tolerance. We found one additional study on the acute response of rainbow trout to cyanide that has been conducted since EPA calculated the 1984 aquatic life criteria. The study by McGeachy and Leduc was published in 1988 and analyzed the influence of season and exercise on the acute responses of rainbow trout to cyanide. The other studies on the lethal responses of rainbow trout to cyanide were available at the time EPA published their cyanide criteria in 1985. In 1985, EPA chose to use only 4 values for calculating the SMAV for rainbow trout (Table 37). EPA's reasoning for choosing those studies at the time, was because in a comparison of acute toxicity values for fishes they confirmed what Doudoroff (1976) had concluded earlier, that static toxicity tests generally produced higher response values than flow-through tests of equal, fairly prolonged duration (EPA 1985). As a result, they based the SMAV on the results of flow-through tests in which the concentrations were measured (EPA 1985). This comports with direction provided by the *Guidelines* (Stephan et al. 1985) which states:

- For some highly volatile, hydrolyzable, or degradable materials it is probably appropriate to use only results of flow-through tests in which the concentrations of test material in the test solutions were measured often enough using acceptable analytical methods
- For each species for which at least one acute value is available the SMAV should be calculated as the geometric mean of the results of all flow-through tests in which the concentrations of test material were measured.

Thus, the estimated mean acute value influences the estimated assessment effects concentration and the preliminary screen for making Section 7 effects determinations (also the estimated level of protection) under the *Method Manual*. For instance, Table 38 compares acute data from: all referenced studies used by EPA in their 1985 published recommendation for cyanide and used by Gensemer et al. (2007) in their recent review of the cyanide criteria, an approximation of EPA's calculated LC_{50} that they used in the BE to make their effects determination[19], and two alternative data sets to calculate the SMAV and EC_As for steelhead. Using only flow-through test data EPA (1985) and Gensemer et al. (2007) derived SMAVs of 44.73 µg CN/L and 46.53 µg CN/L, respectively. The difference in SMAVs is attributed to Gensemer et al.'s (2007) addition of values from the flow-through tests conducted by McGeachy and Leduc (1988), which were not available at the time the criteria document was published. Because the precise values EPA (2007) used in their BE calculation were not clear to NMFS when there were multiple test values available within a particular study, we used data values that allowed us to approximate their final LC_{50} value. For instance, we are aware EPA used data from Markings et al. (1984) but are not clear what particular values influenced their final LC_{50} calculation.

Marking et al. (1984), Bills et al. (1977), and Skibba were not used in the calculation by EPA (1985) or Gensemer et al. (2007) because the data were derived from static tests, which as noted

[19] EPA calculated a mean LC50 for steelhead (rainbow trout) of 59.22 (µg CN/L). We tried, but could not precisely replicate this value.

earlier tend to produce responses at higher concentrations. Neither EPA (1985) nor Gensemer et al. (2007) stated why the data from Dixon and Sprague (1981) were not used in the calculation. Although these studies were not used in the mean LC_{50} calculation, EPA (1985) and Gensemer et al. (2007) considered the studies as “other data”.

Alternatives 1 and 2 in Table 38, NMFS followed suit with the *Guidelines* and relegated static-test data for later consideration but did not include these data in the LC_{50} calculation. The primary difference between Alternatives 1 and 2, however, was in the test data we included from McGeachy and Leduc (1988). McGeachy and Leduc (1988) compared the toxicity of cyanide under different swimming conditions-- “exercised” versus “non-exercised” conditions. The non-exercised trout were placed in white polyethylene tanks and surrounded with Styrofoam and black plastic to minimize disturbance. It appears that Gensemer et al. (2007) chose to use the data from “non-exercised” fish in their calculation. For comparison, we used only the data from “exercised” trout in Alternative 1 because these fish were kept in more realistic test conditions (i.e., more natural), whereas all the data from McGeachy and Leduc (1988) are used in Alternative 2.

Table 40. Comparison of Toxicity Values To Support Species Mean Acute Value Calculations for Rainbow Trout

Mean LC_{50} Value	LC_{50} Value used to calculate SMAV (µg CN/L)					Reference
	EPA 1985	**Gensemer et al. 2007**	**EPA 2007***	**Alt. 1**	**Alt. 2**	
90			90			Bills et al. 1977
57	57	57	57	57	57	Smith et al. 1978; Broderius and Smith 1979
27	27	27	27	27	27	Kovacs 1979
40	40	40	40	40	40	
65	65	65	65	65	65	
98			98			Dixon and Sprague 1981
98			98			
97						
96						
97						
67						
83						
95						
46			46			Marking et al. 1984
52			52			
54			54			
62			62			
75			75			
55			55	55	55	McGeachy and Leduc 1988
53		53	53		53	
50				50	50	
42		42	42		42	
56				56	56	
53		53	53		53	
56				56	56	
66					66	
97			97			Skibba 1981

64.28	44.73	46.53	59.15	49.28	50.49	SMAV
28.32	19.70	20.50	26.06	21.71	22.24	EC_A using EPA's 2.27 LTAF
49.07	34.15	35.52	45.15	37.62	38.54	EC_A using Gensemer et al. LTAF of 1.31
0.79	1.13	1.09	0.86	1.03	1.01	R

*Values used in the calculation were not provided, and are assumed approximately equivalent to those provided herein.

As noted earlier, this risk paradigm was designed to estimate the relative risk of a chemical, such that the farther away from 1 an R-value, the greater the assurance the assessor would have in their Section 7 effect determination. However, the strength of the EC_A (and the effects determination) depends on the availability of pertinent evidence, and ultimately on the identification, appraisal, assimilation, and interpretation of that evidence. A strict interpretation of the risk paradigm indicates that four of the six scenarios illustrated in Table 37 would warrant a preliminary "likely to adversely affect" determination until additional data is provided that demonstrates otherwise (e.g., "other data" not used in the SMAV calculation, and a closer review of the data used in the LC_{50} calculation). While the risk ratio is merely an indication of potential risk, it is clear that the values chosen to calculate the species' LC_{50} value can influence the preliminary screen risk prediction. Based on our comparison, it also appears that the values EPA used to calculate the SMAV for rainbow trout was conservative, given the larger data set.

Nevertheless, we are concerned that this analytical approach can generate misleading results by ignoring meaningful differences among studies. That is, when the data are normalized first by calculating the geometric mean of the LC_{50}s without regard to the underlying distribution of the data, resolution is lost. In addition to examining the pooled data set to see that it is comprehensive, we must also closely examine the distribution of the underlying data, and differences in test methods (doses, schedules, modes of treatment, etc.) to ensure important differences in data are not drowned in a single estimate generated from a pooled data set (Lau et al. 1998). Uncertainty is incorporated in our analysis when we "focus on the tails of the distribution rather than on the measure of central tendency (the mean or best estimate)…. (Taylor and Wade 2002)." A careful examination of the pooled data set is warranted to ensure we have appropriately incorporated uncertainty and to ensure that the method provides a high degree of conservatism (e.g., errs on the side of the protecting the species when data are not sufficient to reasonably conclude the action is "not likely to adversely affect" the listed species or its critical habitat). When we examine the distribution of the data for rainbow trout we see that the lowest test value presented by Kovacs (1979) approaches the CMC (LC_{50} at 6 ºC = 28 µg CN/L). When we apply EPA's extrapolation factor of 2.27 to the lowest LC_{50} value available for rainbow trout we can estimate of the lowest concentration likely to be lethal to 0 to 10 percent of the population. The resulting LC_{10} for very cold-water conditions (6 ºC) is 12 µg CN/L. That is, when exposed to as little as 12 µg CN/L in cold waters, as much as 10% of the exposed threatened and endangered steelhead may die.

EPA derived the lethality threshold adjustment factor, 2.27, from a combined data set on fresh water and marine fish and invertebrates, a number of chemicals tested, as well as whole effluent data. In comparison, Dwyer et al. (2005) looked at five chemicals and seventeen species, including a few listed species, and also found the average factor to calculate a no- or low-effect concentration varied among pollutants and species (0.50 - 0.66), with the geometric mean for all species as 0.56 (f^{-1} = 1.8). More recently, DeForest et al. (in Gensemer et al. 2007) compiled concentration-response curves for rainbow trout, using data from McGeachy and Leduc (1988), and Kovacs and Leduc (1982), estimated the lethality threshold adjustment factor as 0.76 (f^{-1} = 1.316). Applying the extrapolation factor from Dwyer et al. (2005) results in a low effect concentration of about 16 µg CN/L, and DeForest et al. (in Gensemer et al. 2007) would result in a low effect concentration of 21 µg CN/L. DeForest et al. (in Gensemer et al. 2007) estimated the mean LC_{01}:LC_{50} ratio based on the steepness of the concentration-response curves to produce an estimated effect level lower than the LC_{01} (DeForest et al. in preparation, cited in Gensemer et al. 2007). Using DeForest et al.'s calculated adjustment factor, we would expect that 1% of the sample population would be expected to die as a result of their exposure at that calculated cyanide concentration.

In a separate analysis of the lethality threshold adjustment factor, FWS found EPA's 1978 data set upon which the 2.27 was derived from widely varied data and thus recalculated the adjustment factor as a standardized LC_{50}/LC_{10} using 62 acute exposure-response regression equations for cyanide (Appendix C). FWS' recalculated adjustment factor calculated for rainbow trout was 1.14. Had EPA used this, or any of these revised adjustment factors, more species would have been screened out as not likely to be adversely affected by their exposure to cyanide at the CMC. This further suggests that at least for cyanide, EPA's lethality threshold adjustment factor of 2.27, despite having introduced an additional source of uncertainty into estimates of the EC_A, likely produced preliminary estimates in accordance with the approach in the *Methods Manual* that erred on the side of inclusion rather than screening out species. Again, if we look at the distribution of the acute data for rainbow trout, using Kovacs' (1979) LC_{50} of 28 µg CN/L, which was derived in very cold water temperatures, and apply EPA's adjustment factor of 2.27 then an estimated 1 to 10% of individual steelhead may die when exposed when exposed to as little as 12 µg CN/L in very cold waters (6 ºC or less). Alternatively, if we apply the FWS' recalculated adjustment factor for cyanide to the same data, then the LC_{10} concentration would be above the CMC (at 24.56 µg CN/L).

Other Pacific Salmon Species. Based on species-specific estimates for coho and Chinook salmon and estimates for the genus *Oncorhynchus*, ICE predicts that coho, Chinook, sockeye, and chum salmon are relatively more sensitive than steelhead to cyanide (see Table 38). That is, based on the lower bound of the ICE predicted LC_{50}, coho, Chinook, sockeye, and chum salmon are all "likely to be adversely affected" when exposed to cyanide. Of these four fish within the genus *Oncorhynchus*, EPA's ICE results suggest that coho salmon are the most sensitive Pacific salmon with a predicted acute EC_A of 15.51 µg CN/L, with an estimated LC_{50} of 53.16 µg CN/L. In comparison, also using ICE, DeForest (pers. comm.) estimated the LC_{50} for coho salmon as 41.9 µg CN/L and the LC_{50} for Chinook salmon as 50.9 µg CN/L (Table 5). When we recalculated the EC_A using the divisor 2.27 and the expected LC_{50} value, for both species, the EC_A fell above the CMC using EPA's LC_{50} value, but below the CMC using the estimated LC_{50} calculated by DeForest (pers. comm.). Whereas, DeForest concluded, based use of the LC01 divisor, 1.316, that coho salmon and Chinook salmon were protected by the CMC (both LC01 values are greater than 30 µg CN/L). Based on the work by Gensemer et al. (2007), and DeForest (in Gensemer et al. 2007), the ICE model is likely to conservatively overestimate the sensitivity of most species (i.e., produce lower LC_{50} values than would likely be measured). DeForest (pers. comm.) concluded, based on his analysis of the empirical cyanide SMAVs, that there is an eight percent probability that a fish species would be more sensitive to cyanide than rainbow trout; whereas, if the ICE-estimated LC_{50} values are considered in the SSD, then there is about a 20% probability that a fish species would be more sensitive than rainbow trout. Based on our recalculations and information from DeForest (pers. comm.), and EPA's use of the lower 95 confidence level to calculate the EC_A for these species, it appears that EPA's preliminary effects determination that these species should not be screened out would be conservative (i.e., that is it erred on the side of protecting listed species given the uncertainty in the estimates).

The Influence of Other Data

The preliminary screen in the *Methods Manual* was designed to be a first step for reviewing robust response data, and conclusions based on this screen should be carefully reviewed by rechecking each step. That is, studies that have been dismissed because they do not meet basic requirements for the calculation of EC_A require review as "other data". EPA's *Guidelines* explicitly state that

> *Pertinent information that could not be used in earlier sections might be available concerning adverse effects on aquatic organisms and their uses. The most important of these are data on cumulative and delayed toxicity, flavor impairment, reduction in survival, growth, or reproduction, or any other adverse effect that has been shown to be biologically important. Especially important are data for species for which no other data are available. Data from behavioral, biochemical, physiological, microcosm, and field studies might also be available. Data might be available from tests conducted in unusual dilution water, from chronic tests in which the concentrations were not measured, from tests with previously exposed organism, and from tests on formulated mixtures or emulsifiable concentrates. Such data might affect a criterion if the data were obtained with an important species, the test concentrations were measured, and the endpoint was biologically important (Stephan et al. 1985).*

According to the *Guidelines*, EPA ought to consider "other data" in its decision to recommend a criterion. Unfortunately, it's not apparent that this "other data" influenced EPA's final value for the cyanide CMC (or CCC) in their 1985 cyanide recommendation. Nor is there evidence to suggest that particular states incorporated the "other data" in their final state water quality criteria, such that particular exceptions or special management actions were written into the final adopted water quality standard, when applicable. We were particularly interested in the effects of temperature and dissolved oxygen on EPA's decision to recommend the cyanide criteria because these are two factors known to affect cyanide toxicity, and because studies that have directly explored these relationships with listed resources (steelhead and Atlantic salmon). We explore this "other data" in the following sections.

Table 41. Species Specific Toxicity Estimates (EPA 2007**).

Species*	Saltwater v. Freshwater Exposure	Acute EC_A (µg CN/L)	Chronic EC_A (µg CN/L)	Species Specific Toxicity Data	Estimation Method Used	Taxon Represented by EC_A
Coho salmon	SW (adult & smolt) FW (all life stages)	15.51	3.33	N	ICE	*Oncorhynchus kisutch*
Chinook salmon	SW (adult & smolt) FW (all life stages)	16.26	3.49	N	ICE	*Oncorhynchus tschawytscha*
Chum salmon	SW (adult & smolt) FW (all life stages)	21.41	4.60	N	ICE	*Oncorhynchus* (genus)
Sockeye salmon	SW (adult & smolt) FW (all life stages)	21.41	4.60	N	ICE	*Oncorhynchus* (genus)
Steelhead	SW (adult & smolt) FW (all life stages)	26.08	9.80	Y		
Shortnose sturgeon	SW (adult & juveniles) FW (all life stages)	29.28	6.39	N	SSD	*Actinopterygii* (class)
Green sturgeon	SW (adult & juveniles) FW (all life stages)	29.28	6.39	N	SSD	*Actinopterygii* (class)

*EPA also included Totoaba. This is a foreign-listed species and would not be adversely affected by the proposed action because it is not exposed to United States waters.
**Data are from EPA 2007. Life stages exposed were adjusted to account for all possible life stages that could be exposed in fresh water and salt water.

Table 42. Comparisons of LC50 values for coho and Chinook salmon (µg CN/L)

Species	Estimated Mean LC_{50}	Lower 95% CL	Estimated EC_A using expected LC_{50}	Estimated LC01 (f^{-1}= 1.316)
Coho salmon	53.16*	35.21*	23.42*	40.40
	41.9**	25.2**	18.46	31.84**
Chinook salmon	64.35*	36.91*	28.35*	48.90
	50.9**	24.7**	22.42	38.68**

**Data from DeForest, pers. comm.
*Data from EPA 2007

The Influence of Temperature on Tolerance Limits

As a general matter the tolerance of fish to many pollutants tends to decrease with increases in water temperatures. Studies have demonstrated that the effects of temperature on the toxicity of cyanide can vary with concentration and temperature such that cyanide toxicity increases at high temperatures and at very low temperatures. Studies that have evaluated the effects of cyanide at high temperatures have found that the toxic action of cyanide increases with increasing temperatures, but many of these studies were conducted with extremely high doses of cyanide (see Doudoroff 1976). Early studies indicated that the 72-hour median lethal concentration or tolerance limit increased almost threefold with increased temperatures, when rainbow trout were exposed to test temperatures ranging from 4 to 20 ˚C (Great Britain, Ministry of Technology 1969 in Doudoroff 1976). Unfortunately, it is not clear what cyanide concentrations were used in the Great Britain study (Doudoroff 1976). Later, Kovacs (1979) confirmed that there are significant differences in 96-hour LC_{50} values between 6, 12 and 18 ˚C, such that it took 2.4 times less cyanide to kill 50% of the trout in 96 hours at 6 ˚C than it did at 18 ˚C. One of the primary differences between work by Kovacs (1979) and earlier researchers is the rate and concentration of the doses administered. Kovacs (1979) administered cyanide at slowly lethal concentrations, whereas earlier studies tended to focus on rapidly lethal concentrations, suggesting that the potency of cyanide is both temperature and concentration dependent. Doudoroff (1976) suggested that the lethal response at low temperatures is likely a result of a decrease in the rate of detoxification at lower temperatures, which is affected by the decline in the metabolic rate at lower temperatures. Death at lower temperatures may also be caused by the disruption of cytochrome oxidase activity (Kovacs 1979).

Since steelhead rearing and spawning typically occurs in temperatures ranging from about 4˚C to about 15˚C (Barnhart 1991). Consequently, increased cyanide toxicity at lower temperatures could have serious consequences for steelhead fitness. We chose four river basins that we felt were representative steelhead rivers—one from each of the western states, Oregon, Washington, Idaho, and California, where there are listed steelhead populations—and examined the mean monthly water temperatures for comparison to the low temperatures measured by Kovacs (1979). Figure 7 compares the mean monthly water temperatures to the generalized life history stages of steelhead in the Clearwater River, Idaho. Steelhead in this system compose two-runs, an “A” and “B” run, which are distinguished according to their size and ocean life history. Spawning occurs from mid-April to late June, with “A-run” fish returning after one year in the ocean and “B-run” fish returning after two years in the ocean. Due to the long freshwater rearing period of juvenile steelhead and the long holding period of adults, at least two to three age classes of steelhead can be found in the basin during winter. As illustrated in Figure 7, winter water temperatures are at or below 6˚C for several months each year (about 5 months). Similarly, water temperatures are at or below 6˚C in the Puyallup River in Washington for about three months when adult and juvenile life stages would be in the basin (Figure 8). In the North Umpqua River in Oregon water temperatures are at or below 6˚C for about four months of the year, when additional life stages are present including migrating and spawning adults, eggs, fry, and juvenile fish (Figure 9). In the Klamath River in Oregon/California average water temperatures are below 6˚C for a brief period of time (about a month), but these temperatures occur when adults are migrating and spawning, and juvenile steelhead are rearing (Figure 10). Due to the iteroparous life history of steelhead and the propensity for multiple juvenile age-classes to rear together, these basins would generally have at least two age-classes but may have four or more age-classes in the basin during winter.

We looked but did not find information to suggest that states or EPA would generally modify the cyanide water quality standards to minimize the impacts to salmonids in cold water. We looked for this information particularly in state water quality standards for Idaho, California, Washington and Oregon. Generally, we found that when states modified EPA's nationally recommended criteria they did so to increase the cyanide concentration, not decrease acceptable limits. However, we did not search specific permit conditions to evaluate whether permits were adjusted to account for increased toxicity of cyanide during low temperatures.

All of the Pacific salmonids under NMFS' jurisdiction, green and shortnose sturgeon, and Atlantic salmon are exposed to very cold water temperatures during their life cycle. We would not expect that the general response of increased toxicity at low temperatures is species specific response, but is a generalized physiological response of fish that occupy cold streams. The low acute response of steelhead is likely a reasonable predictor of other Pacific salmonids, but we do not know the lowest response value of sturgeon or Atlantic salmon nor do we have a suitable surrogate to estimate this response. Clearly, more studies are warranted in this area.

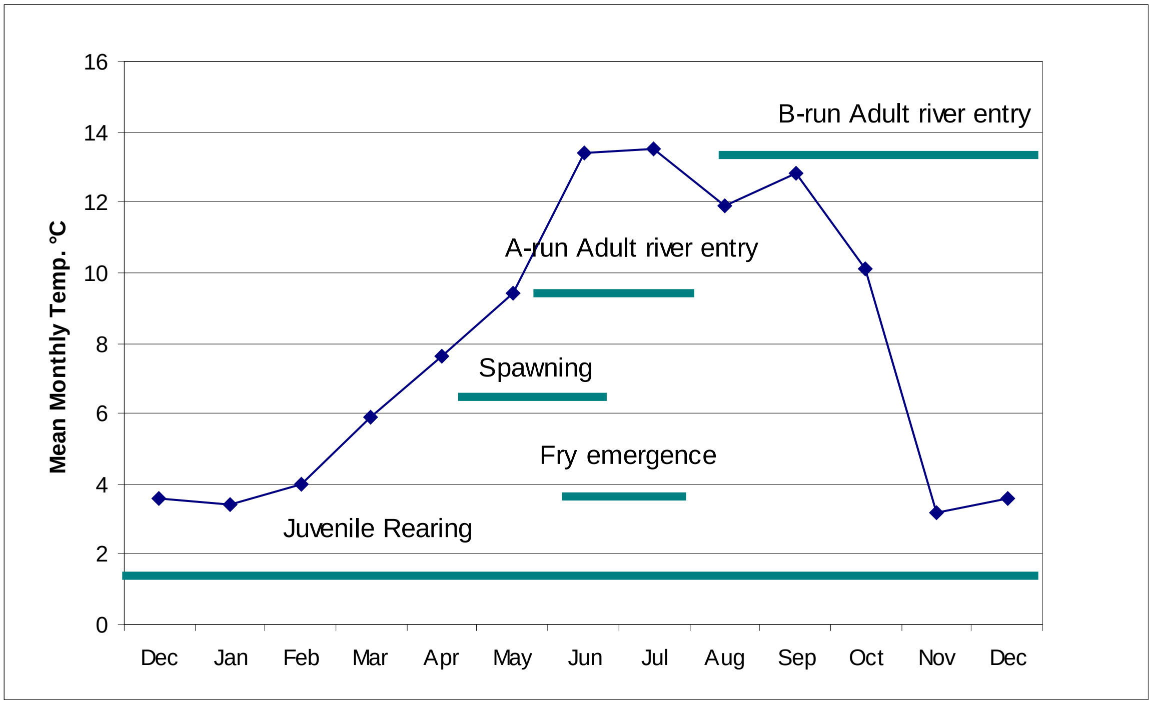

Figure 7. Steelhead life history and mean monthly water temperatures in the Clearwater River, Idaho (Sources: Idaho Department of Fish and Game[20] and USGS Surface-Water Monthly Statistics for the Nation, USGS 13342500 Clearwater River at Spalding ID[21]).

