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Temporal effects of enhanced fine sediment loading on macroinvertebrate community structure and functional traits

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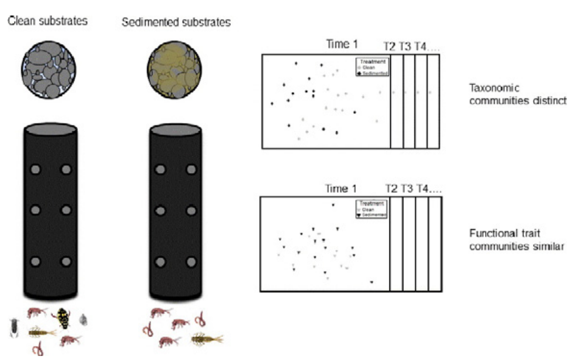
HIGHLIGHTS

- Fine sediment effects on macroinvertebrate traits and assemblages were examined.
- Analysis of taxonomic community structure identified strong fine sediment effects.
- Effects of sediment loading on community structure were not temporally consistent.
- Faunal traits performed poorly in characterising fine sediment effects.
- Taxon life cycles probably influence the effect of fine sediment load.

GRAPHICAL ABSTRACT



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ABSTRACT

Deposition of fine sediment that fills interstitial spaces in streambed substrates is widely acknowledged to have significant negative effects on macroinvertebrate communities, but the temporal consistency of clogging effects is less well known. In this study the effects of experimentally enhanced fine sediment content on aquatic invertebrates were examined over 126 days in two lowland UK streams. Taxonomic approaches indicated significant differences in macroinvertebrate community structure associated with sediment treatment (clean or sedimented substrates), although the effects were variable on some occasions. The degree of separation between clean and sedimented communities was strong within seven of the nine sampling periods with significant differences in community composition being evident. EPT taxa and taxon characterised as sensitive to fine sediment demonstrated strong responses to enhanced fine sediment loading. Faunal traits also detected the effects of enhanced fine sediment loading but the results were not as consistent or marked. More widely, the study highlights the temporal dynamics of sedimentation effects upon macroinvertebrate communities and the need to consider faunal life histories when examining the effects of fine sediment loading pressures on lotic ecosystems.

1. Introduction

Increased instream fine sediment loading is widely regarded as a global threat to ecological integrity and lotic ecosystem health, often leading to reduced macroinvertebrate diversity through direct exclusion of taxa, enhanced drift or reductions in the availability of suitable trophic resources and habitat (Larsen and Ormerod, 2010a; Jones et

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al., 2012; Wood et al., 2016). The infiltration of fine sediment into the river (colmation/clogging) has been reported to modify benthic macroinvertebrate community structure and functioning (Descloux et al., 2013). Substrates characterised by a high proportion of fine sediment are frequently dominated by taxa with low dissolved oxygen requirements (Angradi, 1999; Zweig and Rabeni, 2001) and exhibit an absence of taxa vulnerable to fine sediment due to impairment or damage of filter-feeding apparatus or delicate gills (Wood and Armitage, 1997; Larson et al., 2009). In addition, some taxa may be excluded and unable to colonise habitats where excessive fine sediment is present, for example due to the absence of suitable materials for case building by caddisfly larvae (Higler, 1975; Urbanič et al., 2005). Some functional feeding groups may also be disadvantaged by enhanced fine sediment loading, associated with reduced food quality or impaired access to food resources, notably for algal scrapers and filter feeders (Rabeni et al., 2005; Kreutzweiser et al., 2005). This may lead to shifts in community structure towards those dominated by deposit feeders (Relyea et al., 2012).