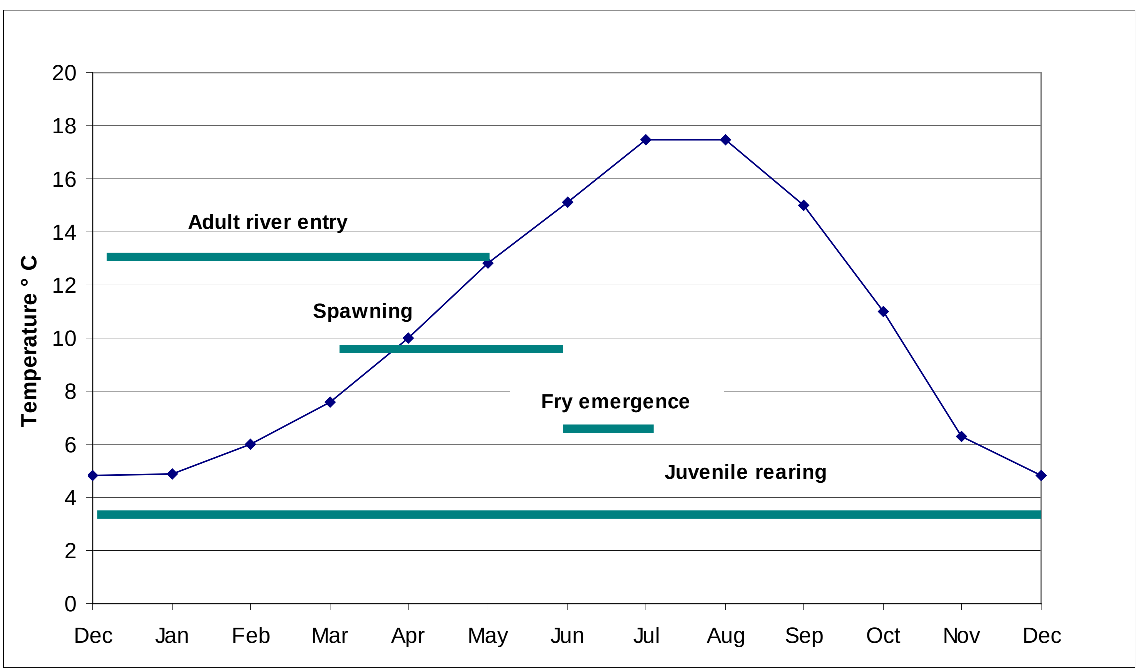

Figure 8. Winter steelhead life-history and mean monthly water temperatures in the Puyallup River Basin, Washington (Ball 2004; and B. Smith, Puyallup Tribe Fisheries, pers. comm., Oct. 14, 2008).

[20] URL:http:fishandgame.adaho.gov/fish/fish_id/steelhead.cfm
[21] URL:http://waterdata.usgs.gov/nwis/monthly?

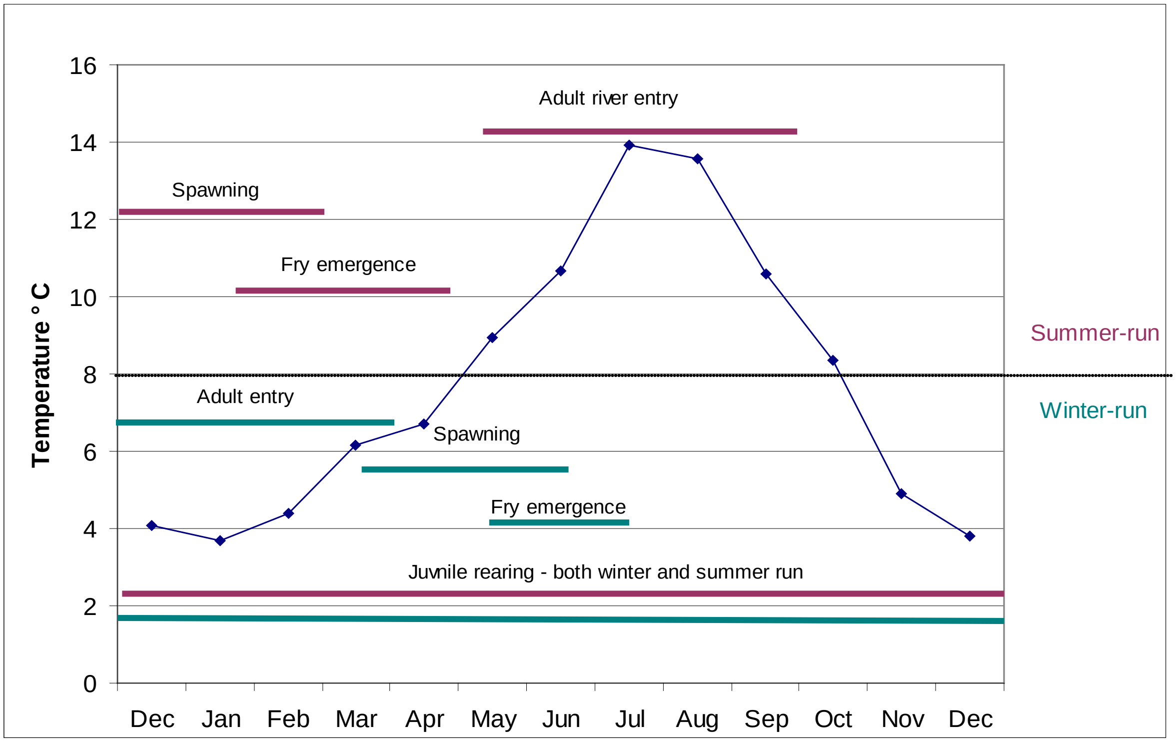

Figure 9. North Umpqua River steelhead life history and average monthly water temperatures (Source: USGS National Water Information System, URL: http://nwis.waterdata.usgs.gov)

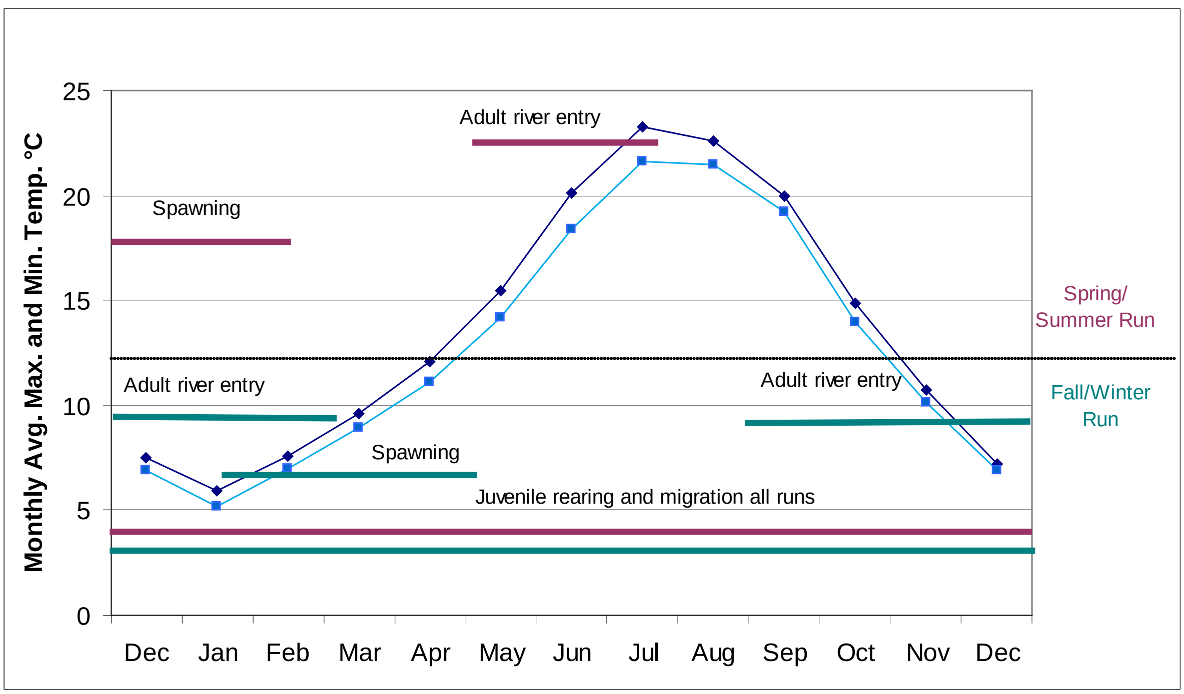

Figure 10. Klamath River steelhead life history and average min. & max. monthly water temperatures (Sources: USFWS 1998 and USGS 2007).

The Influence of Dissolved Oxygen on Tolerance Limits

Generally, in environments where DO is less than optimal fish will compensate for the reduction in DO by increasing gill movement and ventilation volume, in an attempt to maintain adequate oxygen volumes. Cyanide is a powerful asphyxiant, and the addition of cyanide in waters with low DO further stresses fish, reducing the lethal concentration at which survival is typically expected. That cyanide toxicity is influenced by DO is well known (Downing 1954; Smith et al. 1978; Doudoroff 1976; Towill et al. 1978; Alabaster et al. 1983; EPA 1985; Dzombak et al. 2006). Smith et al. (1978) found that a about a 40% reduction in DO levels lead to a 20 to 30% reduction in lethal thresholds for brook trout and rainbow trout. Similarly, Downing (1954) found that rainbow trout survival time increased as DO increased, and the rate of increase did not fall off as DO approach saturation. Alabaster et al. (1983) also demonstrated that the 24-hr LC_{50} value varies with DO concentrations, but not with salinity, and when DO was as low as 3.5 mg/L the LC_{50} value for Atlantic salmon was 24 µg CN/L HCN (well below the acute EC_A reported in Table 4 of the final cyanide BE of 39.65 µg CN/L).

Since the conditions under which this study was conducted are very important to a comprehensive effects analysis, we asked that EPA consider it further in their BE. Unfortunately, EPA responded that (a) it was not considered in the criteria derivation (or was relegated to “other data”), (b) that considerations of toxicity values obtained under a combination of low DO and chemical toxicity is generally not included in their BE because such exposure conditions are not wide-spread across the nation, and (c) confound the toxicity of cyanide alone to which the criteria apply. The first part of the statement, that the data in Alabaster et al. (1983) was not considered in the derivation of the cyanide criteria is interesting, but not necessarily pertinent for the purposes of a Section 7 consultation. First, the criteria were derived without consideration of listed species, but more importantly the question of risk depends upon the environmental decision-making context. That is, Section 7 is first concerned with the risk an action poses individuals of a listed species –this is the level at which a federal agency makes their effect determination. Not until individual effects can be dismissed as insignificant or discountable, would we conclude that an action is “not likely to adversely effect” listed species. The CWA decision-making process begins by focusing not on the individual, but whether community level effects are likely. The effect threshold is considerably different. By the time community level effects are measurable (and noticed) the hazard’s risk may pose substantial impacts to small populations. If EPA meant to imply that because a study was not used to derive a particular criterion that it did not warrant consideration in their Section 7 effects analysis, then EPA is missing the point of Section 7 under the ESA. The relevant inquiry is not whether such a study was used by EPA in their 1985 criteria decision, but whether there is information to suggest that environmental conditions to which listed species are exposed may influence the toxicity of the chemical under review—in this case cyanide. Cyanide can be more toxic to freshwater fish at low dissolved oxygen concentrations.

According to our assessment of water quality conditions across the nation, low DO conditions are a problem in many basins at various times of the year (see the Environmental Baseline section of this opinion, also see EPA 2006). The susceptibility of fish to cyanide at low DO may be correlated with the rate of breathing. That is, as a general matter the rate of gill movement increases with decreasing DO, causing the fish to pump additional water through the gills to obtain more oxygen. When cyanide is present in the water column, this may increase the rate at which tissue that is more susceptible to absorption is exposed to cyanide. Although EPA did not consider the relationship between DO and cyanide to be one that should drive the nationally recommended criteria, there is sufficient information indicating the toxicity threshold for salmonids is reduced in low DO conditions such that additional studies are warranted to make definitive conclusions regarding the effect on fish and whether the criteria would fully protect salmonids at the local or site-specific scale.

Effects of Mixtures

Relatively few studies have been performed to measure the effects of free cyanide in combination with other contaminants. Concurrent exposure to cyanide and ammonia produced greater than additive effects to acute lethality in rainbow trout, salmon, and chub (Smith et al, 1979; Alabaster et al., 1983; and Douderoff 1976), and to chronic sublethal effects to growth in rainbow trout (Smith et al 1979). In rainbow trout and salmon, effects to acute lethality were 1.2 and 1.63 times greater than would be expected by additivity. Concurrent exposure to cyanide and zinc also resulted in synergistic effects to acute lethality in fathead minnows, where toxicity was 1.4 times that predicted by additivity (Smith et al 1979). Though we are unable to quantify the effect of these synergistic mechanisms for this analysis, they should be considered when assessing effects of cyanide to aquatic organisms in waterways with elevated concentrations of ammonia and zinc.

Chronic Toxicity

Chronic cyanide toxicity tests have been conducted with relatively few fish species, however, the available data indicate that cyanide not only reduces survival but also affects reproduction, weight gain, growth and development, swimming performance, condition, and development. Few studies have examined the sublethal responses at cyanide concentrations below the freshwater CCC (i.e., <5 µg CN/L) and many have evaluated the effect of concentrations double that of the CCC, making it difficult to evaluate the effect of exposing individuals at the CCC.

Dixon and Leduc (1981) also found evidence of liver necrosis in rainbow trout from low-level exposures of cyanide; however the lowest concentration that they examined was 10 µg HCN/L (~9.8 µg CN/L). In calculating the chronic EC_A value for rainbow trout, it appears that EPA used the reported NOEC from only one study, Dixon and Leduc (1981). In Table 1 of the final cyanide BE, EPA reports a chronic EC_A value for rainbow trout of 9.8µg/L. However, since Dixon and Leduc (1981) did not evaluate rainbow trout response to cyanide concentrations below 9.8 µg/L, it is equivocal to equate this value to a NOEC for the species since adverse effects could not be distinguished at concentrations below this value.

Given the available data reproduction appears to be one of the most sensitive (and most studied) endpoints. Full and partial life cycle tests with fathead minnow and brook trout have shown that fish exposed to sublethal concentrations of cyanide spawned fewer eggs than non-exposed fish (Koenst et al. 1977; Lind et al. 1977). Fecundity was reduced by 57.8% and 46.9% (compared to controls) in female fathead minnows exposed to cyanide at 19.6 µg HCN/L (the LOEC) and 12.9 µg HCN/L (the NOEC), respectively. Similarly, the mean number of eggs spawned by brook trout was reduced by 53.3% at 11.2 µg HCN/L and by 17.7% at 5.7 µg HCN/L. Koenst et al.

(1977) exposed brook trout to nominal HCN concentrations between 5.7 and 77 µg HCN/L, and found that at the mean number of eggs spawned per female decreased with increasing HCN concentrations above 5.7 µg HCN/L. Using a mean temperature of 13.5 ˚C, to convert to CN^- results in a NOEC value is 5.6 µg CN/L, just above the CCC. In the same study, Koenst et al. (1977) found that exposure to 5.5 HCN/L (5.4 µg CN/L) reduced the length of brook trout at hatching and the percentage of eggs that hatched.

Kimball et al. (1978) studied the chronic toxicity of HCN to bluegill and found that bluegill ceased spawning at 5 µg HCN/L (~4.8 µg CN/L). Of eight tests with different concentrations, ranging from 5 to 80.0 µg HCN/L, no spawning was recorded in seven of the tests. Interestingly, at the highest concentration 80.0 µg HCN/L, one female survived and managed to spawn, although her egg production was markedly reduced in comparison to controls. Although the single spawning is difficult to explain, the fact that spawning was completely inhibited in 42 of 43 cyanide-exposed females suggests that bluegill may be particularly sensitive to cyanide at low levels. Results of the tests conducted by Kimball et al. (1978) suggest there was a 3% probability that a female would spawn at $\geq$4.8 µg CN/L, and since levels less than 5.2 µg HCN/L were not tested the only we can only safely conclude that this is a LOEC and that the NOEC lies at a threshold concentration below 5.2 µg HCN/L. Considering the overwhelming evidence of an adverse effect, it is surprising that additional studies on the effects of cyanide on bluegill reproduction have not been conducted over the past 30 years. Cheng and Ruby (1981) studied the effects of pulsed exposures of cyanide on flagfish reproduction. Unlike the studies describe above, where fish were exposed over an extended period of time to a constant concentration, flagfish were exposed to sublethal concentrations of cyanide for 5 day pulses. Flagfish exposed to cyanide (65 µg/L) for 5 days following fertilization, i.e. as eggs, and then reared to maturity in clean water, spawned 25.6% fewer eggs than flagfish that had not been exposed. In another experiment by the same authors, flagfish that received a second 5-day pulse of cyanide as juveniles had an even greater reduction (39.3%) in number of eggs spawned. These studies demonstrate that cyanide can affect an apical reproductive endpoint in fish.

The mechanism by which cyanide induces these reproductive effects is not fully understood, however, key physiological, biochemical, histological (morphological), and endocrine functions known to be involved in sexual maturation are affected by cyanide. For instance, Lesniak and Ruby (1982) reported abnormal oocyte development in sexually maturing female rainbow trout exposed to cyanide (10 and 20 µg HCN/L) for 20 days. Ovaries from cyanide-exposed fish contained fewer mature oocytes, exhibited altered patterns of secondary yolk deposition (in developing oocytes), had nearly twice the frequency of atresia (oocyte resorption), and had an overall reduction in the number of viable eggs. Ruby et al. (1986) reported that vitellogenic female rainbow trout exposed for 12 days to 10 µg HCN/L had lower levels of plasma vitellogenin and a lower gonadosomatic index (GSI) compared to controls. In two similar studies, oocyte diameter (an indicator of gonadal growth and development) was reduced in sexually maturing female rainbow trout exposed for 12 days to 10 µg HCN/L (Ruby et al. 1993a, Szabo et al. 1991). Reduced oocyte diameter was accompanied by reductions in plasma vitellogenin, 17β-estradiol (E2), and GSI (Ruby et al. 1993a), as well as increased whole brain dopamine levels (Szabo et al. 1991).

Dopamine has an inhibitory effect on gonadotropin-releasing hormone (GnRH) neurons in some fish species and it is GnRH which stimulates the release of gonadotropins (GtH I and GtH II) from the pituitary (Patino 1997; Saligaut et al. 1999). GtH I and GtH II are believed to function similar to follicle-stimulating hormone and luteinizing hormone, respectively, in tetrapods (Patino 1997). In female fish, GtH I acts on target cells in the gonad, stimulating E2 synthesis. E2 induces vitellogenin synthesis in the liver. Vitellogenin is the egg yolk precursor in fish which is produced by the liver, transported via blood, taken up by ovaries, and incorporated into developing oocytes. GtH II also acts on the gonad by inducing the synthesis of maturation-inducing steroid (MIS). MIS induces oocyte maturational competence and ovulation (Park et al. 2007; Patino 1997). The control exerted by dopamine over gonadal maturation has been recognized by fish culturists, who have been successful in treating captive-reared fish with anti-dopaminergic drugs (which block dopamine receptors), such as pimozide and domperidone, to induce ovulation (Jensen 1993; Park et al. 2007; Patino 1997; Szabo et al. 2002). Thus, oocyte development, maturation and ovulation are under the control of gonadotropins and E2 which in turn, are modulated in part by GnRH and dopamine. This interaction between the neuroendocrine system and reproductive organs is referred to as the hypothalamus-pituitary-gonadal (HPG) axis (IPCS 2002).

Cyanide has also been shown to affect male reproductive processes. Exposure of male rainbow trout to cyanide (10 and 30 µg HCN/L) for 18 days disrupted spermatogenesis as evidenced by a reduction in the number of dividing spermatogonia and a blockage of mitotic progress (Ruby et al. 1979). Exposure of rainbow trout for 12 days to 10 µg HCN/L resulted in higher numbers of spermatogonial cysts in testes of male trout as well as higher levels of whole brain dopamine (Szabo et al. 1991). Similar results were reported by Ruby et al. (1993) where the number of spermatocytes decreased and the number of spermatocyte precursors (spermatogonial cysts) increased in two-year-old sexually maturing rainbow trout after 12 day exposure to 10 µg HCN/L. There are indications that the transformation of spermatogonial cysts to spermatocytes is hormonally regulated through GtH along the HPG axis and that, within the pituitary, GtH is released from type I granular basophils (Ruby et al. 1993). Histological examination of pituitary glands from cyanide-exposed fish showed a reduction in the number of type I granular basophils. The authors suggested that elevated levels of brain dopamine may be responsible for the selective loss of type I granular basophils and subsequent alteration of spermatocyte formation.

Ruby et al. (1979, 1993) and Szabo et al. (1991) hypothesized that cyanide acts through the HPG axis to affect reproduction in fish. Their studies (described above) demonstrated (1) that cyanide caused an increase in brain dopamine levels, consistent with neuronal effects observed on mammals, (2) that levels of reproductive hormones (E2) and egg-yolk precursors (vitellogenin) were altered following exposure to cyanide, (3) the selective loss of putative GtH releasing pituitary cells (type I granular basophils) and (4) retarded gonad development in cyanide-exposed male and female rainbow trout. Taken together, these results appear to be consistent with HPG axis involvement. In addition, the authors found that these effects occurred following relatively short, sublethal exposures to cyanide (12 – 18 days). Whether these effects would result in the same type of reduced fecundity and spawning, as was observed in cyanide-exposed female fathead minnow (Lind et al. 1977), bluegill (Kimball et al. 1978), and brook trout (Koenst et al. 1977), was not addressed in the rainbow trout studies because they were terminated before the fish reached full sexual maturity, however, it does seem likely. Results from Cheng and Ruby (1981) indicate that continuous exposure to cyanide through the spawning period may not be necessary to affect fecundity. Short-term, pulsed exposures of cyanide to flagfish were sufficient to induce later effects on the number of eggs spawned, and exposed fish did not appear to recover once the exposure had ceased. Even exposure of eggs, one of the most tolerant life stages in terms of acute toxicity (Smith et al. 1979), resulted in latent effects on fecundity once embryos hatched and survived to maturity. Interestingly, it is during early developmental stages that the HPG endocrine axis is set up and feedback sensitivity of the hypothalamus and pituitary gonadotropes to gonadal steroids is established (IPCS 2002). Although Cheng and Ruby (1981) did not measure specific indicators of endocrine axis function, they did find that the pituitary gland of cyanide-exposed flagfish embryos was significantly smaller than the pituitaries from control fish. It would appear that cyanide, like many EDCs (endocrine disrupting compounds, IPCS 2002), may affect the "set up" of the HPG axis and that these early developmental effects may have long term consequences on reproduction.

Chronic exposure of eggs and larvae to cyanide can result in reduced embryo/larvae survival and altered development. Leduc (1978) exposed newly fertilized Atlantic salmon eggs to cyanide (10 - 100 µg HCN/L) and observed teratogenesis, as well as, delayed hatching and reduced hatching success at higher concentrations. There was a dose-dependent increase in the frequency of abnormal fry, ranging from 5.8% to 18.5%. Abnormalities included malformed and/or absence of eyes, defects in the mouth and vertebral column and yolk-sac dropsy (*Hydrocoele embryonalis*, also known as blue sac disease). Similar eye abnormalities were reported by Cheng and Ruby (1981) in flagfish larvae exposed, as eggs, to cyanide (65, 75, 87, 150 µg HCN/L). Hatching success was also reduced and time to hatch was delayed in all cyanide treatments. In a 28-day embryo/juvenile toxicity test, sheepshead minnow survival was significantly reduced in all treatments >29 µg HCN/L (Schimmel 1981). The author noted there was considerable embryonic mortality and that there was no larval mortality during the last two weeks of exposure, indicating a greater sensitivity during early development. Kimball et al. (1978) exposed bluegill eggs and larvae to cyanide (4.8 – 82.1 µg/HCN/L) and reported that most deaths occurred within the first 30 days after hatching. Survival was reduced in all cyanide treatments and the effects were statistically significant at cyanide concentrations >9.1 µg HCN/L.

As previously mentioned, cyanide effects oxidative metabolism, energy production, and thyroid function; all are important for normal growth and performance. Therefore, it is not surprising that sublethal exposure of fish to cyanide has been shown to impact growth, condition and swimming performance. There is also evidence that the effect of cyanide on these physiological endpoints can be modulated by other factors such as diet/ration and temperature. When cichlids (*Cichlasoma bimaculatum*) were fed unlimited rations and exposed to cyanide for 24 days, those fish exposed to lower concentrations of cyanide (< 0.06 µg HCN/L) were larger than controls, where as, at higher concentrations weight gain was depressed (Leduc 1984). The increased weight gain in the low-dose treatments was attributed to higher food consumption, which was allowed to occur because ration was not restricted. Low-dose stimulation is a common effect across a broad range of chemical and non-chemical stressors (Calabrese 2008). Dixon and Leduc (1981) held juvenile rainbow trout on restricted rations and exposed them to cyanide (10, 20, 30 µg HCN/L) for 18 days and observed significantly reduced weight gain in all treatments compared to controls. The effect was characterized by an initial decrease in specific growth during the first 9 days followed by a significant increase from day 9 through 18. The growth surge during the latter half of the exposure period was not sufficient to offset early reductions.

Cyanide-affected fish were in poorer condition, as indicated by lower fat content, and had higher respiration rates for several days post exposure. In addition, fish in all cyanide treatments exhibited degenerative necrosis of hepatocytes, i.e. liver tissue damage, which increased in severity with the level cyanide exposure. Kovacs (1979) held juvenile rainbow trout on restricted rations and exposed them to cyanide for 20 days. The results were similar to those of Dixon and Leduc (1981). Cyanide reduced the mean specific growth rate and affected-fish gained less fat during the exposure period. Kovacs (1979) also examined the effects of temperature on rainbow trout growth and sensitivity to cyanide, and found that the growth rate of rainbow trout was inversely related to holding temperature (6, 12 and 18˚C), as would be expected, and that trout held at colder temperatures were more sensitive to cyanide. The NOECs for mean specific growth rate were 5, 20, and 30 µg HCN/L for trout held at 6, 12, and 18˚C, respectively. Based on the exposure response curves the author estimated thresholds for effects on growth to be <5 µg HCN/L at 6˚C (<4.9 µg CN/L, just below the freshwater CCC), 10 µg HCN/L at 12˚C, and 30 µg HCN/L at 30˚C. In the same study, swimming performance was found to be affected by cyanide and the effect was also temperature-sensitive. Fish from the growth study were placed in swimming chambers and tested for swimming stamina. Among non-exposed trout, swimming stamina, measured as distance travelled (meters), decreased with decreasing temperature, i.e. fish held a 6˚C travelled a shorter distance than fish held at 18˚C. Cyanide-exposed fish had reduced swimming stamina compared to non-exposed fish and the effect was more severe at colder temperatures. Based on the exposure-response regression equations reported by Kovacs (1979) the predicted reduction in swimming stamina (compared to controls) for fish exposed to cyanide at the chronic water quality criterion (5.2 µg CN/L) would be 52% at 6˚C, 20% at 12˚C, and 3% at 18˚C. Several other authors have studied swimming performance as well. Leduc (1966) studied the effect of sublethal concentrations of cyanide on cichlids and coho salmon; the lowest concentration examined was 7 µg CN/L. At 7 and 8 µg CN/L cichlids exhibited reduced swimming speeds, similar to fish exposed to higher concentrations (30 µg CN/L; Leduc 1966). Neil (1957 in Kovacs 1979, Koenst et al. 1977) also showed that cyanide concentrations as low as 10 µg CN/L reduced the swimming stamina of brook trout by 98%. Similarly Broderius (1970) and Speyer (1975) observed reduced the swimming ability of coho salmon and rainbow trout at concentrations of 10 and 20 µg HCN/L. Thus, chronic exposure of fish to cyanide at sublethal concentrations, can affect growth, condition and swimming performance, and factors such as temperature and diet/ration can modulate cyanide toxicity. Neither fat synthesis nor swimming performance, however, are endpoints that EPA would typically use to establish water quality criteria, yet the two endpoints can significantly influence an individual’s fitness. Fat is an indicator of growth, and is important during migration and reproduction as an energy reserve. Poor swimming performance can reduce ability to escape predators, maintain stream position, migratory performance. That adverse effects occur below the CCC appears unequivocal; a question that merits further investigation is just how far below the CCC is the threshold response for most species?

Chronic Effects Estimation

Ideally, we would use concentration (dose)-response data to build predictive models of the potential sublethal effects of cyanide. Unfortunately, such data do not exist for cyanide or listed species. As recently reviewed by Gensemer et al. (2007), the current inventory of concentration-response data from chronic toxicity testing with cyanide consists of four datasets; one each for reproductive endpoints among fathead minnow (*Pimephales promelas*; Lind et al. 1977) and brook trout (*Salvelinus fontinalis*; Koenst et al. 1977); and for juvenile survivorship among bluegill (*Lepomis macrochirus*; Kimball et al. 1978) and sheepshead minnow (*Cyprinodon variegates*; Schimmel et al. 1981). Upon closer inspection, Gensemer et al. (2007) found the dataset for sheepshead minnow to be insufficient for meaningful predictive modeling and we agree with that conclusion. Thus, we are left with three datasets as the best available scientific basis for estimating toxic effects at the chronic criterion value of 5.2 µg CN/L. In addition to our three useable concentration-response datasets we also possess estimates of LC_{50} values for our listed species as per the procedures established in the *Methods Manual*.

To estimate the chronic effects of the proposed action on listed species, we transformed our three concentration-response data sets into the most precise predictive concentration-response models that the data can support and then used these models to predict the response of chronic toxicity test species to the CCC for cyanide. We assume that the predicted response of a listed fish species to the CCC is the same as the response observed for a chronic toxicity test species at an adjusted chronic CN exposure level based on the ratio of their respective LC_{50} values (example below). This approach is based on two simplifying assumptions:

1. That relative differences in sensitivity to chronic CN exposures between our listed evaluation species and our chronic toxicity test species (i.e., fathead minnow, brook trout, and bluegill) are approximated by the ratio of their respective LC_{50} values, and

2. The slopes of the concentration-response curves are also approximately comparable between our listed evaluation species and our chronic toxicity test species.

These assumptions create a clearly defined basis for a default hypothesis that allows us to proceed within the constraints of minimal data until such time as more data become available. As more data become available appropriate modification (or validation) of our default approach is necessary.

To provide an illustrative example of the outcome from our simplifying assumptions, suppose that one chronic toxicity test species is predicted to exhibit a 20% adverse effect from 5.2 µg CN/L. If a listed species happens to have an estimated LC_{50} value equal to that of the chronic toxicity test species, then a 20% adverse effect would also be predicted for the listed species. If the ratio of LC_{50} values was 1.5 (rather than 1.0) in the direction of greater sensitivity for the listed species than the chronic toxicity test species, then the predicted response at the concentration of interest of 5.2 µg/L for the listed species would be the same as the response observed for the chronic toxicity test species at a CN concentration 1.5 times 5.2 µg/L, i.e., at 7.8 µg/L. We refer to such surrogate currency equivalents for our listed species as $SSEC_x$ values (or sometimes shortened to SS_x). In this example, the predicted adverse effect for chronic toxicity test species at the $SSEC_x$ of 7.8 µg/L would be our surrogate currency predicted effect for the listed species at 5.2 µg CN/L (from that one of three prediction models) for the purposes of this Opinion. A more detailed derivation and explanation of the $SSEC_x$ concept is provided in Appendix D.

Because groups of taxonomically related listed species were assigned identical LC_{50} values from the same ICE model, there are only 17 $SSEC_x$ values that need to be evaluated for any given (chronic toxicity test species) prediction model, but they are different for each prediction model (3 x 17 = 51 total $SSEC_x$ values of interest). For the prediction model based on fathead minnow chronic toxicity data the $SSEC_x$ values range from 6.7 to 45.8 µg CN/L (Table 40). As indicated by the entire range of $SSEC_x$ values being greater than 5.2 µg CN/L, all listed evaluation species have LC_{50} values that are more sensitive to cyanide than the fathead minnow LC_{50} value. For the prediction model based on brook trout chronic toxicity data the $SSEC_x$ values range from 4.2 to 28.4 µg CN/L (Table 40). For the prediction model based on bluegill chronic toxicity data the $SSEC_x$ values range from 6.1 to 41.7 µg CN/L (Table 40). Those $SSEC_x$ ranges define for each prediction model the range of cyanide concentrations over which model fit will be of most relevance to this Opinion. Detailed $SSEC_x$ results and the origins of the LC_{50} values used to calculate the $SSEC_x$ values are presented in Table 40 and Appendix D.

Table 43. Surrogate currency equivalents ($SSEC_x$[1]) for each LC_{50} surrogate taxon/chronic toxicity test species combination

Surrogate taxa used to estimate listed species (LS) LC_{50}	$LSEC_x$ (µg CN/L)	LS LC_{50} (µg CN/L)	Effects on Fecundity		Effects on Early Life Stage Survival
			Fathead Minnow SS LC_{50}=138.4 (µg CN/L)	Brook Trout SS LC_{50}=85.7 (µg CN/L)	Bluegill SS LC_{50}=126.1 (µg CN/L)
			$SSEC_X$ (µg CN/L)	$SSEC_X$ (µg CN/L)	$SSEC_X$ (µg CN/L)
Actinopterygii (class)	5.2	66.5[2]	10.8	6.7	9.9
Cypriniformes (order)	5.2	84.55[2]	8.5	5.3	7.8
Family Catostomidae					
Xyrauchen texanus (species)	5.2	83.8[3]	8.6	5.3	7.8
Family Cyprinidae					
Cyprinella monacha (species)	5.2	36.4[3]	19.8	12.2	18.0
Gila elegans (species)	5.2	50.9[3]	14.1	8.8	12.9
Notropis mekistocholas (species)	5.2	48.5[3]	14.8	9.2	13.5
Ptychocheilus lucius (species)	5.2	43.5[3]	16.6	10.3	15.1
Perciformes (order)	5.2	90.8[2]	7.9	4.9	7.2
Percidae (family)	5.2	42.3[3]	17.0	10.5	15.5
Etheostoma (genus)	5.2	40.0[3]	18.0	11.1	16.4
Etheostoma fonticola (species)	5.2	21.5[3]	33.4	20.7	30.5
Order Salmoniformes, Family Salmonidae					
Oncorhynchus (genus)	5.2	47.0[3]	15.3	9.5	13.9

Oncorhynchus apache (species)	5.2	16.5[3]	43.6	27.0	39.7
Oncorhynchus tschawytscha (species)	5.2	32.0[3]	22.5	13.9	20.5
Oncorhynchus kisutch (species)	5.2	32.4[3]	22.2	13.8	20.3
Oncorhynchus clarki henshawi (species)	5.2	22.8[3]	31.5	19.5	28.7
Salvelinus (genus)	5.2	15.7[3]	45.8	28.4	41.7
Salmo salar (species)	5.2	90[4]	8	5	7.3

[1]$SSEC_x$ values were calculated using equation 5 in Appendix D. Surrogate taxa were used to estimate LC_{50} values for listed species.
[2] LC_{50} based on 5th percentile estimate from species sensitivity distribution (SSD), Table 2 – Cyanide BE.
[3]LC_{50} estimate based on lower bound of the 95% CI from ICE model
[4] LC_{50} based on measured value from the Cyanide BE

Prediction models. We applied statistical regression techniques to model, or "fit", the relationship between cyanide concentrations and toxic effects based on data for our chronic toxicity test species. For nuances of statistical regression specific to toxicological applications we relied substantively on two recent technical guidance documents: (1) Environment Canada (2005: "Guidance Document on Statistical Methods for Environmental Toxicity Tests"), and (2) OECD (2006: "Current Approaches in the Statistical Analysis of Ecotoxicity Data: A Guidance to Application"). We also reviewed other relevant guidance such as that provided in the documentation for EPA's Toxicity Relationship Analysis Program (TRAP) (EPA 2002) and in discipline-specific statistical textbooks such as Gad and Weil (1988) and Sparks (2000).

As noted by Environment Canada (2005) an important principle of regression techniques is to keep the model simple, if that can reasonably be done. We have further incentive to follow that principle because we have a strong interest in evaluating the uncertainty (confidence) associated with point estimates and therefore an interest in avoiding what Environment Canada (2005) noted as the "...obstacle of calculating confidence intervals around nonlinear regression estimates..." Throughout this exercise we have been mindful that because our models are not based on biological or chemical mechanisms of action, but are purely statistical constructs, they have no biological meaning. A statistical concentration-response model only serves to smooth the observed concentration-response, to estimate effect concentrations by interpolating between treatment concentrations, and to provide a tool for assessing confidence intervals. Therefore the choice of model is to some extent arbitrary (OECD 2006). That being noted, we constructed models that conformed to the data we are working with and with statistical standard practices (such as data transformations). The degree of model fit achieved is an artifact of those specific decisions not the result of *post hoc* "model shopping" (EPA 2002).

Generic concentration-response relationship. Figure 11 illustrates a generic concentration-response relationship which typically takes on a sigmoidal form due to threshold effects on the low concentration end of the x-axis and to asymptotic effects at the high concentration end of the x-axis.

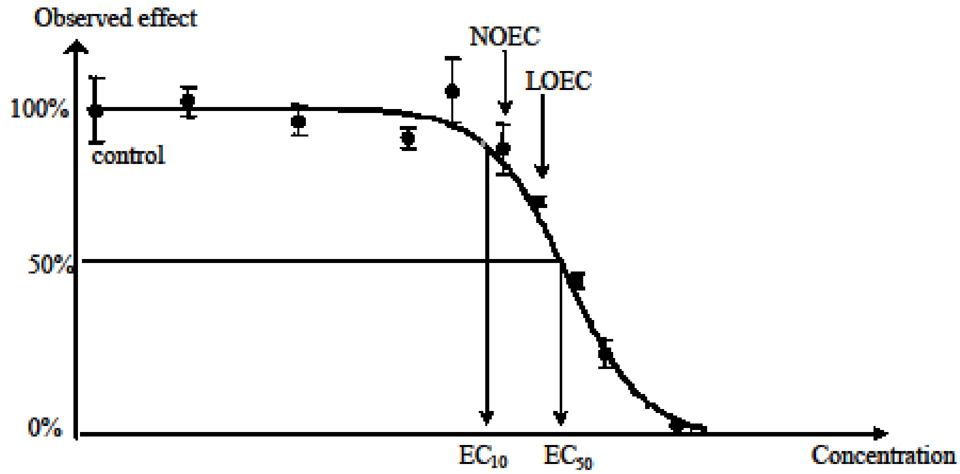

Figure 11. Generalized concentration-response relationship adapted from OECD (2006:Figure 3.2). Note that the illustrated curve is a plot fitted to a real dataset, thus the identification of NOEC and LOEC concentrations. For the purposes of this figure think of the y-axis as a positive attribute that becomes diminished by toxicity, such as percent survivorship.

Note that the superimposed straight line in Figure 11 represents the region of concentrations that induce an intermediate toxic response that are well approximated by a linear fit. This "linear region" is strongest within one probit (also known as "normal equivalent deviate") either side of the median response concentration (EC_{50}), or roughly for concentrations that induce 16 to 84% response (Environment Canada 2005). The narrow ranges of $SSEC_x$ values that we need to evaluate can be expected to overwhelmingly fall within those boundaries as a result of the methods EPA used to set the chronic criterion at 5.2 µg CN/L (see the next section titled, *Derivation of the Criterion Continuous Concentration*). Our approach is conceptually similar to the TRAP program's Piecewise Linear regression option (EPA 2002). Even with regard to the nonlinear regression options in TRAP, EPA (2002) provides a recommendation for segmented analysis when there is a focal region (or subset) of test concentrations of particular concern:

> *Within the limitations of this program, one useful approach can be to exclude (censor) high effects data from the analysis if (a) only low levels of effect are of interest and (b) there are sufficient low-to-moderate effects data to support a good analysis.*

Prediction model based on fathead minnow dataset. Lind et al. (1977) examined fathead minnow fecundity (number of eggs per spawn) and egg hatchability in relation to a series of cyanide treatments (concentrations). The experimental structure and fecundity results are summarized in Table 41. There were five control replicates, and two replicates each for ten exposure concentrations. The response data are reasonably monotonic, especially within the intermediate response range covered by the lowest six treatments. Those treatments range (on a free cyanide basis) from 6 to 45.6 ug/L CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 4).

Table 44. Egg production of adult fathead minnows exposed for 256 days (from larvae through adult) to various concentrations of cyanide (from Lind et al. 1977; Table II).

Treatment HCN (µg/L)	Mean HCN (µg/L)	Free cyanide as CN (µg/L)	Mean eggs per female	Mean eggs per female per treatment	Reduction in the number of eggs per female - percent of control
Control			2530	3476	
Control			4483		
Control			3990		
Control			2718		
Control			3660		
5.7	5.8	6.0	1886	2512	27.7
5.9			3138		
13.0	12.9	13.3[N]	1701	1845	46.9
12.7			1989		
19.6	19.6	20.2[L]	1694	1468	57.8
19.6			1241		
27.1	27.3	28.2	1093	1367	60.7
27.5			1640		
36.0	35.8	36.9	678	1010	71.0
35.6			1341		
43.7	44.2	45.6	2054	1124	67.7
44.7			194		
62.5	63.5	65.6	74	72	97.9
64.5			70		
73.1	72.8	75.1	573	319	90.8
72.4			64		
81.5	80.7	83.3	266	243	93.0
79.8			219		
96.1	100.8	103.9	0	0	100.0
105.4			0		

[N]NOEC
[L]LOEC

To “build” our prediction model we transformed both the concentration data and the fecundity data for *a priori* reasons. We log-transform the concentration data for two reasons: (1) statistically, toxicological tolerance distributions have long been confirmed as log-normal (OECD 2006), and (2) biologically, organisms experience toxicants on a log scale. Toxicological custom is to use log base-10 for the log transformations of test concentrations (Environment Canada 2005). Count data, such as “number of eggs per spawn” typically conform to a Poisson distribution rather than a normal distribution. To normalize such data for regression analysis a square-root transformation is recommended (EPA 2002). Thus, we use the square-root transformed response data for statistical analysis and then back-transform for reporting results. This transform does not change the model, but affects what the best parameter estimates and confidence limits are (EPA 2002). Thus, our model of choice is a log-square root linear regression over our focal segment (subset) of test concentrations.

In agreement with Gensemer et al.’s (2007) treatment of the same dataset, we collapse the fecundity and egg hatchability endpoints into a single endpoint, “eggs hatched per spawn” which is the product of (eggs per spawn) x (egg hatchability) at each treatment concentration. We went a step further than Gensemer et al. (2007) and additionally apply a data smoothing procedure to meet the assumption of monotonicity of response inherent in a linear regression. We did that by calculating three-point moving averages for both the fecundity and hatchability endpoints. This is a standard statistical technique for separating the "signal" from the "noise" in epidemiological and earth sciences (e.g., Borradaile 2003; Rothman et al. 2008). Although we didn't use the control data in our focal segment linear regression, we estimate where the smoothed data would cross the y-axis by double-weighting the control value, which then along with its nearest neighboring data point provided the basis of a three-point moving average for the "endpoint" of the concentration series. This double-weighting is justified conceptually because a treatment to the left of the controls on the concentration axis would be expected to respond the same as the controls (Environment Canada 2005). This enables us to avoid comparing point estimates of eggs hatched per spawn from models fitted to smoothed data with "unsmoothed" control reference points. Note that our "smoothed" estimate of a control reference point is obtained using the actual data nearest to the y-axis and is not extrapolated from our estimated regression equation. Also note that we do not control-adjust the results prior to model fitting, a practice that leads to serious upward bias in EC_x point estimates (Environment Canada 2005; OECD 2006). A summary of response data smoothing and transformation is presented in Table 42.

Table 45. Fathead minnow input data for effects modeling

Treatment (free µg CN/L)	Pooled mean eggs/female	Pooled Proportion Hatch[a]	Unsmoothed Pooled mean hatch/female[b]	3-pt moving average of proportion hatch	Smoothed Pooled mean hatch/female[b]	SQRT transform
Control Mean	3476	0.842	2927	0.763[c]	2652	51.5
6.00	2512	0.606	1522	0.754	1894	43.52
13.30	1845	0.813	1500	0.682	1258	35.47
20.20	1468	0.626	919	0.612	898	29.97
28.20	1367	0.396	541	0.527	720	26.83
36.90	1010	0.559	565	0.354	358	18.92
45.60	1124	0.108	121	0.271	305	17.46
65.60	72	0.147	11	0.149	11	3.31
75.10	319	0.192	61	0.181	58	7.62
83.30	243	0.204	50	0.132	32	5.66
103.90	0	0	0	0.068[c]	0	0

[a]Means weighted by replicate sample sizes; excludes hatchability result for Control B as per authors' (Lind et al. 1977:264-265) recommendation
[b]Rounded to the nearest whole number
[c]Based on double-weighted observed value; assuming any doses to the left of 0% response will be constant and any points to the right of 100% response will be constant
[d]Final effects model based upon the shaded subset of data

The resulting log-square root focal segment linear regression model shows a very close fit to the data with an adjusted r-square of 0.964. The regression equation is:

Square-root (hatched eggs per spawn) = -30.19(LOG CN) + 68.36

The regression plot (Figure 12) and summary regression statistics (Table 43) are presented below. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples, i.e., six points in this case, we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

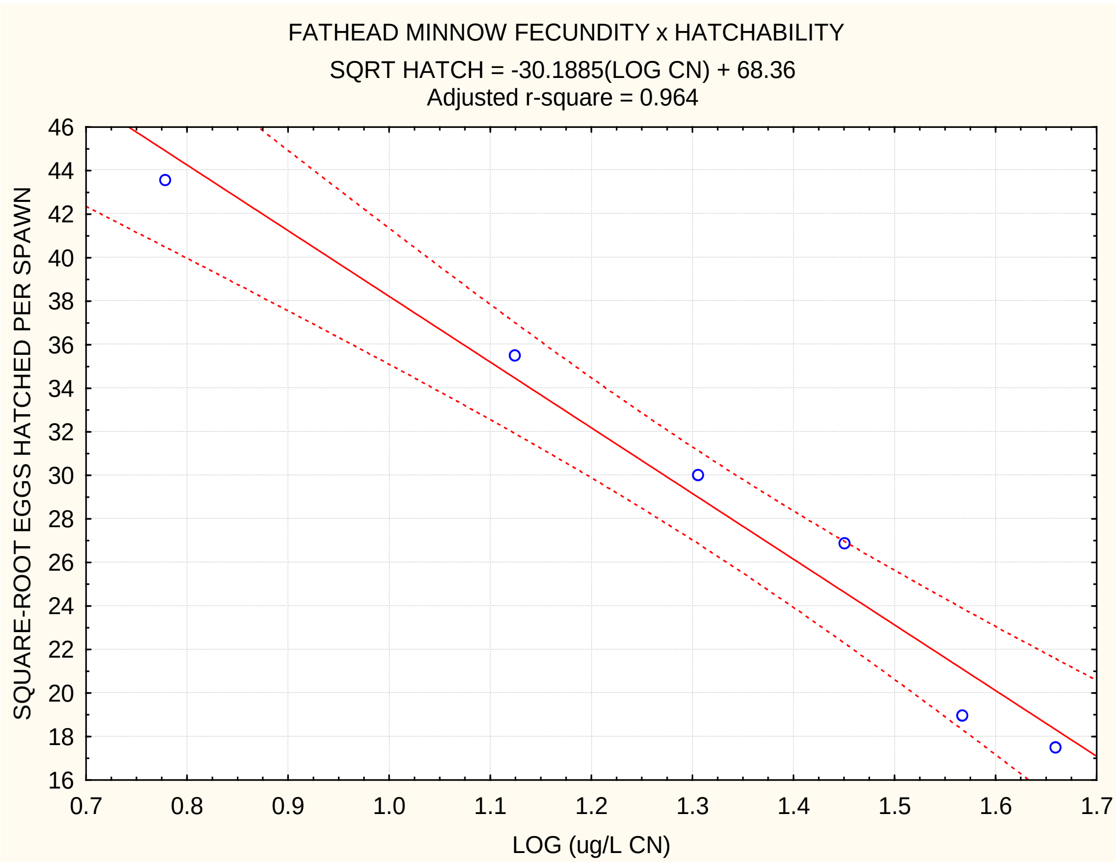

Figure 12. Log- Square Root Focal Segment Regression Plot for Fathead Minnow Fecundity x Hatchability (= Eggs Hatched Per Spawn)

Table 46. Summary regression statistics

Effects Surrogate	N	F value	p-level	Intercept	Std Err	p-level	Slope	Std Err	p-level
Fathead Minnow	6	134.6	<0.00032	68.36	3.505	0.000041	-30.19	2.602	0.00032
Brook Trout	5	12.34	<.039	24.85	2.595	0.0024	-6.594	1.877	0.039
Bluegill	5	11.75	<0.042	0.3514	0.9277	0.73	-2.533	0.7919	0.042

Prediction model based on brook trout dataset. Koenst et al. (1977) examined brook trout fecundity (number of eggs per spawn) and egg viability in relation to a series of cyanide treatments (concentrations). The experimental structure, as well as the fecundity results are summarized below (Table 44). There were two control replicates, and seven cyanide treatments. The lowest five treatments produced intermediate effects responses and covered a range of concentrations from 5.6 to 53.2 µg/L CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 40). There was substantive variability in the results for the two control replicates. This lead Koenst et al. (1977) to exclude control replicate B, but noting that additional testing might indicate that the control results should be averaged. As noted in the footnote to Table 44, subsequent studies with brook trout (Holcombe et al. 2000) have confirmed that control replicate B should be averaged with control replicate A and therefore we use the control mean as our reference point for evaluating model predictions.