Some fauna respond to fine sediment deposition pressures as a function of their morphological characteristics and functional traits (Lamouroux et al., 2004; Bona et al., 2016; Doretto et al., 2017). Recently there has been a growing focus on the incorporation of faunal traits within biomonitoring tools to elucidate on the changes that occur to invertebrate community structure in freshwater ecosystems (Menezes et al., 2010; Göthe et al., 2016; Pilière et al., 2016). Biological traits are based on the habitat model concept (Southwood, 1977), and therefore community traits may reflect spatial and temporal variations in environmental factors (Townsend and Hildrew, 1994). Trait composition can also be used to identify sources of environmental impairment associated with anthropogenic and natural stressors which act as 'filters', selecting taxa with relevant adaptive traits. Consequently, some traits may be particularly sensitive to environmental pressures and it is this possibility that has led to the increasing application of biological traits within biomonitoring tools (Statzner et al., 2004; Friberg, 2014; Turley et al., 2016). However, relatively little information exists regarding how macroinvertebrate faunal traits respond to instream fine sediment loading and the limited studies in this area to date have yielded variable results (e.g. Buendia et al., 2013; Descloux et al., 2014).

The majority of studies conducted on sedimentation to date have focussed on artificial enhanced fine sediment loads (Suren and Jowett, 2001; Larsen et al., 2011) or have been associated with heavily sedimented river beds (Matthaei et al., 2010; Wagenhoff et al., 2012). A small number of studies have experimentally manipulated the volume of fine sediment within the substrate directly through the application of faunal colonisation devices, but these studies have typically examined the effects at a single point in time (Bo et al., 2007; Larsen et al., 2011; Pacioglu et al., 2012; Descloux et al., 2013, 2014). There is an absence of research that considers the temporal variability of fine sediment effects on macroinvertebrate communities and the value of life history traits for understanding and monitoring these effects.

Species phenology within a community affects the composition of macroinvertebrates observed at differing times of the year (Delucchi and Peckarsky, 1989; Murphy and Giller, 2000), and may confound biomonitoring assessments if not acknowledged (Clarke, 2013; Carlson et al., 2013). Temporal and spatial heterogeneity of hydrological regimes is also a fundamental process in shaping riverine macroinvertebrate communities (Dewson et al., 2007; Monk et al., 2008). Natural streams are typically characterised by stable baseflow conditions punctuated periodically by flow disturbances. These flow disturbances have important implications for fine sediment dynamics, initiating entrainment of fine material stored in the channel and increasing suspended sediment concentrations (Leopold et al., 1964; Bond and Downes, 2003). The interaction between flow and fine sediment dynamics (entrainment, suspension and depositional processes) has been identified as a primary factor which influences the turnover of taxa within macroinvertebrate communities (Rempel et al., 2000; Buendia et al., 2014; Jones et al., 2015). Consequently, as a result of temporal variability in flow and species assemblages, it follows that it is important to consider the effects of sediment loading over time.

This study is the first to specifically consider the temporal variability of experimentally manipulated fine sediment loading on macroinvertebrate communities at a fine temporal resolution (weeks). The following research questions were addressed:

(i) Is the effect of increased fine sediment loading on macroinvertebrate communities consistent temporally?
(ii) Which taxa and functional traits are associated with enhanced fine sediment loading?
(iii) Are the observed effects of enhanced fine sediment loading on macroinvertebrate communities evident and consistent for both taxonomic and faunal trait compositions?

2. Materials and methods

2.1. Field sites

The study took place on two small lowland rivers in Rutland, UK; the River Gwash (52°38′ N, 00°44′ W) and the River Chater (52°37′ N, 00° 44′ W). Sites were selected to be as broadly comparable in physical characteristics (channel size, water chemistry, altitude and geology) as possible. Both river channels were characterised by a riffle – pool morphology (channel width 2.9–6.5 m). Catchment geology was dominated by Jurassic mudstones and sandstones (British Geological Survey, 2008) and study sites were located in arable farmland. Close to the catchment outlets, mean daily flows were 0.18 $m^3 s^{-1}$ and 0.52 $m^3 s^{-1}$ for the River Gwash and Chater respectively (record 1978–2015; NRFA, 2017). Subsurface bed material (based on four pooled individual McNeil samples from two riffles per site, average sample weight 20.01 kg [McNeil and Ahnell, 1964]) indicated similar grain size distributions (GSD) between sites; with both being naturally characterised by a moderate fine sediment content (mass <2 mm; Gwash 20% and Chater 28.8%). Hydrological data from local gauging stations indicated that the study coincided with periods of stable flow punctuated by increased river stage associated with summer rainfall events (Fig. 1).