Table 47. Egg production of adult brook trout exposed to HCN for 144 days prior to the start of spawning (from Koenst et al. 1977)

HCN (µg/L)	Free cyanide as CN (µg/L)	Mean eggs spawned per female	Reduction in the number of eggs per female - percent of control*
Control A		502	
Control B		744	
Control Mean		623	
5.7	5.6	513	17.7
11.2	11.1	291	53.3
32.3	31.9	246	60.5
43.6	43.1	442	29.1
53.9	53.2	262	57.9
64.9	64.1	124	80.1
75.3	74.4	0	100.0

* Reductions in the number of eggs spawned relative to controls were calculated using the Control mean (623 eggs per female). Koenst et al. 1977 performed the same calculation using only Control A (502 eggs per female) and reported that the MATC (Maximum Acceptable Toxicant Concentration) lies between 5.7 and 11.2 µg HCN/L. However, the authors went on to say that “When compared to the mean of the two controls, 5.7 µg/L HCN would appear to show a substantial reduction in eggs spawned per female, but due to the high variability in spawning in the two controls, further study would be required to reach this conclusion.” Since that time other studies with brook trout have been conducted (Holcombe et al. 2000). The mean number of eggs spawned per female observed by Koenst et al. 1977 is within the range reported for these other studies, which supports the use of data from both controls in estimating the effect of cyanide on brook trout fecundity.

Again, in agreement with Gensemer et al.’s (2007) treatment of the same dataset, we collapse the fecundity and egg viability endpoints into a single endpoint, “viable eggs per spawn” which is the product of (eggs per spawn) x (egg viability) at each treatment concentration. In the five-point segment of the data that we focus on, there was a substantive deviation from monotonicity at the 43.1 µg/L CN concentration. Therefore, once again we employed data smoothing with a 3-point moving average to restore a monotonic progression of responses. Because the endpoint here is virtually the same as the endpoint for the fathead minnow dataset, other aspects of our treatment of the data for “building” a prediction model are the same as already presented above. A summary of response data smoothing and transformation is presented in Table 45 below.

Table 48. Brook trout input data for effects modeling

Treatment (free CN µg/L)	Mean eggs/female	3-pt moving average of mean eggs/spawn	Proportion Viable	3-pt moving average of proportion viable	Smoothed mean viable/female[a]	SQRT transform
Control Mean	623	586[b]	0.935	0.923[b]	541	23.26
5.60	513	476	0.899	0.872	415	20.37
11.10	291	350	0.781	0.803	281	16.76
31.90	246	326	0.729	0.792	258	16.06
43.10	442	317	0.866	0.745	236	15.36
53.20	262	276	0.641	0.502	139	11.79
64.10	124	129	0	0.214	28	5.29
74.40	0	41[b]	0	0[b]	0	0

[a]Rounded to the nearest whole number
[b]Based on double-weighted observed value; assuming any doses to the left of 0% response will be constant and any points to the right of 100% response will be constant
[c]Final effects model based upon the shaded subset of data

The resulting log-square root focal segment linear regression model does not show as strong a fit to the data as the fathead minnow model does, but still shows a reasonably good fit with an adjusted r-square of 0.739. The regression equation is:

Square-root (viable eggs per spawn) = -6.594(LOG CN) + 24.85

The regression plot is presented in Figure 13 and summary regression statistics are presented in Table 43. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples, i.e., five points in this case, we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

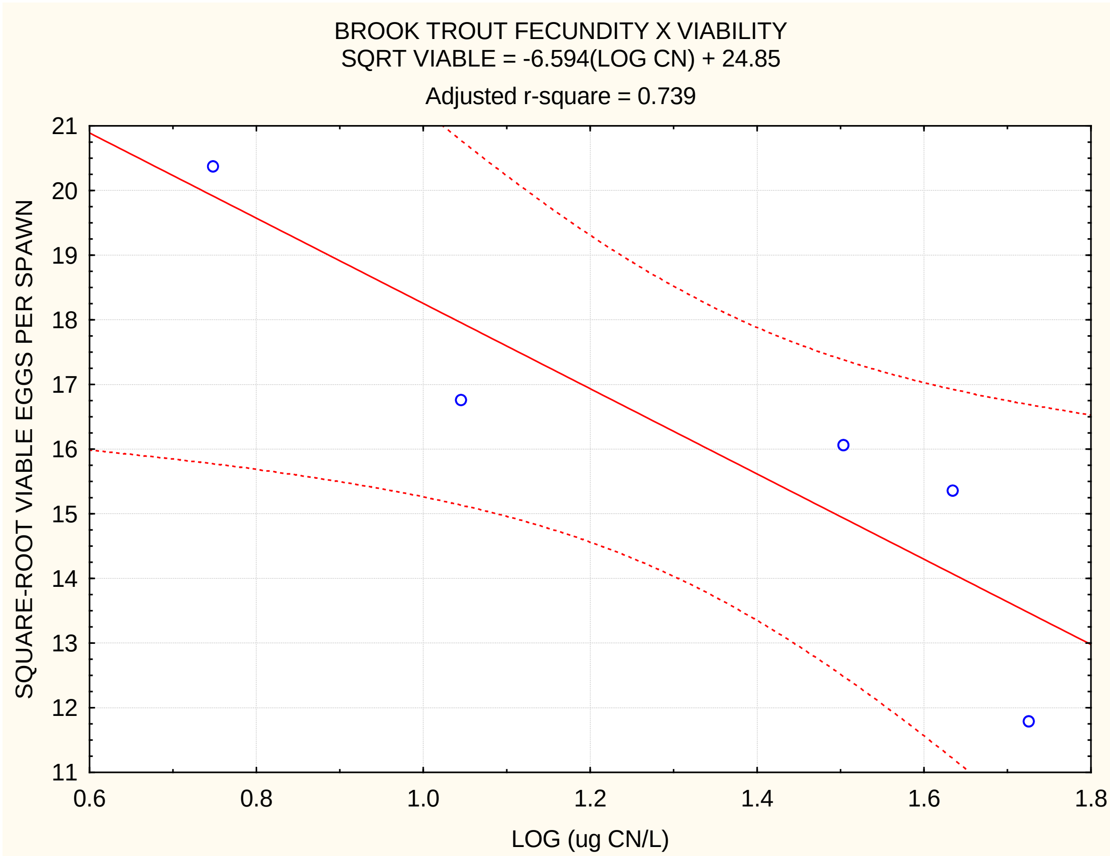

Figure 13. Log-Square Root Focal Segment Regression Plot for Brook Trout Fecundity x Viability(= Viable Eggs per Spawn)

Prediction model based on bluegill dataset. Kimball et al. (1978) examined bluegill juvenile survivorship in relation to a series of cyanide treatments (concentrations). The experimental structure, as well as the survivorship results are summarized in Table 46. There were four control replicates, and two replicates each for eight cyanide treatments. The lowest five treatments produced intermediate effects responses and covered a range of concentrations from 4.9 to 40.6 µg/L CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 40).

Table 49. Survival of bluegill from fertilized egg to the 57-day juvenile state in various HCN concentrations (from Kimball et al. 1978)

HCN (µg/L)	Mean HCN (µg/L)	Free cyanide as CN (µg/L)	Percent survival	Number of surviving juveniles *	Mean percent survival	Reduction in survival compared to controls
Control			37.5	75	23.3	
Control			20.0	40		
Control			10.0	20		
Control			25.5	51		
4.8	4.8	4.9	18.5	37	18.5	20.6%
5.2			lost			

8.9	9.1	9.4[N]	25.0	50	16.3	30.0%
9.2			7.5	15		
19.2	19.4	19.9[L]	3.0	6	2.8	88.0%
19.6			2.5	5		
28.5	29.1	29.9	2.5	5	2.5	89.3%
29.7			2.5	5		
38.7	39.5	40.6	3.0	6	3.8	83.7%
40.2			4.5	9		
49.3	49.3	50.7	13.5	27	13.5	42.1%
51.9			lost			
61.8	62.9	64.6	0.0	0	0.0	100.0%
64			0.0	0		
80.4	82.1	84.4	0.0	0	0.0	100.0%
83.8			0.0	0		

*Number of surviving juveniles was calculated by multiplying the reported percent survival times the starting number of fertilized eggs per treatment (200).
[N]NOEC
[L]LOEC

The bluegill dataset differs qualitatively from the fathead minnow and brook trout datasets because the response variable, juvenile survivorship is a quantal (binary) rather than continuous variable. Quantal variables conform to a binomial distribution. Such data are typically analyzed via either probit transformation, as employed by Gensemer et al. (2007), or logit transformation of the proportions of responding and non-responding test subjects. Probits are normal equivalent deviates and logits are logistic equivalent deviates. These two transforms usually yield similar estimates of EC_{50} values, but differ appreciably in their EC estimates in the tails of the distributions.

Environment Canada (2005) recommends logistic methods over probits for "... mathematical simplicity and other good reasons." Logit = ln (p/1-p), where p is the proportion of effected test subjects (e.g., if juvenile survival were 30% for a particular treatment concentration, p would equal 0.3 and the logit transform would equal -0.8473). The logit transform linearizes the sigmoidal logistic response curve (Environment Canada 2005; StatSoft 2006). Furthermore, in fitting the logit model, the control observations can be excluded, as they do not provide any information, unless a background parameter in included (OECD 2006).

Both Environment Canada (2005) and OECD (2006) note that it is common practice to correct the data for background response prior to analysis (for example via Abbott's correction), but that such pre-treatment of the data is unsound statistical practice that can result in substantive overestimation of EC_x values. The bias increases as the control effect being adjusted for increases. We fit a focal segment of the bluegill dataset to a log-logit regression using results that are not control-adjusted prior to analysis. Thus, our prediction model yields unbiased estimates of proportion effect that can be control-adjusted for reporting purposed after the fact. The dataset is reasonably monotonic until the highly anomalous result for the treatment at a concentration of 50.7 µg/L CN. Gensemer et al. (2007) censored that point as an outlier. Because our $SSEC_x$ range extended up to only 41.7 µg/L CN (Table 40) the 50.7 µg/L CN treatment did not fall within our focal segment of concern. The last three treatments in our focal segment produced results of greater than 84% effect which would place them in the nonlinear upper tail of the sigmoidal curve (Figure 11), but unlike a log-square root regression the logit transform will linearize points in the tails relative to intermediate effects points. Thus, for log-logit regression points that fall in tails do not have to be avoided in order to apply linear regression. The minor deviation from monotonicity in the last two points of our focal segment did not warrant data smoothing. A summary of the logit transformed response data is presented in Table 47.

Table 50. Bluegill input data for effects modeling

Treatment (free CN µg/L)	Mean surviving juveniles	Proportion Survival	Logit Proportion Survival
Control Mean	46.5	0.2325	-1.1942
4.9	37	0.1850	-1.4828
9.4	32.5	0.1630	-1.6361
19.9	5.5	0.0280	-3.5472
29.9	5	0.0250	-3.6636
40.6	7.5	0.0380	-3.2314
50.7	27	0.1350	-1.8575
64.6	0	0.0000	
84.4	0	0.0000	

[a]Final effects model based upon the shaded subset of data

The resulting log-logit focal segment linear regression model does not show as strong a fit to the data as the fathead minnow model does, but with an adjusted r-square of 0.729 shows a reasonably good fit comparable to that achieved for the brook trout dataset. The regression equation is:

$$\text{Logit (proportion juvenile survival)} = -2.533\ (\text{LOG CN}) + 0.3514$$

The regression plot is presented in Figure 14 and summary regression statistics are presented in Table 43. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples, i.e., five points in this case, we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

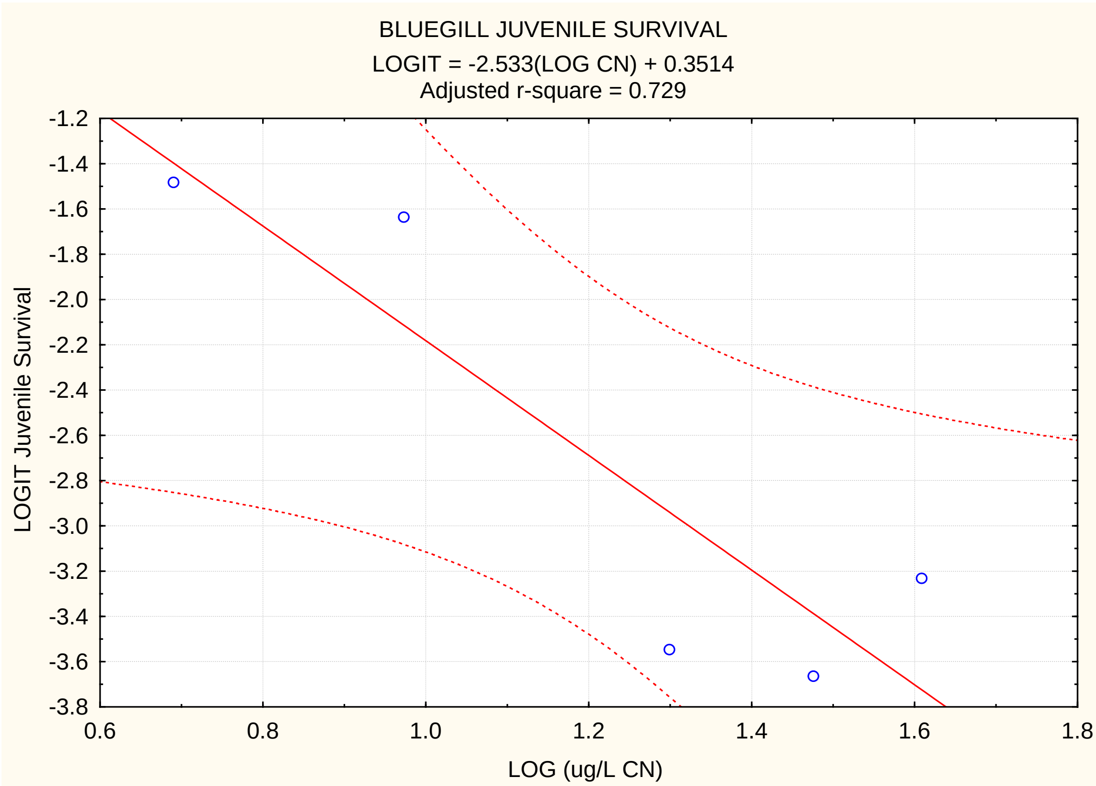

Figure 14. Log-logit focal segment regression plot for bluegill juvenile survival

Prediction Results

Effects predictions are generated by substituting LOG ($SSEC_x$) for LOG (CN) into the prediction regression equations. This was accomplished via the "predict dependent variable" algorithm in the multiple linear regression module of Statistica (StatSoft 2006). That algorithm also uses the estimated standard error of the regression coefficient to generate 95% confidence limits for the predicted point estimates (maximum likelihood estimates). For the fathead minnow and brook trout prediction regressions, the prediction and confidence limit output are in the form of square-roots of numbers of eggs. To convert those predictions to a percent effect, the predicted results were first squared and then scaled for percent change compared to the applicable smoothed control value according to the formula:

% Effect = [1- (predicted egg count / smoothed control value)] x 100

Any predicted egg counts exceeding the smoothed control value were automatically converted to 0% effect. For the bluegill prediction regression, the prediction and confidence limit output are in the form of logit transforms for proportions of juvenile survivorship. The logit transforms are back-transformed to proportions by the formula:

Proportion survival = $e^{(logit)} / 1 + e^{(logit)}$

The predicted survival proportions are scaled for percent change compared to the reported control value according to the formula:

% Effect = [1- (predicted proportion survival / mean control proportion survival)] x 100

Again, any predicted survivorship exceeding the observed mean control survivorship results in a percent effect prediction that is automatically converted to 0% effect. The raw input and output data for effects predictions are presented in Appendix E.

A summary of predicted effects and their estimated 95% confidence limits from each of the three prediction models for each of the 14 surrogate taxa from which listed-species' LC_{50} values were derived are presented in Table 48. The effects estimates are presented in Table 49for the listed species (i.e., matches up the effects estimates for surrogate taxa in Table 48 with the listed species linked to each surrogate taxon).

The EC_{10} and EC_{20} concentrations for each of our three regression models were also estimated. The fathead minnow regression yielded an estimated EC_{10} of 4.4 µg/L CN (95% CI = 2.6-6.2 µg/L CN) and an estimated EC_{20} of 5.5 µg/L CN (95% CI = 3.5-7.4). By comparison, Gensemer et al. (2007) estimated an EC_{20} of 6.0 µg/L CN from a log-probit analysis of the fathead minnow data, but did not report confidence limits for that estimate. The brook trout regression yielded an estimated EC_{10} of 2.6 µg/L CN (95% CI = 0.0-8.4 µg/L CN) and an estimated EC_{20} of 4.1 µg/L CN (95% CI = 0.0-11.1). Gensemer et al. (2007) estimated an EC_{20} of 7.7 µg/L by linear interpolation of the brook trout data, and again did not report confidence limits for that estimate. The bluegill regression yielded an estimated EC_{10} of 4.6 µg/L CN (95% CI = 0.0-10.5 µg/L CN) and an estimated EC_{20} of 5.3 µg/L CN (95% CI = 0.0-11.5). Gensemer et al. (2007) estimated an EC_{20} of 5.6 µg/L CN from a log-probit analysis of the bluegill data, and also estimated an EC_{20} of 8.9 µg/L CN for the bluegill data from EPA's TRAP program. All of Gensemer et al.'s (2007) estimates fall within our 95% confidence limits, and in general show excellent agreement with our results even though Gensemer et al's methods differed from ours. This suggests that our results are not highly dependent on the particular statistical approach that we chose for our analysis.

Table 51. Estimated magnitude of effect of cyanide (at the CCC, 5.2 µg CN/L) on surrogate taxa for listed fish species (95% CL)*

Surrogate taxa used to estimate magnitude of effect on listed species	**Surrogate species**		
	Fathead Minnow	**Brook Trout**	**Bluegill**
	Reduction in the mean number of hatched eggs per spawn compared to controls	**Reduction in the mean number of viable eggs per spawn compared to controls**	**Reduction in the number of surviving larvae/juveniles compared to controls**
Actinopterygii (class**)**	48% (39%, 56%)	30% (1%, 55%)	56% (3%, 82%)
Order Cypriniformes	39% (28%, 49%	26% (0%, 54%)	44% (0%, 80%)

Family Catostomidae			
Xyrauchen texanus (species)	39% (28%, 49%)	26% (0%, 54%)	44% (0%, 80%)
Cyprinidae (family)	29% (15%, 42%)	21% (0%, 53%)	30% (0%, 78%)
Cyprinella monacha (species)	68% (63%, 72%)	42% (23%, 58%)	76% (50%, 89%)
Gila elegans (species)	57% (51%, 63%)	36% (12%, 56%)	66% (30%, 84%)
Notropis mekistocholas (species)	59% (53%, 65%)	37% (14%, 56%)	68% (34%, 85%)
Ptychocheilus lucius (species)	63% (57%, 68%)	39% (18%, 57%)	71% (41%, 86%)
Order Perciformes	36% (24%, 47%)	24% 0%, 53%)	40% 0%, 79%)
Percidae (family	63% (58%, 68%)	39% (18%, 57%)	72% (43%, 87%)
Etheostoma (genus	65% (60%, 70%)	40% (20%, 58%)	74% (46%, 88%)
Etheostoma fonticola (species)	81% (76%, 85%)	52% (37%, 64%)	86% (64%, 95%)
Order Salmoniformes, Family Salmonidae			
Oncorhynchus (genus)	60% (54%, 65%)	37% (15%, 57%)	69% (36%, 85)
Oncorhynchus apache (species)	87% (82%, 91%)	56% (42%, 68%)	90% (67%, 97%)
Oncorhynchus tschawytscha (species)	71% (67%, 76%)	45% (27%, 59%)	79% (55%, 91%)
Oncorhynchus kisutch (species)	71% (66%, 75%)	44% (27%, 59%)	79% (55%, 90%)
Oncorhynchus mykiss (species)	52% (45%, 59%)	33% (6%, 55%)	61% (16%, 83%)
Oncorhynchus clarki henshawi (species)	80% (75%, 84%)	51% (36%, 63%)	85% (63%, 94%)
Salvelinus (genus)	87% (83%, 92%)	57% (43%, 69%)	90% (68%, 97%)

*The magnitude of effect was estimated using the regression model for each surrogate response species and SS EC_X value for each surrogate taxa (Table 40). For each surrogate taxa there were two estimates of effects on reproductive performance and one estimate of effects on early life stage survival.

Table 52. Estimated magnitude of effect of cyanide (at the CCC, 5.2 µg CN/L) on listed fish species (95% CL). There are two estimates for effects on fecundity and one estimate for effects on early life stage survival for seven listed species due to exposure at based on surrogate species data.

Listed Species	Surrogate Taxa	Estimated reduction in fecundity and larvae/juvenile survival due to cyanide exposure at the CCC based on surrogate species data sets		
		Fathead minnow[1] (Percent reduction in the mean number of hatched eggs per spawn compared to controls)	Brook trout[2] (Percent reduction in the mean number of viable eggs per spawn compared to controls)	Bluegill[3] (Percent reduction in the number of surviving larvae/juveniles compared to controls)
Coho salmon *(Oncorhynchus kisutch)*	*Oncorhynchus kisutch*	71 (66, 75)	44 (27, 59)	79 (55, 90)
Chinook salmon (*Oncorhynchus tschawytscha)*	*Oncorhynchus tschawytscha*	71 (67, 76)	45 (27, 59)	79 (55, 91)
Chum salmon *(Oncorhynchus keta)*	*Oncorhynchus (genus)*	60 (54, 65)	37 (15, 57)	69 (36, 85)
Sockeye salmon *(Oncorhynchus nerka)*	*Oncorhynchus (genus)*	60 (54, 65)	37 (15, 57)	69 (36, 85)
Steelhead (*Oncorhynchus mykiss)*	*Oncorhynchus mykiss*	52 (45, 59)	33 (6, 55)	61 (16, 83)
Shortnose sturgeon *(Acipenser brevirostrum)*	Actinopterygii (class)	48 (39, 56)	30 (1, 55)	56 (3, 82)
Green sturgeon *(Acipenser medirostris)*	Actinopterygii (class)	48 (39, 56)	30 (1, 55)	56 (3, 82)

[1]Based on data contained in Lind et al. 1977
[2]Based on data contained in Koenst et al. 1977
[3]Based on data contained in Kimball et al. 1978
[4]Based on data contained in Schimmel 1981

Other Effects Estimates

The estimates of effects presented in Table 49 are based largely on ICE LCL (lower confidence limit) LC_{50} values for listed fish evaluation species. Those are the LC_{50} values that we accept as sufficiently accounting for the uncertainties inherent in relying on surrogate data and numerous other untested assumptions to estimate the sensitivity of listed species to cyanide. The Service, NMFS, and EPA agreed that using ICE LCL values was preferable to the practice of applying arbitrary uncertainty factors.

However, EPA has, at various times, questioned whether the use of ICE LCL values might not be overly conservative. Therefore, we also estimated effect levels using ICE MLE (maximum likelihood estimates) LC_{50} values for listed fish evaluation species (via revised $SSEC_x$ estimates). Those results are presented in Appendix F. Based on the fathead minnow prediction model, which was the strongest model, the median levels of effect predicted for the 15 ICE surrogate taxa were 51% and 65%, respectively, for ICE MLE and ICE LCL. The number of surrogate taxa with a predicted effect of 35% or greater was 11 and 14, respectively, for ICE MLE and ICE LCL. Those differences indicate only modest conservatism conferred by ICE LCL-based effects estimates as compared to ICE MLE-based estimates. Such modest differences would not have a decision-making impact. For both sets of results, unacceptably high levels of effect would overwhelmingly be the predominant prediction.

Empirical Test of Method Performance

Because only three concentration-response datasets are available, there is almost no basis for testing our method performance (i.e., there are no known directly measured "true" values for effects to our listed fish evaluation species at a concentration of 5.2 µg/L CN). However, because the fathead minnow and brook trout datasets focused on essentially the same response variable (number of hatchable/viable eggs produced per spawn) we can perform two tests of method performance. For each species, we can directly estimate a predicted effect level at 5.2 µg/L CN using the species-specific regressions. Those would be our estimates of the "true" effect level. Next, we can use our surrogate method and estimate an $SSEC_x$ for each species on the other species' response curve and evaluate the predicted effect level for that $SSEC_x$ value and compare the surrogate estimate to the estimated "true" value. The results are as follows:

> The directly estimated fathead minnow effect level at 5.2 µg/L CN is 18% with a 95% CI of 0%-34%. The fathead minnow $SSEC_x$ value on the brook trout response curve would be 3.2 µg/L CN, which yields an effects estimate of 15%. That is nearly identical to estimated "true" value and easily within the 95% CI for the "true value".

> The directly estimated brook trout effect level at 5.2 µg/L CN is 25% with a 95% CI of 0%-54%. The brook trout $SSEC_x$ value on the fathead minnow response curve would be 8.4 µg/L CN, which yields an effects estimate of 38%. Again, that is within the 95% CI for the "true" value, although our estimate of the "true" value is not very precise and therefore the 95% CI is fairly wide.

In summary, in both test cases, the estimated effect level derived from our surrogate methodology is not significantly different from the estimated "true" value in a statistical sense, but the second comparison has low statistical power. Further validation testing of this sort should be done as concentration-response datasets become available for more species using a comparable response variable, but it is reassuring that in these test cases our method yielded results that were nearly identical to the "true" value in one case and reasonably close to the "true" value in the other case.

Derivation of the Criterion Continuous Concentration (CCC)

Our analysis predicts that the listed fish species considered in this Opinion would be severely affected by exposure to cyanide at the CCC. National criteria are typically derived using chronic toxicity data from laboratory tests. As noted earlier, most aquatic life criteria that have been derived thus far, including the cyanide criterion, chronic values have been obtained by calculating the geometric mean of the lower and upper chronic limits. In practice, the upper and lower chronic limits are often statistically determined by hypothesis testing. The lower limit is typically the NOEC, which is defined as the highest test concentration where the effects are not statistically significantly different from controls. The upper limit is typically the LOEC, which is defined as the lowest test concentration where the effects are statistically significantly different from controls. The guidelines recommend that the magnitude of effect associated with the upper and lower chronic limits should be considered when determining values that appropriately estimate acceptable and unacceptable levels of adverse effect:

> *Because various authors have used a variety of terms and definitions to interpret and report results from chronic tests, reported results should be reviewed carefully. The amount of effect that is considered unacceptable is often based on a statistical hypothesis test, but might also be defined in terms of a specified percent reduction from the controls. A small percent reduction (e.g., 3%) might be considered acceptable even if it is statistically significantly different from the control, whereas, a large percent reduction (e.g., 30%) might be considered unacceptable even if it is not statistically significant."*

Based on this guidance, the threshold for unacceptable adverse effects would be estimated by the chronic value. The magnitude of effect at the threshold would then be equivalent to the magnitude of effect at the chronic value. For chronic criteria derived using hypothesis tests, this would be the magnitude of effect occurring at a concentration equal to the geometric mean of the NOEC and LOEC, that is, somewhere between an acceptable and unacceptable level of adverse effect. The guidelines do not specify a level of adverse effect on which the threshold for unacceptability should be based. The only mention of a numeric value or range is provided in the guidance for selecting chronic limits (mentioned above) and suggests that this threshold may lie between 3% and 30%.

Thus, for a given species or test the magnitude of effect at the chronic value will depend on the magnitude of effect at the lower and upper chronic limits. We followed this approach for estimating the magnitude of effect occurring at the cyanide CCC. The freshwater cyanide CCC was derived based on chronic toxicity data for 4 species (Table 50): 3 fish (fathead minnow, brook trout, and bluegill) and 1 invertebrate (*Gammarus pseudolimnaeus*). Chronic values for each species were obtained by calculating the geometric mean of the lower and upper chronic limits. The magnitude of effect at the lower and upper chronic limits was calculated by comparing responses at the lower and upper limits to controls. For fathead minnow and brook trout these effects were expressed as reduction in the mean number of eggs spawned per female compared to controls; for the bluegill the effect was reduction in larvae/juvenile survival compared to controls; and for *G. pseudolimnaeus* the effect was a reduction in the mean number of eggs or young per gravid female relative to controls.

We then estimated the magnitude of effect at the chronic value by linear interpolation between lower and upper chronic limits (Table 50). Based on these calculations the magnitude of effect at the chronic values for the fathead minnow, brook trout, bluegill and *G. pseudolimnaeus* would be 52%, 32%, 54%, and 47%, respectively. According to the guidelines, if there were a sufficient number of chronic values (i.e., chronic values for species in 8 phylogenetic families) the chronic criterion could be computed directly from the distribution of chronic values (see earlier discussion under *Derivation of Criteria*). If there were fewer chronic values, as was the case for cyanide, the chronic criterion would be computed using ACRs. ACRs for the 4 freshwater species were reported in the cyanide criterion document and are shown in Table 50. The ACRs were calculated by dividing the species mean acute value (i.e., mean LC_{50} for the species) by the chronic value. For example, the ACR for fathead minnow (7.633) was computed by dividing 125.1 µg CN/L (the mean LC_{50} for the species) by 16.39 µg/ CN/L (the chronic value). Thus, the ACR is the ratio between the concentration of cyanide causing 50% lethality (following acute exposure) and the concentration following chronic exposure that causes a level of adverse effect that is at the threshold of unacceptability, i.e., 52% for fathead minnow. The guidelines require that, for criteria derivation, the geometric mean of individual species ACRs is used to obtain the Final ACR. For cyanide, the freshwater Final ACR was 8.562 (Table 50). We estimated the magnitude of chronic effects associated with the Final ACR to be about 45% (Table 50).

The Final ACR and the FAV were then used to derive the CCC. The guidelines describe how the FAV is computed. In short, the FAV is set equal to the 5th percentile estimate from the distribution of genus mean acute values. In other words, the FAV represents the genus with acute sensitivity (LC_{50}) in the *sensitive tail* of the distribution where, theoretically, approximately 5% of the genera would be more sensitive and about 95% of the genera would be less sensitive. Based on this analysis, the FAV for cyanide was determined to be 62.68 µg CN /L. Since the guidelines include provisions for adjusting the FAV to protect commercially and recreationally important species, EPA lowered the FAV from 62.68 µg/L to 44.73 µg/L because the SMAV for rainbow trout (44.73 µg/L) was below the calculated FAV. The cyanide criterion (5.2 µg/L) was then derived by division of the FAV (44.73 µg/L) by the Final ACR (8.562). Thus the chronic criterion, 5.2 µg CN/L, was based on the concentration intended to protect rainbow trout from unacceptable adverse effects. Based on our estimate of the magnitude of effect associated with the Final ACR, we estimate the magnitude of adverse effects occurring to rainbow trout at the chronic criterion to be approximately 45%. This value is higher than we would have expected considering it is intended to represent the threshold for unacceptable adverse effects. However, the magnitude is in line with effects we predicted for the other listed fish species, most of which were estimated to be as (or more) sensitive to cyanide as rainbow trout.

Table 53. Chronic toxicity data used by EPA to derive the freshwater chronic criterion for cyanide. **Effect levels were calculated using data from the original papers.**

Species	Chronic Limits[1]				Chronic Value[2]		LC_{50}[3] (µg CN/L)	ACR[3]
	Lower		Upper					
	(µg CN/L)	Effect	(µg CN/L)	Effect	(µg CN/L)	Effect		
Fathead Minnow	13.3	47%	20.2	58%	16.39	52%	125.1	7.633
Brook Trout	5.6	18%	11.0	53%	7.849	32%	83.14	10.59
Bluegill	9.3	30%	19.8	88%	13.57	54%	99.28	7.3
Gammarus	16	0%	21	100%	18.33	47%	167	9.111
Geometric mean						45%		8.562

[1] Lower and upper chronic limits were taken from the cyanide criteria document. For fathead minnow and bluegill these values were determined statistically (i.e., NOEC and LOEC identified via hypothesis tests). Effect levels were take from Tables 5, 8 and 10 in the Effects section of the BO and from Oseid and Smith 1979.
[2] Chronic values were taken from the cyanide criteria document. Effect levels associated with the chronic values were estimated by linear interpolation between the effects at the lower and upper chronic limits.
[3] Acute-Chronic Ratios were taken from the cyanide criteria document.

This same conclusion, that NOEC/LOEC-based estimates of "chronic values" can correspond to $\geq$ 40% adverse effect, has also been reached by others. Decades ago Suter et al. (1987) reported that MATC's for fish fecundity, on average, corresponded to a 42% level of adverse effect (MATC = Maximum Acceptable Toxicant Concentration; a term for the geometric mean of the NOEC and LOEC from a given toxicity test and often assigned by EPA as the estimated "chronic value" from a test). Other response endpoints were found to correspond to average adverse effect levels of 12-35%. More recently, SETAC (Society for Environmental Toxicology and Chemistry) convened a panel of experts (Reiley et al. 2003) who concluded that "*...[toxicity] tests with high variability may result in an(sic) NOEC corresponding to a response greater than 40% different from the control.*" Moore and Caux (1997) statistically examined nearly 200 toxicity data sets and found that most NOEC's (76.9%) exceeded a 10% adverse effect level and most LOEC's (62.4%) exceeded a 30% effect level. Various other researchers have noted a variety of adverse effect levels for NOEC's, such that Crane and Newman (2000) were led in summary to conclude that "*… [NOEC] effect levels from individual tests ranged from nearly 0% to nearly 100%.*" For seven cyanide toxicity tests with sufficient data for comparison, Gensemer et al. (2007: Figure 3-7) found in all cases that the geometric mean of the NOEC and LOEC corresponded to an adverse effect level of $\geq$ 20% (how much greater was not reported).

Because of the highly variable and often substantive levels of effect associated with NOEC's, LOEC's, MATC's, and with the "chronic values" based on them, and for numerous other reasons, a strong professional consensus recommendation to avoid using NOEC/LOEC-based estimates for regulatory thresholds (when possible) has been expressed repeatedly. For example, there was an ISO (International Organization for Standardization) resolution (ISO TC147/SC5/WG10 Antalya 3) as well as an OECD (Organisation for Economic Co-operation and Development) workshop recommendation (OECD 1998) that the NOEC should be phased out from international standards (OECD 2006:14). Environment Canada (2005) notes, that there is a growing literature which points out many deficiencies of the NOEC approach (Andersen et al. 2000; Bailer and Oris 1999; Chapman 1996; Chapman et al. 1996; Crane and Godolphin 2000; Crane and Newman 2000; Miller et al. 1993; Moore and Caux 1997; Noppert et al. 1994; Pack 1993; Pack 1998; Suter et al. 1987; Suter 1996). Moving away from the NOEC/LOEC approach was also among the recommendations of the SETAC panel for improving the scientific basis of water-quality criteria (Reiley et al. 2003).

Accordingly, EPA has begun employing a regression approach for estimating "chronic values" whenever sufficient data are available to do so. For example, in the 1999 update for ammonia water quality criteria EPA used regression analyses to estimate 20% effect concentrations (EC_{20}s) from individual toxicity tests and used those EC_{20}s as estimates of chronic values (EPA 1999). Likewise, estimated EC_{20}s have been the basis for estimating chronic values in recently proposed updates for copper and selenium water quality criteria (EPA 2003a, 2004). EPA's choice of the EC_{20} as a basis for estimating chronic values was justified from statistical considerations rather than from biological or demographic considerations:

> *To make [chronic values] reflect a uniform level of effect, regression analysis was used here both to demonstrate that a significant concentration-effect relationship was present and to estimate [chronic values] with a consistent level of effect. Use of regression analysis is provided for on page 39 of the 1985 Guidelines (Stephan et al. 1985). The most precise estimates of effect concentrations can generally be made for 50 percent reduction (EC50); however, such a major reduction is not necessarily consistent with criteria providing adequate protection. In contrast, a concentration that caused a low level of reduction, such as an EC5 or EC10, is rarely statistically significantly different from the control treatment. As a compromise, the EC20 is used here as representing a low level of effect that is generally significantly different from the control treatment across the useful chronic datasets that are available for ammonia.*

Pack (1993) asserted that most ecotoxicologists consider effects in the range of 5-20% to be biologically acceptable depending on the species involved and the type of effect. However, EPA appears to have chosen the top end of that range based more on the expected statistical power of toxicity tests than on a serious examination of the typical demographic sensitivity of biotic populations to a 20% adverse effect on survival, growth, or reproduction. Furthermore, 95% statistical confidence limits for most EC_{20} estimates are likely to extend well into adverse effect levels that would be of unquestionably serious demographic concern for most organisms. As evident from the above discussion, most chronic criteria derived by EPA, including for cyanide, are highly likely to be associated with $\geq$ 20% adverse effect level for species at the vulnerable end of species sensitivity distributions (such as the subset of ESA-listed species we are evaluating). Therefore, it should be no surprise that our estimated effect levels for such species at the current cyanide CCC of 5.2 µg/L are almost always higher than 20% and in some cases substantially higher.

Population Responses to Reductions in Fecundity and Juvenile Survival

Laboratory experiments have demonstrated that even closely related fish species can demonstrate great differences in sensitivity when exposed to the same chemical, as measured by differences in acute or chronic toxicity values. This variability in sensitivity has been related to differences in species' physiology and life history strategies. Similarly, population modeling and experimental studies have shown that variation in population-level responses to environmental toxicity can also be expected among species as a consequence of factors such as life history strategies, life stage affected, and density dependence. Studies have also demonstrated that chronic toxicity can lead to population decline and extirpation.

Under the ESA, in determining whether a proposed Federal action is likely to jeopardize the continued existence of a listed species under the ESA, we assess whether the proposed activity reasonably would be expected to appreciably reduce the likelihood of survival and recovery of a listed species by reducing its reproduction, numbers, or distribution. Two common metrics used in population modeling to assess effects of perturbations on populations are population growth rate and time to or probability of extinction.

Population growth rate is the change in a population size over a unit time period. Long-term reductions in population growth rate as low as 5% has been shown to significantly increase a population's likelihood of extinction (Snell and Serra 2000). Population growth rate can be positive when the population is increasing, negative when decreasing, or zero when the net difference between births, deaths, and migration is zero and the population is stable. For listed species, populations may exist in any of these states depending on its recovery status. Our analysis determines the relative predicted effects of the action to the population growth rate, regardless of its starting value.

Using known parameters of a species' life history, sensitivity analyses can be conducted to determine which parameters, when modified, will have the greatest impact on the species' population growth rate. Elasticity analysis is one type of sensitivity analysis that is commonly used in conservation biology to demonstrate the relative contributions to population growth rate made by life cycle transitions, based on vital rate statistics for survival, growth and fertility. While these types of analyses cannot predict absolute effects to population size, because they quantify the relative importance of an element to changes in population growth rate, they can help focus management decisions on those demographic parameters that exhibit the largest elasticity, and thus, the largest impact on population growth (de Kroon et al. 2000). However, elasticity analysis requires the development of a population model, for which adequate data are often scarce. Because this type of demographic data is often lacking for threatened and endangered species in particular, the need to develop generalized approaches for classifying population responses to perturbation for rare species has been recognized (Dennis et al. 1991; Heppell et al. 2000).

Several authors have examined the effect of life history strategies on the elasticities of various demographic measures. In evaluating demographic parameters of 50 mammal populations with different life history strategies, Heppell et al. (2000) found that phylogeny alone is often not a reliable indicator of which vital rates (survival, growth and fertility) will have the greatest impact on elasticity. Instead, the authors found that species that mature early and have high reproductive output had high fertility elasticities and low adult survival elasticities. Conversely, for those which mature late and have long lifespans, fecundity and early offspring survival are less important than survival of juveniles to maturity to changes in population growth rate. Calow et al. (1997) also found that the relative importance of juvenile fish survival can vary according to reproductive strategy. These authors concluded that reductions in juvenile survival would have the greatest impact on semelparous fish species, in which adults die after reproduction, a lesser impact on a moderately iteroparous population, in which adult postreproductive survival is intermediate, and the least impact on strongly iteroparous species, in which adult survival after reproduction is high. These assumptions held true for elasticity analysis of the green sturgeon, a fish species with life history patterns such as late-maturity and long-life that are common to other sturgeon (Heppell 2007).

Juvenile survival had relatively lower elasticity values than adult and subadult survival, with compensation for the loss of adults requiring much larger increases in young-of-the-year survival than would be commensurate with the loss. However, other authors have found increased importance of juvenile survival for sturgeon, despite their lifespan (Gross et al. 2002; Paragamian and Hansen 2008). Gross et al. (2002) hypothesized that this difference was due to the vastly larger fecundity of sturgeon as compared to other long-lived species.

Vélez-Espino et al. (2006) argue the need for a broadscale summary of species' population dynamics to help guide the conservation biology of freshwater fishes, for which information on life history is often limited. Using information, on adult survival, juvenile survival, and fecundity, the authors performed elasticity analyses on 88 species of freshwater fish and found that they could be classified into 4 functional groups with regard to the sensitivity of their population growth rates:

1. species most sensitive to perturbations in adult survival
2. species most sensitive to perturbations to adult and juvenile survival
3. species most sensitive to perturbations to juvenile survival
4. species most sensitive to perturbations to juvenile survival and fecundity

These groups are characterized by decreased age at maturity, longevity, and reproductive lifespan as one moves from group 1 to group 4. Age at maturity, reproductive lifespan, fecundity, juvenile survivorship, and longevity were all correlated with adult survival and fecundity. However, the best predictors of elasticity patterns were longevity, which explained 93% of the variability in the elasticity of adult survival, and age at maturity, which explained 92% of the variability in the elasticity of fecundity. The authors also found that elasticities are highly conserved among genera within the same taxonomic family

Spromberg and Birge (2005) also found that life history strategies influence effects to populations. The five life history strategies they modeled encompassed differences in stage-specific survival, fecundity and hatch success, number of spawning events, and life-span. The authors found that regardless of strategy, changes in the number of young-of-the-year stage individuals had the greatest impact on population growth rate. However, the relative contribution of this parameter was greatest for life history strategies with multiple spawnings, high fecundity, and short lifespans as opposed to those with longer lifespan, which had increased elasticity of adult survival.

Spromberg and Meador (2005) linked toxicant effects on immune suppression, reproductive development, and growth reduction to demographic traits in Chinook salmon and modeled their influence on population growth rate. Overall, effects to first- and second-year survival had the greatest elasticities, with constant reductions to first year survival as low as 10% achieving population declines ranging from 35-78% compared to controls. Other studies have demonstrated the importance of first year survival in this species (Kareiva et al. 2000). Spromberg and Meador (2005) also found that models which incorporated effects to both survival and reproduction were additive, indicating the importance of evaluating the overall impact of all potential impacts to population growth.

Many listed species populations are limited by the amount of adequate habitat or resources and experience some degree of density dependence. Density-dependence at any life stage must be considered in elasticity analysis in order to yield appropriate results (Grant and Benton 2000; Hayashi et al. 2008). In a review of toxicant impacts on density-limited populations, Forbes et al. (2001) noted that the full range of interactions have been found between toxicant stress and density dependence, including less than additive, additive, and more than additive effects. Also, the type of effect may vary with increasing toxicant concentration from one that ameliorates density dependent effects at low toxicant concentrations to one that exacerbates density dependent effects at higher toxicant concentrations. Case studies which incorporate density-dependence into population modeling demonstrate this variability, with overall impacts to populations shown to be both lesser (Van Kirk and Hill 2007) and greater (Hayashi et al. 2008) than the level of effect that would be predicted from individual response depending on the situation. In time, density-dependant populations may rebound, stabilize at a lower absolute population number, or continue to decline until the population is extirpated (Forbes et al. 2001). Modeling exercises have demonstrated cases in which populations stabilize at new, lower equilibrium abundances in response to a constant impact (van Kirk and Hill 2007; Spromberg and Meador 2005).

A species' likelihood of persistence can also be estimated a number of ways. There are no standard methods or protocols to estimate the risk of extinction. Instead, the method used is usually dependent on the availability of data available on the species in question and species' biology. Extinction risk analyses methodologies may be qualitative, semi-quantitative, or quantitative. One quantitative method that is used widely for modeling a species' time to extinction or probability of extinction is Population viability analysis (PVA). PVAs use simulation modeling to identify threats to species and to assess the vulnerability of populations to extinction risks. These models incorporate demographic parameters such as fecundity, survivorship, age structure, and population size, but can also incorporate effects to the environment such as habitat degradation and catastrophic events. As for the evaluation of population growth rate, sensitivity analysis is used to determine which factors have the greatest impact on population persistence, and many experts feel that parsing out these influential factors for management purposes is the best utilization of these models, as opposed to absolute predictions of population decline. However, PVA models require a depth of demographic data that is often lacking for listed species.