2.2. Colonisation columns

Macroinvertebrate colonisation columns were installed at the two sample sites. These comprised PVC cylinders (diameter 65 mm, height 200 mm) perforated with twelve horizontal holes (diameter 6 mm) to permit horizontal and vertical exchange of water and the free movement of macroinvertebrates and fine sediment (Fraser et al., 1996; Pacioglu et al., 2012; Descloux et al., 2013; Mathers and Wood, 2016). All columns were filled with a pre-washed gravel framework collected from each of the respective sample sites (truncated at 8 mm). This substrate was enclosed in a net bag (7 mm aperture) within each column. Columns were assigned to one of two treatments; a) clean substrates which were free from fines upon installation or; b) heavily sedimented substrates comprising gravel and 250 g of fine sand (63–2000 μm). Preliminary tests indicated that this volume of sand filled 100% of interstitial volume. For the sedimented columns, a circular disk (64 mm diameter) was attached to the mesh bag to effectively seal the base of the column and reduce the loss of fine sediment vertically into the riverbed.

Columns were inserted into the river bed by placing the PVC cylinders onto a steel pipe (35 mm diameter) that was driven into the river bed sediments until a sufficient depth was obtained to insert it flush with the substrate surface (200 mm). The surrounding stream bed remained unchanged and consisted of non-uniform cobbles and gravel. Columns were left in-situ for the entire sampling campaign, but every 14 days the gravel netting bag was removed and replaced without disturbing the surrounding gravel framework. At the end of

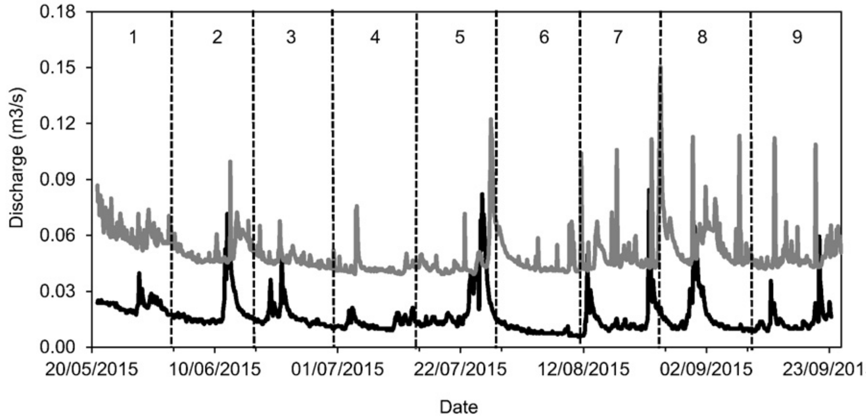

Fig. 1. River discharge (hourly average $m^3 s^{-1}$) for the River Gwash (black) and River Chater (grey) Rutland, UK during the sampling campaign. Dashed lines indicate the two week sampling periods (21st June–24th September 2015).

each 14-day sampling period, the net bag (containing the substrate and macroinvertebrates) was carefully removed, placed in a sample bag and preserved in 10% formaldehyde for subsequent processing in the laboratory. Empty columns were then replaced immediately with the corresponding gravel bag treatment (clean or sedimented).

Colonisation columns were installed every 14 days between 21st May and 24th September 2015 providing a 126 day record (9 sample sets). A time period of 14 days was adopted because preliminary tests indicated that this represented sufficient time to allow for colonisation by macroinvertebrates whilst minimising the amount of fine sediment lost during occasional high flows (See Supplementary Material and Fig. S1). At each riffle site (three on the Gwash and two on the Chater; one until the fourth sampling set), four columns of each type (clean or sedimented) were installed providing a total of 20 replicates (16 initially for three sample sets) for each 14-day sampling period. In total 162 clean and 163 sedimented substrate samples were examined (6 clean and 5 sedimented samples were lost or not retrieved during the field campaign). Two additional sampling timeframes (ca one month: 28 days and ca two months: 56 days) were examined to capture potential temporal variability in environmental conditions (i.e. rising or falling discharge or suspended sediment concentrations) and to confirm the most appropriate time-frame to consider in the main study (See Supplementary Material - Fig. S1).