For Pacific salmon, NMFS has not found a PVA that completely represents the various risks facing salmon populations (McElhany et al. 2000). Consequently NMFS created the viable salmonid population concept to provide useful benchmarks for evaluating actions that directly affect natural populations and for which incremental increases in extinction risk may be difficult or impossible to accurately quantify. Where PVAs have been conducted for specific populations of salmon, these have informed NMFS in status assessments. While the VSP concept isn't meant to replace quantitative models where they can be properly used because the VPS employs a combination of quantitative and qualitative methods for determining the extinction risk of listed species it is more flexible and easier to use where data are limited (McElhany et al. 2000). Under the VSP approach, risk is first addressed at the population level and then the ESU. Individual populations are assessed according to four parameters: abundance, growth rate/productivity, spatial structure, and diversity. NMFS focuses on these parameters because they are reasonable indicators of extinction risk (viability). Although, there is no formal link between VSP and jeopardy under Section 7, the same population level parameters used in VSP, whenever available, are a significant part of our analysis in determining whether an agency's action is likely to jeopardize the continued existence of a listed species.

Summary of Population Responses to Reductions in Fecundity and Juvenile Survival

Modeling and experimental studies have shown that chronic toxicity to pollutants can lead to population decline and extirpation. Variation in population-level responses to environmental toxicity can be expected among species as a consequence of factors like species life history strategies, life stage affected, density dependence, and magnitude of toxicant stress. Although the degree varied among different life history strategies, fecundity and juvenile survival remained a highly influential demographic parameter throughout modeled scenarios, with adult survival taking on greater importance in long-lived species. These results must be coupled with other influences on the population status, such as the degree of density dependence and additional environmental perturbations such as catastrophes. Although population modeling often requires more demographic information than is available for threatened and endangered species, careful selection of surrogates and use of their data may allow for extrapolation from models for species with similar life histories.

Summary of the Direct Effects

According to our analysis, Chinook, chum, coho, and sockeye salmon, and green and shortnose sturgeon exposed to cyanide are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above. Our analysis demonstrates that acute and chronic toxicity may be exacerbated by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column. That is, the threshold of adverse effects is diminished in the very cold waters and low dissolved oxygen conditions.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Because no data for cyanide toxicity to sturgeon exist, LC50 values for sturgeon were derived from the 5% SSD concentration for the class Actinoptergyii, which encompasses all known cyanide toxicity data for fish. From this data, we developed quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 49). Given the limited data set, our estimates are the same for green sturgeon and shortnose sturgeon, as well as sockeye salmon and chum salmon (the latter are based on data from the genus *Oncorhynchus*). Based on our analysis, we estimate that green and shortnose sturgeon exposed to cyanide at the CCC may experience a reduction in juvenile survival that is as high as, but not likely to be greater than, 56%. Our estimates reveal that the green and shortnose sturgeon may experience a reduction in the number of hatched eggs and that reduction could be as high as, but is not likely to be greater than, 30%.

Similarly, we expect that coho and Chinook salmon would experience a reduction of juvenile survival and that reduction could be as much as, but is not likely to be greater than 79%. We estimate that, when exposed to cyanide at the CCC, coho and Chinook salmon may experience a reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 45%. Similarly we expect that chum and sockeye salmon would experience a reduction in juvenile survival and that reduction could be as much as, but is not likely to be greater than 69%. We estimate that, when exposed to cyanide at the CCC, chum and sockeye salmon may experience a reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 37%. Our estimates reveal that steelhead would experience a reduction in juvenile survival and that reduction could be as much as, but is not likely to be greater than 61%. We estimate that, when exposed to cyanide at the CCC, steelhead may experience a reduction in the number of hatched eggs and that reduction could be as high as, but is not likely to be greater than, 33%.

Young of the year fish, and juvenile fish that do survive exposure to cyanide could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. We expect that such exposure could also delay reproductive maturity and productivity. These reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole, and could be measured in terms of changes in population growth rates and changes in risk of extinction.

Sturgeon have naturally high adult survival, and the loss of juvenile life stages is particularly problematic. Several authors have suggested that the rate of survival may be so high that management at the levels of these age classes is unlikely to improve their survival or increase population growth rate (Gross et al 2002; Heppell 2007). As such, recovery efforts are often based upon increasing survival in juvenile age classes. Gross et al (2002) modeled population growth rates for three species of sturgeon that varied in life history traits such as size, lifespan, age to maturity, and migration. All three sturgeons showed similar elasticity profiles, and thus the authors concluded that general interpretation could be applied to sturgeon across species. In contrast to other elasticity profiles for long-lived species, elasticity in sturgeon was highest in individual young-of-the-year and juvenile age classes, dropped at the onset of maturity, and continued to decline for each successive adult age class. Fecundity had relatively low elasticity, as the effects of changes in fecundity are shared among all adult age classes of these long-lived species, and the value of changes to egg numbers is lessened by the high mortality of the young-of-the-year age class. The authors concluded that population growth rate would show little response to improvements in fecundity, but greater responses in survival at either the young-of-the-year or juvenile age classes. However, since survival of the juvenile and adult age classes is naturally high, improvements at these stages will have smaller effects to improving population growth rate than increases to survival of young-of-the-year, when natural mortality is greater. The authors note that among biologists and managers involved in sturgeon conservation, habitat improvement was regarded as the most important conservation undertaking for sturgeon. Results from this study indicate that restoration efforts should target the survival of age classes with high elasticity, specifically young-of-year and juvenile. Paragamian and Hansen (2008) drew similar conclusions in modeling effects on population growth of the Kootenai River white sturgeon. The authors found that subadult and adult survival (>90%) was much higher than that of juveniles (40% in the first year), and recovery was dependent on increasing first-year survival. The authors suggested that to have the largest effect on recovery, managers should increase the current targeted recruitment rate.

Unlike sturgeon, most Pacific salmon (with the exception of steelhead and cutthroat trout) are semelparous, such that they spawn only once. Consequently, reductions in the number of viable eggs and juvenile survival through their first year would likely have greater population-level effects on Chinook, coho, sockeye, and chum salmon. Low fresh water is survival is considered typical of most salmon populations, although estimates for many populations are nonexistent, mortality rates are recorded from 40-90% (Sandercock 1991; Bradford 1997). According to Brandford, the coefficient of variation (CV) for interannual survival in fresh water is about 30% averaged over all species. The factors that influence the freshwater survival rate for the likely differs somewhat between widely-dispersed spawning species (e.g., steelhead, coho and Chinook salmon) compared to those that spawn in dense aggregations (e.g., sockeye and chum salmon), as well as the length of time spent in freshwater rearing (e.g., coho salmon versus early migrant Chinook salmon or chum salmon). For Pacific salmon, mortality appears to be roughly equally divided between fresh water and marine waters, suggesting that each habitat contributes to recruitment variation (Bradford 1997). Consequently, significant reductions in freshwater production would be expected to significantly affect the number of adults returning to fresh water to spawn.

As discussed earlier, there are several factors that can influence the relative toxicity of chemical contaminants under natural exposure conditions. When organisms are stressed due to environmental factors outside their normal optima they may become more sensitive to a given toxicant. This can occur when homeostasis is disrupted in organisms that are infected with a pathogen, outside their normal range for various water quality parameters (salinity, pH, or temperature), diseased, or debilitated due to other toxic insults. Very cold temperatures and low DO conditions increase the toxicity of cyanide. Despite the limited number of studies on these influencing factors, until more work can be done we have little evidence to suggest species specific responses to cyanide under low DO conditions or low water temperatures. Considering that cyanide is a respiratory toxin that inhibits oxidative metabolism, it is not surprising that the effects are exacerbated under conditions where oxygen availability is limited. Any factor that affects gill ventilation will also likely affect the amount and speed at which the toxin is distributed in the body. A fish is under stressed conditions like oxygen depletion, would typically increase their ventilation rate to compensate for the low DO and would, in this situation also increase their rate of uptake of aqueous cyanide.

In summary, exposure to aqueous cyanide at the approved CCC and CMC is likely to lead to the fitness consequences for Chinook, coho, sockeye, and chum salmon, and green and shortnose sturgeon. In particular, exposure to cyanide concentrations at the chronic criterion could substantially reduce reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survival of young fish through the first year.

Sturgeon and salmon may also experience effects on growth, swimming performance, condition, and development. While sturgeon have developed a life history that allows them to cope with low survivorship to maturity and occasional decreases in recruitment, these adaptations are unlikely to compensate for a constant reduction in both fecundity and early life stage survival. The reductions we estimate in survival of young fish through the first year in particular would substantially decrease survival and recovery of this species. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed.

Based on our analysis, we expect that the proposed action would significantly reduce the absolute numbers of green sturgeon, shortnose sturgeon, Chinook salmon, chum salmon, sockeye salmon, coho salmon, and steelhead. Based upon the magnitude of effects we anticipate could occur, the distributions of green and shortnose sturgeons are likely to be reduced in waters where they are exposed to cyanide at the levels defined by the chronic criterion, and may be reduced when cyanide exposure overlaps with low water temperatures or low DO concentrations.

Critical Habitat

We evaluated the effect of EPA's approval of the cyanide water quality standards on the effect of critical habitat by first reviewing the essential features or primary constituent elements of critical habitat for listed and proposed designations. Based on our analysis, the primary features that may be affected by EPA's approved water quality criteria are those designated as "water quality" areas for growth, development and reproduction (salmon and green sturgeon). We evaluated the "water quality" feature according to whether the acute or chronic criteria were likely to reduce the amount of clean water available for supporting essential patterns of growth, development or reproduction.

Approval of the CCC in state water quality standards would allow states to manage cyanide in waters to these levels. Even if waters never systematically reached these levels, the use of the aquatic life criteria in NPDES permits, TMDL limits, indicates the importance that these numeric values play in the overall success and operation of the water quality program. Our analysis demonstrates that where cyanide concentrations reach the approved standard, the proposed action would likely adversely affect the quality of water to the degree that it would impair individual reproduction and survival of green sturgeon, Chinook salmon, coho salmon, chum salmon, sockeye salmon and steelhead, and would cause these species to experience adverse effects to growth, swimming performance, condition, and development. For green sturgeon, we estimate the reduction in the number of hatched eggs could be as high 48% and the reduction in the survival of young fish through the first year as high as 56%. For coho and Chinook salmon, we estimate the reduction in the number of hatched eggs could be as high 45% and the reduction in the survival of young fish through the first year as high as 79%. For chum and sockeye salmon, we estimate the reduction in the number of hatched eggs could be as high 37% and the reduction in the survival of young fish through the first year as high as 69%. For steelhead, we estimate the reduction in the number of hatched eggs could be as high 33% and the reduction in the survival of young fish through the first year as high as 61%. These effects are estimated to be of a magnitude great enough to reduce numbers of both sturgeon and salmon. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth would be severely reduced, and sturgeon and salmon may be extirpated from critical habitat containing cyanide at approved values. Not only would impacts to water quality resulting from management of cyanide to the CCC diminish the ability of critical habitat to provide for conservation of the these species, our analysis also suggests that the conservation value of critical habitat for these species would likely be diminished at concentrations below EPA's recommended CCC for fresh water.

The Impacts of Reduced Salmon Populations – Summary of Indirect Effects

Salmon are a significant contributor to the overall ecological food web throughout their range, whether they are from listed populations or unlisted populations. Two significant indirect effects of the proposed action, attributable not to the direct toxicity of cyanide, but the action's impact on Chinook, coho, sockeye and chum salmon and steelhead population abundance would include the further loss of primary prey species for southern resident killer whales and Cook Inlet beluga whales, and the loss of salmon nutrient transport to freshwater systems, which indirectly affects their own productivity. Bilby et al. (1996) demonstrate that juvenile and older age classes of salmon grow more rapidly with the appearance of spawners because these younger fish will feed on eggs and spawner carcasses. Salmon carcasses strewn along river reaches and streambanks are a significant source of food to a wide number of animals and affect the overall productivity of nutrient-poor systems (Bilby et al. 1996; Cederholm et al. 2000). The loss of these "marine derived nutrients" likely significantly reduces the survival of their own species, particularly in nutrient poor streams. Bilby et al. (1996) demonstrated that the mean fork length of juveniles and up to 45% of the carbon in cutthroat trout and 40% of the carbon in young of the year coho comes from the decaying carcasses of the previous generation of salmon. The increased body size is directly correlated to increases in over winter survival and marine survival. Based on historical cannery records and current records of escapement, Gresh et al. (2000) estimate this nutrient source has declined to about 13 to 17 percent of the historic biomass of return salmon to Pacific Northwest streams (Washington, Oregon, Idaho, and California). They suggest that this loss is one important indicator of ecosystem failure, contributing to the observed reductions in abundance we have seen in many salmon populations, and could further hamper recovery efforts. Thus, while we may have estimated the direct loss of individuals attributable to the proposed action, further reductions in many populations would be expected as adult spawner numbers decline from reduced recruitment attributable to the proposed action.

Similarly, although not obligate feeders, southern resident killer whales feed primarily on salmon and salmon are seasonally an important prey for Cook Inlet beluga whales. The reductions in salmon populations anticipated as a result of this action can be expected to have significant affects on southern resident killer whales and their critical habitat, and Cook Inlet beluga whales and their proposed critical habitat. Based on killer whale stomach contents from stranded whales and field observations of predation, Ford et al. (1998) determined that 95% of the diet of resident killer whales consists of fish, with a significant portion being Chinook salmon (about 2/3 of the samples that were identified to species). The authors suggested that Chinook salmon may be preferentially hunted by killer whales because of their large body size, high fat content, and seasonal distribution patterns. Although, Cook Inlet beluga whales feed on a variety of other fish species Pacific salmon are an important prey species for these animals as they build their lipid body stores essential to their winter survival. The significant reduction in the southern resident killer whale's primary prey species, Pacific salmon in general and in particular Chinook salmon, from the proposed action is likely to have significant effects on the fitness of southern resident killer whales and their population viability. As noted earlier, a 50% reduction in killer whale calving has been correlated with years of low Chinook salmon abundance (Ward et al. 2009a). Cook Inlet beluga whales would similarly experience a significant reduction in their most abundant summer and fall prey species (most of which, are non-listed Chinook, coho, sockeye, and chum species, although some listed species may be consumed during their marine migrations to Alaska). The proposed action, based on our analysis would significantly reduce freshwater production of all listed salmon species, as well as non-listed salmon species where cyanide concentrations are allowed to reach EPA's recommended aquatic life criteria concentrations. As noted earlier, we expect the proposed action would cause as high as a 79% reduction in the survival of juvenile (young fish through their first year) Chinook salmon, and as high as a 45% reduction in the number of viable eggs. These losses would severely reduce the number of adult Chinook salmon in the Puget Sound ESU, and would reduce the forage base for southern resident killer whales. Southern resident killer whales are not restricted to Puget Sound, but do spend a large portion of time in Puget Sound, the Strait of Juan de Fuca, and Haro Strait. Prey losses would also be realized throughout their range, including Oregon and California. Consequently, we expect that the proposed action would significantly reduce the absolute numbers of southern resident killer whales by reducing the absolute numbers of their primary prey. Based upon the magnitude of effects estimated to salmon, we expect the numbers, distribution and reproduction of southern killer whales would likely to be reduced due to significantly a reduced forage base.

Similarly, we expect the proposed action would cause as high as a 79% reduction in the survival of juvenile (young fish through their first year) coho salmon, as high as a 69% reduction in the survival of juvenile sockeye and chum salmon, and as high as a 44% reduction in the number of viable coho salmon eggs, and as high as a 37% reduction in the number of viable sockeye and chum salmon eggs. These losses would severely reduce the forage base of Cook Inlet beluga whales, and as a result we expect that the proposed action would significantly reduce the absolute numbers of Cook Inlet beluga whales by reducing important prey species. Based upon the magnitude of effects estimated to salmon, we expect the numbers, distribution and reproduction of Cook Inlet beluga whales would likely be reduced due to a significantly a reduced forage base.

Critical Habitat of Southern Resident Killer Whales

We evaluated the effect of EPA's approval of the cyanide water quality standards on the effect of critical habitat by first reviewing the essential features or primary constituent elements of critical habitat for listed designations. Based on our analysis, the primary features that may be affected by EPA's approved water quality criteria are those designated as "prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth." Based on our analysis, we estimate that coho and Chinook salmon will experience reductions in the number of hatched eggs as high 45% and the reduction in the survival of young fish through the first year as high as 79%. For chum and sockeye salmon, we estimate the reduction in the number of hatched eggs could be as high 37% and the reduction in the survival of young fish through the first year as high as 69%. For steelhead, we estimate the reduction in the number of hatched eggs could be as high 33% and the reduction in the survival of young fish through the first year as high as 61%. These effects are estimated to be of a magnitude great enough to reduce numbers of these listed salmon species. Approval of the CCC would adversely affect the quality of water to the degree that normal salmon population growth would be severely reduced, and salmon may be extirpated from areas containing cyanide at approved values. These losses would severely diminish the ability of critical habitat to provide for conservation of the southern resident killer whales.

Proposed Critical Habitat of Cook Inlet Beluga Whales

We evaluated the effect of EPA's approval of the cyanide water quality standards on the effect of critical habitat by first reviewing the essential features or primary constituent elements of critical habitat for the proposed designation for Cook Inlet beluga whales. Based on our analysis, the primary features that may be affected by EPA's approved water quality criteria are those primary prey species consisting of Chinook, coho, sockeye, and chum salmon. Based on our analysis, we estimate that coho and Chinook salmon will experience reductions in the number of hatched eggs as high 45% and the reduction in the survival of young fish through the first year as high as 79%. For chum and sockeye salmon, we estimate the reduction in the number of hatched eggs could be as high 37% and the reduction in the survival of young fish through the first year as high as 69%. These effects are estimated to be of a magnitude great enough to reduce numbers of these salmon species. Approval of the CCC would adversely affect the quality of water to the degree that normal salmon population growth would be severely reduced, and salmon may be extirpated from areas containing cyanide at approved values. These losses would severely diminish the ability of critical habitat to provide for conservation of Cook Inlet beluga whales.

Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this biological opinion. In this section we focus on the status and trends of land-uses across the United States and the consequences of those land uses for listed and proposed resources. Since our action area encompasses a very broad spatial scale, we focused on key properties of ecosystem condition and the actions that influence those properties. According to the Consultation Handbook (USFWS and NMFS 1998), the "reasonably certain to occur" clause may include such indicators of actions such as approval of an action by a state, tribal or local agencies or government; indications that granting authorities for the action are imminent; project sponsor's assurance that actions will proceed, etc. Although speculative non-federal actions are not factored into the analysis, at the same time "reasonably certain to occur" does not require a guarantee that an action will occur, therefore a degree of uncertainty is acceptable when characterizing cumulative effects.

Due to the scale at which a national consultation occurs, the degree of uncertainty increases, particularly with respect to anticipating the cumulative effects of future non-federal actions across the action area. We necessarily relied on types of human activity (e.g., regional trends and projections in population increases, and associated industrial and commercial development) as proxies for the suite of hydrological, chemical, and biological changes that would reasonably be expected in the surrounding landscape. Metrics of land use (e.g., percent impervious or urbanization; road density) are strongly correlated to a variety of ecological indicators of stress (e.g., changes in aquatic community; increases in chemical constituents, physical stream-channel condition; NRC 2008). Based on our knowledge of past changes within a watershed and the effects landscape changes have had on aquatic ecosystems, we can anticipate the general types and patterns of future land uses will have on the physical, chemical and biological conditions of downstream waterways. The specific factors that are important within a specific locality will vary from place to place, and over time.

The information we present herein is based on data produced by recognized organizations using demographic data, and economic and labor statistics and include their reasoned rough-trend estimates of population and economic change stemming from these data. Changes in the near-term (5-year; 2013) are more likely to occur than longer-term projections (10-year; 2018). Because the anticipated effects are based upon projections that are subject to error and alteration by complex economic and social interactions, our analysis does not address small or localized changes in aquatic habitats. Further, since the effects of future federal actions that are unrelated to the proposed action are not to be considered herein because they require separate consultation pursuant to Section 7, wherever possible, we eliminated known or typical future federal actions from our analysis (e.g., construction of new oil platforms). Many of the actions we discuss herein, such as construction and industrial development, are planned, approved and permitted through wholly local and state approvals and with private funds. However, in many instances we found it impossible to differentiate between non-federal and federal actions, and therefore we erred on including a general type of action in our analysis recognizing that a portion may qualify as federal actions and would not normally be included in our cumulative effects analysis. For example, transportation projects may be undertaken by local and state entities, and others may qualify as federal actions for reasons of federal funding, permitting, etc. In this instance, we were unable to discern federal transportation related actions from future non-federal transportation-related actions therefore we focused on general patterns we might in various regions and the generalized impacts of transportation projects on water quality.

Sources queried for the information herein include the United States Census Bureau, Department of Labor, and Lexis-Nexis information system. With the latter (which was our source for state legislation), we reviewed bills passed in 2007 to 2008 and pending bills under consideration were included as further evidence that actions “are reasonably certain to occur”. Bills that died in process or were vetoed are not included in our review.

Northeast Projection

We began our review for each region by examining current and pending state legislation for regional and local policy and political trends that may impact future development and management directions within the area. For instance, we looked for regulatory and political impetus for changes in zoning, fisheries, environmental standards, and development of commerce and industry. For the Northeast, we selected Maine as a representative state for this effort because of the extent of coastline and waterways, as well as the presence of habitat for several listed species from different taxa. We found that legislation in the state shows tendencies towards controlling invasive species, chemical (wastewater, pesticide, oil, nutrients, bacteria, and other toxic contamination) and sedimentation impacts humans have on rivers and nearshore waters, emissions associated with global warming, and the ability of fish to migrate past river infrastructure. As a general matter, we expect that other coastal states within this region likely have programs or interests engaged in many similar activities, many of which are designed to minimize some of the adverse effects associated with increasing development and extraction industries.

In general, the northeast region is one of the most densely populated regions in the United States. Based upon 2000 United States census data, the northeast United States was predicted to contain 54.8 million people in 2005, and population growth is predicted to decrease over the foreseeable future from 0.41%/year between 2000 and 2010 to 0.24%/year from 2010 to 2020 (USCB 2005a). Much of the regional population is contained in concentrated metropolitan centers. If these cities were to continue to grow at the rate which they did from 2000 to 2007 (USCB 2008), the largest growth will occur in Dover, DE (2.89%/yr), Washington, D.C. metro (1.51%/year), and York-Hanover, PA (1.47%/year). The only population center greater than one million people growing at greater than one percent per year is Washington, D.C. Overall, the northeast United States is predicted to have 55.8 million people in 2010, 56.6 million in 2015, and 57.1 million in 2015. Growth of metropolitan centers will increase discharge of wastewater from water treatment systems into rivers and streams, which will increase the loads of contaminants carried by these waterways to the marine environment, and would have concomitant effects on such parameters as biological oxygen demand, chemical oxygen demand, DO, and water temperature. It is likely that development will continue along the coast and waterways, which will add sediment to river systems and potentially alter spawning habitat. Oil and other roadway pollutants may increase as a result of additional vehicular traffic. Additional recreational use of lakes, waterways, and coastal areas will increase fish takes and add additional discharges from vessels.

Industrial changes can indirectly add pressures to ESA listed species' survival and the health of their habitats. From 2006 to 2016, output of the mining industry is expected to increase by 1.0%/year (Figueroa and Woods 2007), which is a 25% decline in growth from what it was between 1996 and 2006. However, technological advancements will likely increase output in this sector. It should be noted that 60% of this industry is comprised of oil and natural gas, very little of which exists in the northeast United States. Coal output is likely to increase with demand for power through the electrical grid. Most significantly for the northeast, metal mining is anticipated to increase 4.3%/year with demand by various technologies and rising metal process. Currently, granite, peat, roofing slate, iron ore, sulfur, magnetite, manganese, copper, zinc, mica, and precious metals are mined in the region, with numerous others on an infrequent or historical basis (see baseline for additional information). Increasing output by existing and new mines can place additional pressures on species recovery in the foreseeable future by increasing waste runoff into streams and rivers.

Nationwide, construction is forecasted to be one of the most extensively growing industries in the United States. From 2006 to 2016, the construction industry is expected to grow by 1.4%/year and employ an additional 600,000 people during that time (Figueroa and Woods 2007). However, this represents a 30% slow-down from the 1996 to 2006 time period. Construction will be most likely to occur in school, industrial, and medical areas, as well as infrastructure (bridge and road) repair and replacement. An increase in construction will entail additional development in urban and non-urbanized areas that can introduce large amounts of sediment into waterways via run-off, altering riverine habitat relied upon by salmonids. Sediments can also reduce water clarity and food availability resulting from loss of primary productivity. Sediment run-off can also introduce nutrients into marine environments that can cause algal blooms, which have been documented in nearshore habitats of the northeast United States, and introduce neurotoxins to large areas and cause wide-scale mortality (Vitousek et al. 1997).

Output of the transportation industry is expected to increase by 2.9%/year from 2006 to 2016 (Figueroa and Woods 2007), placing additional pollution pressures on listed species and their habitats. Although this rate is slower than the trend from 1996 to 2006, additional movement of freight by truck, plane, and train introduces pollutants, especially oils, to waterways that can increase petroleum concentrations in streams and estuaries. Greater potential for moderate- to large-scale pollutant release by spills and accidents also exists. Carbon dioxide released from petroleum combustion is a significant component of global warming (Vitousek et al. 1997; Nordhaus 2007; EIA 2007) and increases in the transportation will likely mean greater contributions of carbon dioxide and exacerbation of the global warming phenomenon. Based upon these factors, additional recovery pressures are likely to occur from the future growth of the transportation industry.

With increasing population, the leisure and hospitality industry is forecasted to grow by 2.1%/year from 2006 to 2016 (Figueroa and Woods 2007). As with other industries, this is a decline from the 1996 to 2006 rate by about 25%. In addition, most growth will likely occur in food services or drinking places, which is not expected to have impacts to listed species. However, this industry includes personnel and activities that utilize natural and protected areas. Additional use will likely include more debris and pollution discharge into areas frequently used by protected species. It can be contended that additional use of parks can increase outreach and public awareness of protected species and their habitats, which can benefit recovery of these species and areas. It is not known whether growth in the leisure and hospitality industry will have a net positive or negative impact on ESA listed species, but likely will include both helpful and hurtful aspects.

In contrast to other industries, agriculture is forecasted to increase in rate of growth from 2006 to 2016 versus the growth experienced from 1996 to 2006 (Figueroa and Woods 2007). Growth will increase from 1.3%/year to 2.2%/year, a change of roughly 75%. The increase results from increased efficiency from technological improvements and the rise of ethanol from crops. In this sector, agriculture accounts for over 80% of production, which masks regionally important factors. Agriculture in the northeast overshadows a projected output decline in forestry (-0.9%/year) and fisheries/hunting/trapping (-2.9%/year). Agriculture is not as extensive as in other regions of the United States and growth. However, additional growth will increase pollution and sediment runoff into streams, placing additional stress on salmon habitat and making bloom conditions more likely in marine areas where rivers discharge. Based upon the declines in fisheries and forestry, it is unlikely that extensive additional pressures will be placed on ESA listed species recovery by these two industries.

Southeast and Mid-Atlantic Projection

State legislation frequently shows regional and local policy and political trends that can significantly impact future directions within the area. Florida was selected as an example of legislative trends in the mid-Atlantic and Gulf of Mexico because of the extent of coastline, presence of diverse and numerous listed species, socio-economic similarities to other states, large population, and progressive tendencies. Here, legislative regulation is moving towards management of beaches, control of watersheds and vessel discharges, protecting marine resources, restoration of freshwater habitats, identifying issues and contributing factors to climate change, limitation of oil and gas development, and lowering harmful chemical inputs into systems.

Mid-Atlantic states (including Florida) are predicted to increase in population from 55.7 million people in 2005 to 59.8 million in 2010 and 64.0 million in 2015. This is the fastest rate of anticipated regional growth in the nation except for western states (USCB 2005b). The rate of regional growth is anticipated to remain above 10% through 2030 and will be greatest in Florida and North Carolina and lowest in West Virginia. Although this region includes a larger area than the northeast, urban growth is much more extensive in the mid-Atlantic; 12 metropolitan areas experienced population growth of 3%/year or greater from 2000 to 2007, including the Atlanta area, once considered the most rapidly developing area in human history. However, half of these urban centers were in Florida. Cities of over one million people that grew at a rate of 1%/year or greater from 2000 to 2007 included Raleigh, NC (4.49%/year), Atlanta, GA (3.47%/year), Charlotte, NC (3.44%/year), Orlando, FL (3.37%/year), Jacksonville, FL (2.27%/year), Tampa-St. Petersburg, FL (1.96%/year), Richmond, VA (1.51%/year), and Miami, FL (1.16%/year). This rapid and concentrated population increase places much larger demand upon natural systems. Wastewater systems must handle larger loads of sewage. As soil is covered by asphalt and concrete, run-off must be channeled into local stormwater drains increasing contaminant load in streams. Regional areas of development are frequently in low-elevation locations, limiting water retention and movement. Both of these are sources of concern for sediment and contaminants entering local waterways and flowing into rivers, estuaries, and nearshore marine habitats.

Economic development will contribute additional pressures to ESA-listed species of the mid-Atlantic region. West Virginia is mined extensively for coal and demand for this resource to meet the needs of coal-fired power plants will drive increasing production (Figueroa and Woods 2007). Production of North Carolina's cement constituents, Georgian clay, and Florida's phosphate rock are likely to increase with demand in other sectors, such as construction. These and other mining sources can produce excessive sedimentation in streams as well as affect pH and metal concentrations. Expansion or increased production from regional mines is expected to have increased negative impacts to freshwater systems, estuaries, and bay systems in the foreseeable future.

Changes in the leisure and hospitality, transportation, and construction sectors are likely to have similar effects in the mid-Atlantic as were identified for the northeast. However, regional differences will likely lead to different local effects. Low-lying estuaries can collect oil and contaminant run-off from rapidly developing roads, leading to habitat degradation.

The mid-Atlantic region has significantly greater agriculture than in the northeast; a difference that will likely affect the health of streams, estuaries, and marine habitats. Extensive agriculture in the region requires the use of pesticides, fertilizers, and other chemicals in large scale that migrate into freshwater systems. The expansion of agriculture, regardless of crop, will likely entail additional chemicals entering freshwater systems. This can have negative impacts on the survival and recovery of sturgeon populations in fresh water and bay systems, both by accumulation in fish tissues, and general degradation of habitat (i.e., Chesapeake Bay).

West Coast Projection

For the west coast, we selected California as a state representative in legislation. This is because of the large population, complex geography, diverse socio-economic and demographic structure, extent of waterway and coastline, and presence of several listed species of varied taxa. Trends in legislation address the impact and causal regulation of climate change, control of marine debris and harmful substances in waterways and marine areas, regulation of fisheries and invasive species, limitation of oil and gas development, clarification of state listed species takes, and aid for salmon recovery.

States along the Pacific coast, or which contribute water to major river systems here, are projected to have the most rapid growth of any area in the United States within the next few decades. This is particularly true for coastal states and those of the desert southwest. California, Oregon, Washington State, Arizona, Idaho, Utah, Nevada, and Alaska are forecasted to have double digit increases in population growth rates for each decade from 2000 to 2030 (USCB 2005b). New Mexico, Montana, and Wyoming will have far slower growth, with Wyoming forecasted to eventually experience population contraction. Overall, this region had a projected population of 65.6 million people in 2005 and will likely grow to 70.0 million in 2010 and 74.4 million in 2015, making it by far the most populous region (but also containing the greatest land area). As with other regions, growth stems from development of metropolitan areas. However, western growth will come generally from enlargement of smaller cities than from major metropolitan areas. Of the 42 metropolitan areas that experienced 10% growth or greater between 2000 and 2007, only seven have populations greater than one million people. These major cities include Las Vegas, NV (4.79%/year), Phoenix, AZ (4.07%/year), Riverside-San Bernadino-Ontario, CA (3.63%/year), Sacramento-Arden-Arcade-Roseville, CA (2.34%/year), Salt Lake City, UT (1.93%/year), Denver, CO (1.87%/year), and Portland-Vancouver-Beaverton, OR (1.83%/year). It should be noted that most of these metroplexes border coastal or riverine systems. Diffuse, but extensive, growth in the region will increase contaminants from wastewater treatment plants and sediments from sprawling urban and suburban development that enter riverine, estuarine, and marine habitats. This is of particular concern in western states, where numerous rivers and their tributaries are designated critical habitat for listed salmon. Increased contaminant loads have the potential to influence fry and smolt development in freshwater systems. Sediments may alter spawning grounds so as to make them unusable by salmon. Unlike other areas of the United States, the west coast region has extensive fluctuations in elevation and pooling oil and pollutants from developing roadways will likely not be as significant an issue in this region as elsewhere. Western states are widely known for scenic and natural beauty. Increasing resident and tourist use will place additional strain on maintaining the natural state of park and nature areas, also utilized by protected species.

Mining has historically been a major component of western state economies. With national output for metals increasing at 4.3% annually (little oil, but some gas is drawn from western states), output of western mines should increase markedly (Figueroa and Woods 2007). This will increase already significant levels of mining contaminants entering river basins. This future increase is all the more problematic because many western streams feed into or provide spawning habitat for threatened and endangered salmonid populations. These fishes rely upon healthy streams for breeding and their offspring spend the first parts of their lives feeding in rivers, lakes, and streams that heavier contaminant burdens will be affecting. Sturgeon also live in these waterways and will similarly experience negative impacts from growth in the mining sector.

Western states boast large tracts of irrigated agriculture. The rise in agricultural output (Figueroa and Woods 2007) will likely result in two negative impacts upon protected species. With increased production, pesticide, fertilizer, and herbicide use will be used in greater amounts and enter freshwater systems in greater concentrations. Like mining, this has the potential to harm salmonids and sturgeon or their habitats. Further, increased output could place greater demands upon limited water resources. This will reduce flow rates and alter habitat throughout freshwater systems, and likely lead to increased water temperatures and decreases in DO. As water is drawn off, contaminants will become more concentrated in these systems, exacerbating contamination issues in habitats and protected species.

Summary of Cumulative Effects

At the large spatial scale of this consultation, we could not identify specific future state, tribal, local or private actions that were reasonably certain to occur in the action area. Instead we looked at demographic and economic trends to discern general patterns of land use change anticipated by states and federal organizations and their potential effects on listed species. Assuming recent increases in unemployment and poor performance of the dollar are fair indicators of rates potential land use change, regional growth is expected to continue on a slower pace than observed in the past decade. In January 2010, however, unemployment dropped a modest amount from 10 percent to 9.7 percent, which may signal a shift to a more promising economy. However, much uncertainty surrounds whether we will see near term measurable increases in the construction and industrial arenas. We suspect that spatial patterns of growth and development, and redevelopment would likely continue as it has in the past for the near future, but expect that the pace of new development and redevelopment will continue to remain at a slower pace than the past decade.

In general, we expect that the threatened and endangered aquatic species and designated critical habitats considered in this biological opinion are likely to be adversely affected by non-federal activities that affect the quantity, and quality of water, waterways, and habitats important to listed aquatic species and their critical habitat. Non-federal activities that change vegetative cover, soil structure, and water use ways that increase erosion and sedimentation, increase introduction of pollutants into waterways, and result in introductions and spread of non-native invasive species will likely continue to directly and indirectly affect listed species and critical habitats. These species and their critical habitats could also be affected by illegal harvest. At the same time, states or private entities may also engage in activities to restore, enhance, and improve water quality and quantity and restore more natural hydrographic patterns that benefit listed species and their habitats. All of the species and critical habitats considered in this document are likely to be exposed to these types of activities in the future to varying extents.

Integration and Synthesis

The U.S. Environmental Protection Agency proposes to approve state or tribal water quality standards, or federal water quality standards promulgated by EPA, that are identical to or more stringent than EPA's recommended 304(a) aquatic life criteria for cyanide. This approval would authorize states and tribes and EPA to establish source controls (e.g., permits, 401 certifications, waste load allocations, etc.), define and allocate control responsibilities (allocate loads under TMDLs), measure and enforce compliance with the CWA, and measure progress in meeting the goals of the CWA (whether a water body should be listed as impaired; see *Understanding the Water Quality Program* earlier in this Opinion for a summary of the activities that are influenced by or rely upon the water quality standards approved by EPA and implemented by states, tribes and EPA.

In the *Approach to the Assessment* section of this Opinion, NMFS explained that we would assess the effects of EPA's programmatic approval of state, tribal, and federal water quality standards that rely upon their nationally recommended 304(a) aquatic life criteria for cyanide at the CCC and the CMC, by asking:

> Is EPA's approval of state, tribal and federal water quality standards consistent with (or more stringent than) the 304(a) criteria for cyanide, likely to prevent the exposure of endangered species, threatened species, and designated critical habitat to aqueous cyanide concentrations that are toxic, given the approach EPA uses to approve a water quality standards?

If, after considering the best scientific and commercial data available, we conclude that listed resources are not likely to be exposed to activities the water quality standards would authorize, both individually and cumulatively, we stated we would conclude that EPA's proposal to continue recommending the 304(a) aquatic life criteria for cyanide is not likely to jeopardize the continued existence of endangered species, threatened species, or result in the destruction or adverse modification of designated critical habitat under NMFS' jurisdiction. When an agency's national action is likely to prevent exposure of listed resources to their activities, then we would expect an agency's program would generally ensure that actions taken under the program are not likely to individually, or cumulatively, jeopardize the continued existence of threatened and endangered species, and are not likely to result in the destruction or adverse modification of critical habitat that has been designated for those species.

If our assessment determined that listed resources are likely to be exposed to these activities, we stated we would examine whether and to what degree listed species are likely to respond to their exposure, given the approach EPA uses to approve a water quality standards. As part of this analysis, we stated we would examine whether and to what degree EPA has identified chemical, physical and biological scenarios that influence cyanide toxicity and presence in the environment inhabited by listed species and their critical habitat, the nature of any in situ effects, and the consequences of those effects for listed resources under NMFS' jurisdiction, to determine if EPA can insure that the approval of state, tribal and federal water quality standards that they are proposing is not likely to jeopardize the continued existence of endangered species or threatened species, or result in the destruction or adverse modification of critical habitat that has been designated for these species. We stated that we measure risks to listed individuals using changes in the individual's "fitness" or the individual's growth, survival, annual reproductive success, and lifetime reproductive success. When we do not expect listed plans or animals exposed to an action's effects to experience reductions in fitness, we would not expect that action to have adverse consequences on the viability of the populations those individuals represent or the species those population comprise (Mills and Beatty 1979; Stearns 1992; Anderson 2000). As a result, if we conclude that listed plants or animals are not likely to experience reductions in their fitness we would conclude our assessment.

Based on the analysis contained in their BE and on the results of the preliminary screen as introduced by the *Methods Manual*, EPA was able to screen out (or make not likely to adversely affect) determinations on all but 32 species. The 32 species included: several darters, perch, salmonids, and one amphipod. Next, EPA applied a secondary screen that relied primarily on evaluating whether the waters where the 32 listed species occurred were listed as impaired pursuant to the CWA as well as data that would indicate the species had been (1) listed for reasons attributed to cyanide, (2) or whether there were known dischargers of cyanide within the range of the listed species. Using these metrics EPA concluded that of the 32 potentially sensitive species, none would be adversely affected by their action of approving state or tribal water quality standards or federal water quality standards that are equal to or more stringent than the nationally recommended section 304(a) aquatic life water quality criteria for cyanide.

Based on data available in STORET and TRI, as well as information about cyanide in general, the patterns of cyanide exposure are variable and probably not reflective of only permitted discharges. A number of non-permitted (non-point) sources likely also contribute to ambient cyanide concentrations in waters of the United States. Since state, tribal and federal water quality standards form the foundation for, not only permitting, but also evaluating the measuring the progress of the goals of the CWA, it is important to consider non-point sources of a contaminant in evaluating exposure scenarios. Our analysis also demonstrates that permitted discharges likely exceed criterion values from time to time, and can be as much as ten times higher than criterion values without being in violation of CWA. Because we lacked long term data sets for our analysis, we could not evaluate an upper exposure limit nor do we know what a typical exposure scenario would necessarily look like. Our analysis demonstrates that all listed species considered herein would likely be exposed to cyanide during the course of their typical life histories. However, because we could not determine the typical concentrations of exposure, our analysis is premised on the assumption that a suitable concentration for evaluating exposure and response are the proposed criteria values. We believe this is a reasonable threshold for evaluating the effects of cyanide at the national level, since it forms the foundation for a host of water quality management actions in waters of the United States and is the basis for EPA's proposed approval of state, tribal and federal water quality standards.

Our analysis demonstrates that EPA may identify chemical and biological scenarios that influence cyanide toxicity and presence in the environment, but that such information often has little influence (or at least no obvious influence) on the concentration of cyanide that EPA recommends to states and tribes as a "safe dose" for water quality standards. Since the information relegated to "other data" is not considered at the national level in publishing a 304(a) recommendation, then we looked for information to suggest that states would use the information to modify their water quality standards to incorporate site or situation specific modifications as appropriate. That is, we found no evidence that states adopted cyanide water quality standards that were modified by expected water temperatures, unless it was to increase the accepted concentration of the cyanide standard. For instance, since the cyanide standard is driven by rainbow trout data, states with warm water basins often increased the threshold of their water quality standard. In contrast, states where cold water species (e.g., steelhead and salmon) reside did not have modified standards for winter (very cold) water situations that account for the increased toxicity of cyanide at cold temperatures.

In general for cyanide, EPA's decision to recommend and approve water quality standards for cyanide was based on a paucity of data in general, and in particular for listed species. The paucity of data was particularly apparent for saltwater species. However, data was also extremely poor for characterizing a few good case studies on cyanide or what might be considered typical cyanide exposures. Based on our limited review of a few general permits, which incidentally, happen to be one of the most routinely issued permit types issued by EPA and states, generally too few samples are required to result in meaningful monitoring data by which to manage cyanide discharges, or to evaluate the frequency and severity of cyanide entering most basins.

EPA's strict interpretation of what they deemed adequate data for the purposes of decision-making under the CWA is also particularly disconcerting. While both EPA and NMFS are required to use the best available data in their decision-making, when there is data on the listed taxa despite whether there are numerous studies that confirm the findings, NMFS would generally consider that data and the strength of the data in its decision. For instance, EPA often narrowly constrains their decision on a criterion to "avoid confounding factors". However, what might be considered a "confounding factor" in a laboratory setting is often a realistic mixture of conditions in the wild and is relevant for the purposes of evaluating whether a particular action or set of actions is not likely to jeopardize the continued existence of listed resources. For instance, the interplay between DO and cyanide or cyanide and temperature received little attention in EPA's 304(a) aquatic life criteria, despite that there is a wide problem of low DO in many watersheds inhabited by anadromous fish species both on the west coast and the east coast, and salmonids generally inhabit very cold waters during winter months. At least with cyanide, EPA's decision-making process is based on limited very controlled test situations that may be poor predictors of real exposure scenarios and at a minimum, would be strengthened by some field experiments or at least mesocosm studies that are more representative of typical aquatic communities.

Based on our analysis, it also appears that guidance to states and tribes may be prudent for recognizing the potential impacts of cyanide, and the ability of the various forms of cyanide to interact and change within a system. Although we did not search for specific examples of guidance, sources of cyanide within a watershed are numerous and are not limited to expected dischargers and certainly are not limited to the mining industry, which is often the misconception. Based on a review of wastewater treatment facilities, Kavanaugh et al. (2003) caution that managers need to acknowledge that multiple forms of cyanide typically coexist, introconvert, and degrade in a waterbody. It is for this reason, that Kavanaugh et al. (2003) recommended that water quality standards ought to reflect the ability of cyanide compounds to undergo transformation, increasing or decreasing in impact; in so doing, EPA could establish water quality standards for certain classes of cyanide that would be measured using appropriate analytical methods. Kavanaugh et al. (2003) also recommend that the water quality criteria and discharge standards for cyanide be revised to ensure that monitoring methods can distinguish between cyanide forms, and that methods with the greatest potential for use should receive EPA and state approval.

Nevertheless, based upon our analysis we concur with EPA's effect determination that a number species are not likely to be adversely affected when exposed to cyanide at criterion values. Our determination, however, is based on uncertain evidence because for the most part suitable data upon which to make this determination is weak at best. As noted earlier, Gensemer et al. (2007) declined to evaluate the effects of several marine species, acknowledging that the data is too poor to evaluate the protectiveness of the saltwater cyanide criteria on marine species. We concur with Gensemer et al. (2007) that "this represents an area requiring further research" since only three fish genera and five invertebrate genera were used to establish the saltwater criteria. That said, based on the available data as discussed in the preceding analysis, we would not expect the following threatened or endangered species to respond physically, physiologically, or behaviorally to exposure at the CMC or the CCC, whether exposed in saltwater or fresh water or both: Blue whales, bowhead whales, fin whales, humpback whales, North Atlantic right whales, North Pacific right whales, sei whales, sperm whales, beluga whales, southern resident killer whales, Guadalupe fur seals, Hawaiian monk seals, Western Steller sea lions, Eastern Steller sea lions, Florida green sea turtles, Mexico green sea turtles, hawksbill sea turtles, Kemp's ridley sea turtles, loggerhead sea turtles, leatherback sea turtles, Mexico's breeding colonies of olive ridley sea turtles, other olive ridley sea turtles, smalltooth sawfish, elkhorn coral, staghorn coral, white abalone, black abalone and Johnson's seagrass.

Species under NMFS' jurisdiction that demonstrate sensitivity to cyanide at criterion values are: chum salmon, coho salmon, sockeye salmon, Chinook salmon, steelhead, shortnose sturgeon, and green sturgeon, representing 30 DPS/ESUs of these species. Of these species, empirical and modeled evidence suggests that some salmon may die when exposed to cyanide at the CMC of 22 µg/L. According to modeled estimates chum, coho, Chinook, and sockeye salmon, are all more sensitive to cyanide than steelhead, suggesting that some individuals may die when exposed to cyanide at the CMC. However, lethal effects on steelhead salmon are predicated on an exposure to cyanide at low temperatures. That is, the risk of death increases at lower temperatures, while exposure to cyanide in waters at about the average test temperature of 12-13 ºC would probably not lead to the death of steelhead.

While, the relationship between temperature and cyanide may merit further examination to increase confidence in the relationship, existing information suggests that coldwater species may be more sensitive to cyanide at temperatures that are typical of winter months. We have no evidence that the interplay between cyanide and temperature is species specific. If temperature influences the sensitivity of other salmonids, then that would increase the risk of death for not only steelhead, but also coho, sockeye, chum, and Chinook salmon. Our best estimate of effect for steelhead is that roughly 1% of steelhead exposed to cyanide in winter months may die from their exposure, since coho, Chinook, sockeye and chum salmon are all more sensitive to cyanide than steelhead, the percent lethal effect would also increase. We do not know which ages or stages of salmon are most likely to be affected at low temperatures.

Based on our review of chronic studies, we estimate that female sturgeon and Pacific salmon may experience a 40-60% reduction in the number of eggs spawned, and these species would experience a 40 to 70 % reduction in early life stage survival. This should only be considered a rough estimate of the magnitude of the true effect expected at the CCC of 5.2 µg CN/L. Other sublethal responses to low levels of cyanide include reduced swimming performance and reduced weight gain.