2.3. Laboratory procedures and statistical analysis

Within the laboratory, the contents of the column bags were passed through a sieve nest (4 and 2 mm sieves) to remove larger gravel clasts. The remaining material was passed through a 250 μm sieve and processed for invertebrates. All macroinvertebrates were identified to the lowest taxonomic level possible usually species or genus with the exception of *Oligiochetea* (order), Diptera families (including *Ephydridae*, *Ptychopteridae*, *Chironomidae*, *Psychodidae*, *Simuliidae*, *Ceraptogonidae* and *Stratiomyidae*), *Sphaeriidae* and *Zonitidae* (family) and *Ostracoda*, *Hydracarina* and *Collembola* which were recorded as such.

Compositional differences in communities between the two sediment treatments were examined via non-metric multidimensional scaling (NMDS) using Bray-Curtis similarity coefficients for the entire data set and for each individual sampling period. This approach enabled an examination of the consistency in the community effects or if they varied over time as a function of environmental conditions (i.e. discharge over the 14-day period). A One way ANOSIM (Analysis of Similarities) was used to examine differences in the communities amongst sediment treatments for the overall data set and for each individual sample set (1–9) using a random Monte Carlo permutations test (999 permutations). Both p and R ANOSIM values were examined, with R values >0.75 indicating strong separation amongst groups, $R = 0.75–0.25$ indicating separate groups with overlapping values and $R < 0.25$ as barely distinguishable groups (Clarke and Gorley, 2006). Taxa contributing to the divergence of communities were identified through the application of the similarity percentage (SIMPER). The top six taxa identified as driving dissimilarity between clean and sedimented communities were selected for further detailed analysis of their sensitivity to fine sediment.

The functional composition of macroinvertebrate communities was determined through the assignment of fauna into 6 categories which were comprised of 44 biological traits from the Tachet et al. (2010) database (Table 1). Categories represent grouping features including 'maximum body size' and 'functional feeding group', whilst traits signify modalities residing within these such as 'shredder' or 'filter-feeder'. Traits were assigned based on a fuzzy-coding approach with scores ranging from zero (indicating no affinity) to three or five (the strongest affinity based on available literature; Chevene et al., 1994). Affinity scores were subsequently rescaled as proportions for each category ($sum = 1$) for each taxon. *Chironomidae* and all taxa recorded at a coarser resolution than family-level were excluded due to the large species diversity within the groups. To produce a trait abundance matrix, taxon-trait categories were multiplied by $\ln(x + 1)$ transformed abundances and were rescaled to sum to one for each trait and each river reach (Larsen and Ormerod, 2010b; Descloux et al., 2014; White et al., 2017). Functional compositional differences for each sampling set were visualised via NMDS plots. All ordination analyses were performed in PRIMER Version 7.0.11 (PRIMER-E Ltd., Plymouth, UK).

The macroinvertebrate communities of the two study streams represent distinct community structures as a function of signal crayfish (*Pacifastacus lencisuclus*) invasion within the River Gwash in 1996 (global ANOSIM $p < 0.001$; Mathers et al., 2016). Following invasion, signal crayfish typically have significant, long-term and persistent effects on macroinvertebrate communities (McCarthy et al., 2006; Twardochleb et al., 2013). As a result, preliminary analyses were conducted on the individual rivers to determine whether the gross effects of sediment loading were comparable for the communities. This analysis took the form of temporal group centroid (clean and sedimented) NMDS plots using Bray-Curtis similarity coefficients. These results indicated that the temporal trajectory of community change and sedimentation effects were comparable for both community composition and biological traits. Taxonomic plots determined a significant sediment treatment effect for both rivers (ANOSIM Gwash $p = 0.035$; Chater $p = 0.012$; Fig. S2) whilst biological traits indicated no divergence in trait composition (ANOSIM Gwash $p = 0.143$; Chater $p = 0.252$). Consequently, as both river communities reacted in a similar manner to sediment loading, the final analyses outlined above were conducted on the combined datasets.

Table 1
Macroinvertebrate functional traits examined within this study (adapted from Tachet et al., 2010).

Category	Trait
Maximal potential size	<0.25 cm >0.25–0.5 cm >0.5–1 cm >1–2 cm >2–4 cm >4–8 cm >8 cm
Reproduction	Ovoviviparity Isolated, free eggs Isolated eggs, cemented Clutches, cemented eggs Clutches, free Clutches, in vegetation Asexual
Respiration	Gill Plastron Spiracle Hydrostatic vesicle Tegument
Locomotion	Flier Surface swimmer Full water swimmer Crawler Burrower Interstitial Temporarily attached Permanently attached
Feeding group	Absorber Deposit feeder Shredder Scraper Filter-feeder Piercer Predator Parasite
Substrate preference	Coarse substrates Gravel Sand Silt Macrophytes Microphytes Twigs/roots Organic detritus Mud

Community abundance, taxa richness and richness of *Ephemeroptera*, *Plecoptera* and *Trichoptera* (EPT) taxa were derived from the raw data. Abundances of taxa characterised as sensitive to sediment according to sensitivity weights provided in the Empirically-weighted Proportion of Sediment-sensitive Invertebrates index (E-PSI; Turley et al., 2016) were also calculated for each sample. To examine statistical differences associated with sediment treatment for individual taxon abundances (as previously selected from the global SIMPER), generalised linear mixed effects models were employed (GLMMs). Models were fitted using the '*lme4*' package in R version 3.2.2 using the '*glmer*' function (R development Core Team, 2015). To examine differences associated with the volume of fine sediment, sediment treatment was specified as a fixed factor and riffle was nested within site as a random factor (based on columns at individual riffles and sites being less independent of each other). Models were fitted using a Poisson error distribution and log link structure. Linear mixed models were fitted to the functional traits and community metrics using the '*nlme*' package and '*lme*' function. The same model structure (outlined above) was employed and the models were fitted using the restricted maximum likelihood (REML) estimation function. A Bonferroni correction was applied to all models to account for the large number of models constructed.

3. Results

3.1. Community composition associated with sediment treatment

63 taxa were recorded in the clean sediment treatment (mean 6.79 taxa per sample, range 2–13) and 58 taxa in the sedimented treatment (mean 6.94 taxa per sample, range 1–16). A total of 9656 individuals were recorded in the clean sediment samples (mean 59.98 individuals per sample, range 14–136) and 8078 in the sedimented samples (mean 49.86 individuals per sample, range 9–168). Communities in the clean sediments were dominated by *Gammarus pulex* (67.68% of total abundance), *Chironomidae* (9.67%) and *Potamopyrgus antipodarum* (6.73%). The most abundant taxa within the sedimented substrates were *G. pulex* (53.50%), *Chironomidae* (12.17%) and *Oligochaeta* (10.84%). A total of 11 taxa were unique to clean sediments (3 *Gastropoda*, 2 *Trichoptera*, 2 *Diptera*, 1 *Ephemeroptera*, 1 *Hirudinea*, 1 *Coleoptera* and 1 *Ostrocoda*) and 2 to the sedimented substrates (1 *Tricladida*, 1 *Trichoptera*) although these occurred at low abundances (constituting 29 and 2 individuals respectively).

Non-metric Multi-Dimensional Scaling (NMDS) ordination diagrams indicated distinct clusters of macroinvertebrate communities associated with the sediment treatment on seven out of the nine sampling occasions (Fig. 2). The degree of separation between the groups varied over time with highly significant divergence in sets 2, 4, 5 and 7 (ANOSIM $p < 0.005$; Fig. 2b, d, e & g), and moderate separation in set 1 (ANOSIM $p = 0.041$; Fig. 2a), whilst a number of sets were less significantly dispersed; sets 6 and 8 (ANOSIM $p < 0.05$; Fig. 2f & h; Table 2). Two 14-day periods, sets 3 and 9 (Fig. 2c & i), demonstrated no significant differences in the macroinvertebrate communities inhabiting the two substrate types. The global dataset indicated some divergence of communities when all timeframes were considered ($p = 0.001$; ANOSIM) although analysis of the R value ($R = 0.083$), indicated that the groups were barely distinguishable from each other (Fig. 2j). This low degree of separation reflects the varying stability of these patterns between the individual sample sets. The top six taxa driving dissimilarity were *Oligochaeta* (5.75% dissimilarity), *Chironomidae* (5.42%), *P. antipodarum* (5.12%), *G. pulex* (4.49%), *Dicranota* sp. (3.10%) and *Habrophlebia fusca* (2.70%).