In the *Status of the Species* section of this Opinion, we established that Chinook, coho, sockeye, and chum salmon, steelhead, and green and shortnose sturgeon species have declined throughout their range. Some ESUs have demonstrated modest increases in recent years, like Lower Columbia River Chinook salmon and Hood Canal chum salmon, and others like Sacramento winter-run Chinook salmon, Puget Sound steelhead, and Lower Columbia River coho salmon continue to decline. For some ESUs like California coastal Chinook salmon and Central California coast coho salmon, current trends are unknown.

In the *Environmental Baseline* section of this Opinion, we established that salmon and sturgeon are exposed to a myriad of habitat alterations attributable to urban and agricultural development, as well as fishing pressure. Land-use patterns have a profound impact on the contribution of chemicals to the waterways where salmon, steelhead, and sturgeon migrate, rear, spawn, feed and grow. In many basins, these fish are exposed to persistent “legacy” chemicals, as well as there is a relatively constant influx of common-use chemicals like copper and PAHs. At the same time, migratory barriers continue to impact population movement and expansion, loss of riparian forest has lead to increased water temperatures in some areas and the loss of allochthonous input, reduced stream bank complexity, loss of spawning gravels, and altered flow regimes, to name a few. Salmon and sturgeon are also commonly impacted by low DO in many areas throughout their ranges. In the *Cumulative Effects* section of this Opinion, we established that salmon and sturgeon are likely to be exposed to the combined effects of similar habitat modifications for the next ten years, and given expected human population increases and economic development in many regions these impacts will likely increase. The combined effect of these habitat alterations means that chemical loading in many watersheds and coastal areas will likely continue to increase, despite pollution control efforts. Non-point sources for pollutant loading will likely continue to be a significant portion of the problem.

Killing 30-45% of the viable eggs spawned per salmon and sturgeon and killing 56-79% of their larvae is certain to reduce the likelihood of survival and the reproductive success of coho salmon, Chinook salmon, chum salmon, sockeye salmon, green sturgeon, and shortnose sturgeon populations. Reducing the swimming performance of these species would likely reduce their fitness and possibly their survival, through reductions in prey capture, weight gain, displacement, predator escapement, and possibly lead to death. Although there is uncertainty in this analysis, which incidentally is not limited to these calculations, based on the evidence available, we do not believe EPA’s decision-making process mitigates or minimizes these potential losses. Worse yet, EPA and the states are not in a position to detect these losses if or when they occur.

If the intent of the 304(a) aquatic life criteria is to define a level in the waterbody of a pollutant that will be fully protective of the designated uses of a water body and that a state or tribe identify as part of their water quality standards (see BE page 11, and also 40 CFR 131.2), then it would follow that EPA would have to review whether their recommended criteria can protect the specific uses that states and tribes have identified in their designated uses. Instead, our analysis suggests that Gaba (1983) was correct when he noted that EPA and the states are engaged in a water quality process “merely to justify the specific numbers contained in pollutant criteria.” That uses are designated without meaningful linkages between the chemical criteria indicators and the biological condition of the waters they are meant to protect, means neither EPA or states or tribes can know how well the chemical criteria are protecting the aquatic assemblages or biological community diversity they are meant to protect. That is, available evidence suggests that EPA (nor states or tribes) is not likely to monitor (a) the direct, indirect, and cumulative impacts of the activities their approvals would authorize on biological community diversity, (b) the nature of those effects on the aquatic assemblages in which they occur, or (c) the consequences of those effects on listed resources. Given the lack of measured endpoints for biological condition, EPA will not know if the aquatic assemblages or species identified as designated are actually protected by the water quality standards, much less whether those water quality standards protect endangered species, threatened species or designated critical habitat under NMFS’ jurisdiction.

Based on our review, it is not even clear that EPA would consider listed species as part of the biological community to which Congress directed them to consider in establishing 304(a) aquatic life criteria. EPA’s decision-making process (the *Guidelines*) places special emphasis on commercially, recreationally, and other important species, and aquatic assemblages. If, as EPA stated, their only metrics for evaluating the protection of the aquatic assemblage are species richness and species evenness (see EPA 2008a), then EPA could argue (albeit a poor argument) that they are protecting aquatic assemblages if their recommended aquatic life criteria and approved state water quality standards protect non-native aquatic assemblages. Yet, listed species, arguably, are “important” as Congress saw fit to provide for their protection under the ESA and ensure federal agencies have a prominent role in providing for their protection. Moreover, many of NMFS’ listed species are also commercially and recreationally valued, and many of the species discussed herein are part of the same aquatic assemblage. Given, EPA’s lack of clarity on what constitutes an “important” species, and the indicators they stated they use to evaluate an aquatic assemblage (species richness and species evenness) EPA has placed themselves in a position to exclude the needs of native species in general, and listed species in particular, as part of the biological communities they intend to protect.

All of the endangered species, threatened species, and designated critical habitat under NMFS’ jurisdiction depend upon the health of the aquatic ecosystems they occupy for their survival and recovery. EPA’s 304(a) aquatic life criteria are designed to reflect the latest scientific knowledge including on the kind and extent of all identified effects on …. fish, shellfish, wildlife, and plants…which may be expected from the presence of pollutants in any body of water…; the concentration and dispersal of pollutants or their byproducts, through biological, physical and chemicals processes; and on the effects of pollutants on biological community diversity, productivity, and stability….. (CWA section 304(a)(1)). As such, 304(a) aquatic life criteria have a prominent role in the success of the overall water quality program designed “to restore and maintain the chemical, physical and biological integrity of the Nation’s waters.”

Nevertheless, degraded water quality has been one of the contributing factors for the decline of almost all of the anadromous fish species NMFS has listed since the mid-1980s. While cyanide has not been identified as a specific concern in any listing, poor water quality has generally been identified as cause contributing to their need for listing. Generally, it has not been the case that NMFS has isolated poor water quality to only one chemical, physical, or biological stressor for the species that have been listed. To use this lack of evidence, as evidence that an effect is lacking is simply not a persuasive argument that cyanide is not problem for listed species.

Based on our analysis we believe it is reasonable to expect that the number of cyanide sources is likely to increase commensurate with land use changes and expansion of industrial and extraction activities. Our analysis illustrates that the exposure of listed salmon and sturgeon species to cyanide at the proposed chronic criterion concentration is likely to substantially reduce their reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and by reducing the survivorship of young fish in their first year. These fish may also experience effects on growth, swimming performance, condition, and development. Based upon the magnitude of adverse effects caused by the exposure of these listed species to cyanide at the proposed criteria concentrations, these fish species are likely to become extirpated from waters where they are exposed to approved cyanide discharges that are compliant with approved water quality standards. Continued approval of the EPA's aquatic life criteria for cyanide at the range wide scale of these listed species is likely to reduce their reproduction, numbers, and distribution. Unfortunately, it appears that not only does EPA fail to consider biologically, chemically, and physically relevant exposure scenarios that influence cyanide toxicity, EPA is not and has not put themselves in a position of knowing whether their 304(a) aquatic life recommendations and subsequent approvals of state and tribal water quality standards are in fact, protecting the biological community diversity, productivity and stability they intend to protect. Therefore, we do not believe the EPA can insure that the approval of water quality standards for cyanide are not likely to jeopardize the continued existence of endangered species or threatened species or result in the destruction or adverse modification of critical habitat that has been designated for these species.

Because the proposed action, based on our analysis, is likely to reduce the viability of one or more populations throughout the range of listed Pacific salmon, steelhead, and sturgeon species, we expect that the action is likely to reduce the viability (that is, increase the extinction probability or appreciably reduce their likelihood of both surviving and recovering in the wild) of the listed species as a whole. The specific listed species at risk are: California coastal Chinook salmon, Central Valley spring-run Chinook salmon, Lower Columbia River Chinook salmon, Upper Columbia River spring-run Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal summer-run chum salmon, Central California Coast coho salmon, Lower Columbia River coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, southern green sturgeon, shortnose sturgeon, Lake Ozette sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Middle Columbia River steelhead, Northern California steelhead, Puget Sound steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California coast steelhead, Upper Columbia river steelhead, and Upper Willamette River steelhead.

Finally, a reduction in Puget Sound Chinook salmon would in turn significantly reduce the forage base of southern-resident killer whales. Therefore, while we agree that southern resident killer whales are not likely to respond physically, physiological, or behaviorally to their direct exposure to cyanide at the CCC or the CMC, we expect that the action, through indirect effects to their primary prey, Pacific salmon, is likely to appreciably reduce the likelihood of southern-resident killer whales surviving and recovering in the wild. Similarly, a reduction in Chinook, coho, sockeye, and chum salmon would in turn significantly reduce the forage base of Cook Inlet beluga whales. We also agree with EPA that Cook Inlet beluga whales are not likely to respond physically, physiological, or behaviorally to their direct exposure to cyanide at the CCC or the CMC, we expect that the action, through indirect effects to their primary prey, Pacific salmon, is likely to appreciably reduce the likelihood of Cook Inlet beluga whales surviving and recovering in the wild.

The proposed action is likely to reduce the habitat qualities for these species that are essential to their conservation. Specifically, reduced availability of clean quality water for the purpose of reproduction, rearing and growth, and a reduction in prey species of sufficient quantity and quality would affect the conservation value of designated critical habitat for these species. The functional value of critical habitat exposed to cyanide at criterion values would be severally reduced and could not serve the intended conservation role for the species. Based on our analysis, the functional value of critical habitat would be reduced throughout the areas designated as critical habitat for: southern resident killer whale, California coastal Chinook salmon, Central Valley spring-run Chinook salmon, Lower Columbia River Chinook salmon, Upper Columbia River spring-run Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal summer-run chum salmon, Central California Coast coho salmon, Lower Columbia River coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, southern green sturgeon, Lake Ozette sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Middle Columbia River steelhead, Northern California steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California coast steelhead, Upper Columbia river steelhead, and Upper Willamette River steelhead. Similarly, the proposed action would significantly reduce the functional value of proposed critical habitat for Cook Inlet beluga whales when their salmon prey species are exposed to cyanide at criterion values. The result of the exposure of salmon species outside of the geographic area designated as critical habitat would severally reduce the numbers of salmon available to beluga within proposed critical habitat and therefore, the critical habitat could not serve the intended conservation role for the species.

Conclusion

Listed Species and Critical Habitat

After reviewing the current status of the listed species, the environmental baseline for the action area, the effects of the EPA's continuing approval of state water quality standards that rely on their nationally recommended criteria for cyanide and the cumulative effects, it is NMFS' biological opinion that EPA's approval of state water quality standards for cyanide is likely to jeopardize the continued existence of the following species:

> California coastal Chinook salmon, Central Valley spring-run Chinook salmon, Lower Columbia River Chinook salmon, Upper Columbia River spring-run Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal summer-run chum salmon, Central California Coast coho salmon, Lower Columbia River coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, southern green sturgeon, shortnose sturgeon, Lake Ozette sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Middle Columbia River steelhead, Northern California steelhead, Puget Sound steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California coast steelhead, Upper Columbia river steelhead, Upper Willamette River steelhead, southern resident killer whales, and beluga whales.

After reviewing the current status of the listed species, the environmental baseline for the action area, the effects of the EPA's continuing approval of state water quality standards that rely on their nationally recommended criteria for cyanide and the cumulative effects, it is NMFS' biological opinion that EPA's approval of state water quality standards for cyanide is likely to destroy or adversely modify designated critical habitat for the following species:

> Southern resident killer whale, California coastal Chinook salmon, Central Valley spring-run Chinook salmon, Lower Columbia River Chinook salmon, Upper Columbia River spring-run Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal summer-run chum salmon, Central California Coast coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, southern green sturgeon, Lake Ozette sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Middle Columbia River steelhead, Northern California steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California coast steelhead, Upper Columbia river steelhead, and Upper Willamette River steelhead.

For species that have no designated critical habitat, then none can be affected.

Species and Critical Habitat Proposed for Listing

After reviewing the current status of bocaccio, canary rockfish, spotted seal, and yelloweye rockfish, the environmental baseline for the action area, the effects of the EPA's continuing approval of state water quality standards that rely on their nationally recommended criteria for cyanide and the cumulative effects, it is NMFS' conference opinion that EPA's approval of state water quality standards for cyanide is not likely to jeopardize the continued existence of bocaccio, canary rockfish, spotted seal, and yelloweye rockfish. NMFS' conclusion for these proposed species is based on the limited data available on marine species. Based on the foregoing analysis, NMFS expects that the approval of cyanide water quality standards is likely to destroy or adversely modify the proposed critical habitat for beluga whales because salmon are an important prey species for beluga whales and are identified as a PCE. NMFS' conclusion for the area designated as proposed critical habitat for Cook Inlet beluga whales is based on the proposed action's effects on salmonids.

Reasonable and Prudent Alternatives

This Opinion has concluded that EPA's approval of state or tribal water quality standards, or federal water quality standards promulgated by EPA for aquatic life criteria that are identical the section 304(a) aquatic life criteria for cyanide, is likely to jeopardize the continued existence of 31 species under NMFS' jurisdiction, and result in the destruction or adverse modification of critical habitat that has been designated for these species. The clause "jeopardize the continued existence of" means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of listed species in the wild by reducing the reproduction, numbers or distribution of that species (50 CFR §402.02).

Regulations implementing Section 7 of the Act (50 CFR 402.02) define reasonable and prudent alternatives as alternative actions, identified during formal consultation, that: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) would, NMFS believes, avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat.

NMFS reached this conclusion because the evidence available suggests that EPA does not (a) use biological, chemical, or physically relevant information of the natural conditions to which aquatic species would be exposed to derive their numeric recommendations for 304(a) aquatic life criteria or to approve state and tribal water quality standards that rely on their recommended criteria, (b) that EPA is not in a position to know whether the water quality standards they approve actually protect native biological communities, or (c) the listed species that are part of the native biological community. Given the decision structure employed by EPA, EPA will not know whether designated uses are protected, much less whether the direct, indirect, or cumulative impacts of their approval of state and tribal water quality standards that rely on their

304(a) aquatic life criteria recommendations protect endangered species, threatened species, or designated critical under NMFS' jurisdiction.

To satisfy its obligation pursuant to section 7(a)(2) of the ESA of 1973, as amended, EPA must put itself in a position to (a) use biological, chemical, or physically relevant information of the natural conditions to which aquatic species would be exposed to derive their numeric recommendations for 304(a) aquatic life criteria or to approve state and tribal water quality standards that rely on their recommended criteria, (b) monitor whether the water quality standards they approve actually protect native biological communities, and (c) the listed species that are part of the native biological community. What follows is a single reasonable and prudent alternative, consisting of several sub-elements that must be implemented in its entirety to insure that the activities EPA's approval of state and tribal water quality standards would authorize are not likely to jeopardize endangered or threatened species under the jurisdiction of the NMFS or destroy or adversely modify critical habitat that has been designated for these species.

The U.S. Environmental Protection Agency must, by December 1, 2012:

A). Revise the *Guidelines* and any relevant regulatory guidance to:

1. Address how they will incorporate relevant information on biological, chemical, or physical processes that alter a particular chemical's toxicity in nature, in their recommendations such that states and tribes that adopt 304(a) aquatic life criteria as recommended will be required to account for relevant exposure scenarios that affect chemical toxicity, in their state water quality standards.

2. Explicitly address (a) endangered species, threatened species, and designated critical habitat as part of the "important" species the aquatic life criteria are designed to protect, and (b) the native biological community, of which listed species are a part, as the relevant community endpoint to which they intend to protect.

B). Develop and implement the research necessary to replace modeled estimates of species sensitivities to cyanide with direct evidence, using listed species or more closely related surrogates, as the basis for defining cyanide criteria to insure an appropriate level of protection is afforded to listed species and critical habitats addressed by this RPA.

Because this biological opinion has concluded that the U.S. Environmental Protection Agency's proposed approval of state water quality standards that rely on their 304(a) aquatic life criteria is likely to jeopardize the continued existence of endangered and threatened species under the jurisdiction of NMFS, and is likely to result in the destruction or adverse modification of critical habitat, the Environmental Protection Agency is required to notify NMFS of its final decision on the implementation of the reasonable and prudent alternatives.

Incidental Take Statement

Section 9 of the ESA and Federal regulation pursuant to section 4(d) of the ESA prohibits the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the Act provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Take

As described earlier in this Opinion, this NMFS' review of EPA's national approval of state and tribal water quality standards that are consistent with or more stringent than the nationally recommended 304(a) criteria for cyanide. The goal of this national level Opinion is to evaluate the general impacts to NMFS' listed resources from the national approval of the 304(a) cyanide criteria when adopted by states and tribes for implementation as part of their water quality standards. It is not possible to identify take that would occur from specific permitted actions or the specific exposure scenarios typical in a particular state. Instead, this Opinion anticipates the general effects that would occur from the approval of cyanide water quality standards across the landscape. Therefore, this Opinion does not exempt incidental take of listed fish from the prohibitions of section 9 of the ESA for the EPA's approval of cyanide water quality standards.

NMFS anticipates that with implementation of the RPA, incidental take of the listed species considered in this biological opinion is not likely to occur from exposure to cyanide at revised criteria concentrations. However, other elements of water quality standards could allow for exceedance of criteria concentrations and may result in incidental take. The other elements of water quality standards will be the focus of subsequent tiered consultations on individual state and tribal water quality standards. In each of these instances, EPA must conduct a separate, tiered consultation, and if necessary NMFS would issue a separate biological opinion before any endangered or threatened species might be "taken"; the amount or extent of "take" would be identified in those subsequent consultation on site-specific, state or tribal specific, or permit specific activities. Therefore, no incidental take exemptions are provided in this programmatic biological opinion.

Conservation Recommendations

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultation involving EPA's approval of state water quality standards:

1. The EPA should work with states to develop more meaningful linkages between designated uses and the water quality standards they intend to protect, to create monitoring programs that are capable of actually evaluating whether designated uses are being protected by approved water quality standards.

In order to keep NMFS' Endangered Species Division informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the United States Environmental Protection Agency should notify the Endangered Species Division of any conservation recommendations they implement in their final action.

Reinitiation Notice

This concludes formal consultation on the United States Environmental Protection Agency's approval of water quality standards that are identical to or are more stringent than the section 304(a) cyanide aquatic life criteria. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of authorized take is exceeded, the United States Environmental Protection Agency must immediately request reinitiation of section 7 consultation.

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
WASHINGTON HABITAT BRANCH OFFICE
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May 4, 2007

OWW-135

Mr. Mike Gearheard
Director, Office of Water and Watersheds
U.S. Environmental Protection Agency, Region 10
(OWW130)
1200 Sixth Avenue
Seattle, WA 98101

Dear ~~Mr. Gearheard~~ Mike:

The State of Washington Department of Ecology (Ecology) has recently issued a Public Notice Draft National Pollution Discharge Elimination System (NPDES) Industrial Stormwater General Permit for public review and comment. The National Marine Fisheries Service (NMFS) offers the following brief comments on the proposed permit pursuant to our role as providers of biological and technical assistance under the Endangered Species Act of 1973 (16 U.S.C. 1531 *et seq.*), as amended (ESA), and the Fish and Wildlife Coordination Act (16 U.S.C. 661 *et seq.*). We are sending these comments to you because of the Environmental Protection Agency's (EPA) acknowledged oversight role in objecting to or approving the issuance of this permit under Section 402(d) of the Clean Water Act (CWA), and acknowledged responsibility to comply with Section 7(a)(2) of the Endangered Species Act (ESA) (as stated in the National Association of Home Builders v. Defenders of Wildlife, et al. and EPA v. Defenders of Wildlife, et al. argument before the Supreme Court). In addition, these comments are provided per the processes outlined in the Memorandum of Agreement between the EPA and the NMFS regarding enhanced coordination under the CWA and ESA (hereafter "MOA") (May 22, 2001, 66FR 11202-11217).

With the CWA authority delegated from the EPA, Ecology proposes to reissue the Industrial Stormwater General Permit to over 1,100 industrial facilities in Washington State, revoking and replacing the current permit. The permit relies heavily on the Permittees' diligent implementation of the permit requirements to result in compliance with state water quality standards. In addition the permit retains the concept of benchmarks and action levels (levels of industrial contaminants that will require the permittee to take further actions).

The geographic area covered by the permit overlaps the range of 14 federally listed threatened or endangered salmon, as well as designated critical habitat for 13 of these populations. The permit area overlaps areas addressed by the Puget Sound Shared Strategy Recovery Plans, Lower Columbia River Fish Recovery Board, the Upper and Mid-Columbia Fish Recovery Boards, the Governor's Salmon Plan, and the Puget Sound Partnership. Most of these plans have identified stormwater runoff as a significant factor in reaching salmon recovery. In addition, the Puget Sound Partnership has developed recommendations for addressing stormwater effects with the goal of achieving a healthy Puget Sound by the year 2020.

We support Ecology's objectives in permitting this large number of industrial facilities, which will reduce the discharge of contaminated stormwater from industrial activities into receiving waters, and help protect fish and wildlife resources including threatened and endangered species. However, we believe that the Draft Industrial Stormwater General Permit, as currently proposed, will authorize stormwater discharges that have more than a minor detrimental effect on federally listed salmon and their critical habitat.

NMFS bases this conclusion on the body of scientific evidence which identifies that these discharges are likely to produce water quality conditions that result in negative behavioral and physiological consequences for these species, leading to reduced viability of populations exposed to those conditions. This point is illustrated in a previous comment letter that NMFS Headquarters provided to the EPA Headquarters regarding the issuance of the national multi-sector general permit (MSGP) for stormwater discharges associated with industrial activities (attached). Especially relevant to Ecology's permit is the section in the NMFS letter describing the likely consequences on Pacific salmon of issuing the MSGP for industrial activities. This section examines the effects of metals, chemical mixtures, turbidity, water quantity, etc., on salmon, and NMFS concludes that more than minor detrimental effects on salmon and their prey base will occur.

Also in this section of the NMFS comment letter, some metals including dissolved copper are identified as being particularly problematic for salmon. Further support for this conclusion is provided by a recent NMFS paper entitled "Technical White Paper on Dissolved Copper's Effects on Juvenile Salmonid Sensory Systems" (attached). The paper concludes that benchmark concentrations (calculated using EPA methodology) ranging from 0.18 to 2.1 $\mu g/L$ of dissolved copper in fresh water result in reductions of 8 to 57 percent in predator avoidance by juvenile salmon. Since the proportion of dissolved copper in stormwater discharges may be quite high relative to total copper concentrations, we do not believe that the proposed benchmark value of 11.9 $\mu g/L$ total copper and an action level of 23.8 $\mu g/L$ total copper are adequate to protect salmon in fresh waters.

Adverse impacts to the estuarine, riverine and marine waters resulting from inadequate regulatory protection will lead to direct and indirect adverse impacts to salmon and their habitats including spawning and rearing areas. Over the 5 years the proposed permit will be in effect, the proposed permit standards are likely to contribute to a direct loss of salmon and their habitat and exacerbate water quality problems in adjacent waters. We do not believe that this proposed permit will do enough to achieve the goal of the Puget Sound Partnership for a healthy Puget Sound by the year 2020. Similarly, while it represents an improvement over the status quo, it does not go far enough towards the goal of salmon recovery for the 14 listed populations as described in various salmon recovery plans in the State.

Because we fully support development of an effective permit, we look forward to working further with EPA to minimize the effects of the industrial stormwater permit and the discharges authorized under the permit to waters that contain listed species and their designated critical habitat.

Sincerely,

Steven W. Landino
Washington State Director
for Habitat Conservation

Attachments: (2)

cc: Dave Peeler, Ecology
Ken Berg, USFWS