3.2. Biological traits associated with sediment treatment

NMDS ordination analysis indicated no clear and consistent differentiation between sediment treatments over time when trait community composition was examined (Fig. 3). Trait based community composition demonstrated some degree of separation in five out of the nine sampling sets (i.e. sample sets 1-4; Fig. 3a, b, c & d), but this was not consistent or clear for all sample sets (i.e. sample sets 5 & sets 7-9; Fig. 3e, g, h & i; Table 2). The global dataset indicated little divergence of communities when all timeframes were considered ($p = 0.001$; ANOSIM) with analysis of the R value ($R = 0.056$) indicating that the groups were barely distinguishable from each other (Fig. 3j). When individual traits were considered, eight trait modalities varied significantly as a function of sediment treatment. The trait profile of locomotion was the most significant with individuals characterised as being full water swimmers ($t_{1,\ 320} = -4.53$, $p < 0.001$; LME), crawlers ($t_{1,\ 320} = -3.224$, $p = 0.001$) or interstitial dwellers ($t_{1,\ 320} = -4.93$, $p \leq 0.001$) demonstrating significant reductions for the sedimented treatment. Species demonstrating ovoviviparity ($t_{1,\ 320} = -4.51$, $p \leq 0.001$), respiring via plastron ($t_{1,\ 320} = -4.90$, $p \leq 0.001$) or spiracles ($t_{1,\ 320} = -3.12$, $p \leq 0.001$) and/or demonstrating shredder affinities ($t_{1,\ 320} = -3.43$, $p \leq 0.001$) all demonstrated a reduction within sedimented substrates. Maximum potential size of individuals also varied between treatments with a decline in larger taxon

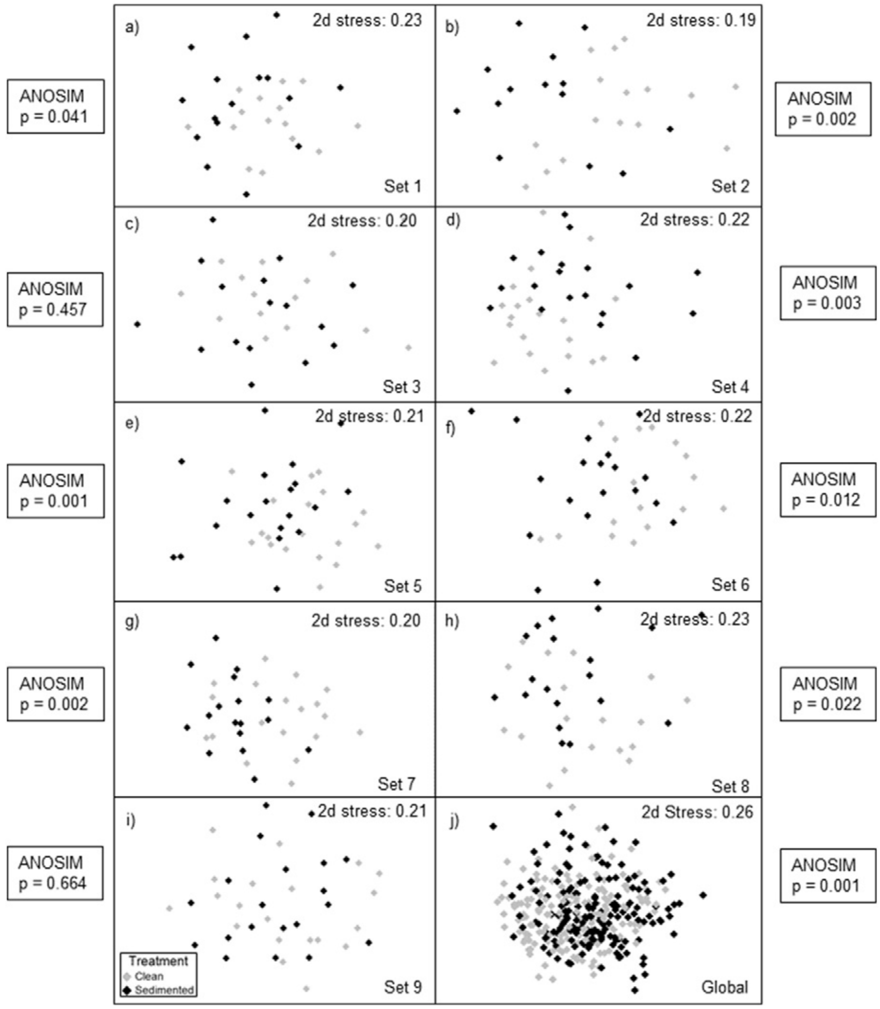

Fig. 2. Non-metric multidimensional scaling (NMDS) of macroinvertebrate community data from the River Gwash and River Chater by sediment treatment using the Bray-Curtis similarities coefficients for cylinder sets 1–9 (panes a–i) and global dataset (pane j). Grey rhombus = clean substrates and black rhombus = sedimented substrates.

characterised with a body size of 1–2 cm within the sedimented columns ($t_{1,\,320} = -3.59$, $p \leq 0.001$).

3.3. Community metrics and individual taxon abundances associated with sediment treatment

Community abundance, taxa richness and EPT richness did not vary by sediment treatment (LME $p > 0.05$). Sediment sensitive taxa (as defined under the E-PSI metric) were recorded in significantly greater abundances in the clean sediments ($t_{10,\,310} = -2.94$, $p < 0.001$). The divergence of clean and sedimented substrates was not apparent during Set 1, 3 and 9 with similar abundances of sensitive taxa in both treatments whilst the greatest distinction between sediment treatments was during sets 4–8 (Fig. 4). When individual taxon abundances were considered, *Dicranota* sp. and Oligochaeta were found in significantly greater abundances in sedimented columns ($Z_{1,\,320} = 8.76$, $p < 0.001$ and $Z_{1,\,320} = 15.84$, $p < 0.001$; GLMM). Clean sediment treatments were found to support greater abundances of the ephemeropteran *H. fusca* ($_{Z1,\,320} = -6.76$, $p < 0.001$) and the amphipod *G. pulex* ($Z_{1,\,320} = -20.03$, $p < 0.001$). No significant sediment treatment differences were determined for any other taxa ($p > 0.05$) although EPT richness demonstrated significant variability over time within this study ($t_{10,\,320} = -3.45$, $p < 0.001$; LME; Fig. 5).

Table 2
Summary of ANOSIM values over time by sediment treatment for taxonomic and functional trait community compositions.

	Taxonomic		Functional trait	
Set	r value	p value	r value	p value
1	0.078	0.041	0.040	0.153
2	0.231	0.002	0.035	0.201
3	−0.003	0.457	0.016	0.297
4	0.107	0.003	0.060	0.069
5	0.127	0.001	0.030	0.158
6	0.096	0.012	0.065	0.037
7	0.166	0.002	0.041	0.121
8	0.082	0.022	−0.006	0.455
9	0.018	0.664	0.047	0.991

4. Discussion

4.1. Macroinvertebrate community composition

This study sought to examine the temporal variability of experimentally enhanced fine sediment loading on macroinvertebrates communities. The results indicate colonisation by macroinvertebrates may be impeded as a result of enhanced fine sediment loading but that the effects vary temporally. Analysis demonstrated a significant difference in macroinvertebrate community composition associated with sediment treatment during seven of the nine 14-day sampling periods. However, the effects of sedimentation were not temporally consistent

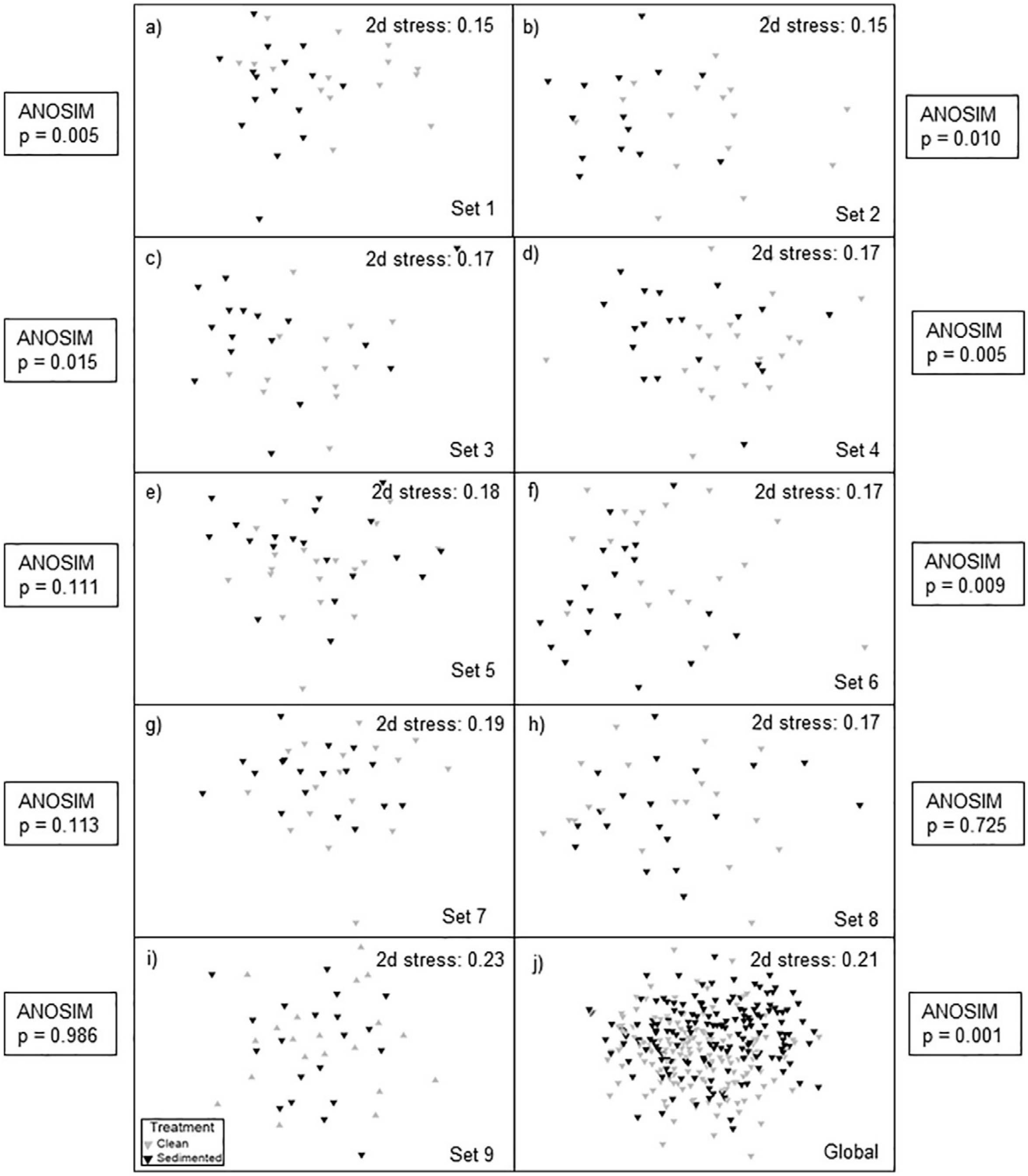

Fig. 3. Non-metric multidimensional scaling (NMDS) of macroinvertebrate community functional traits from the River Gwash and River Chater by sediment treatment using the Bray-Curtis similarities coefficients for cylinder sets 1–9 (panes a–i) and global dataset (pane j). Grey rhombus = clean substrates and black rhombus = clogged substrates.

with differences between community composition being stronger in some periods and breaking down completely in others.

No evidence was found to suggest that spate periods affected the degree of separation between communities within sedimented and clean substrates. A number of sample sets experienced periods with high flows (e.g. sets 6 and 8) but this did not appear to have any effect on the colonisation of the sediments. Similarly, sample sets which demonstrated little separation did not correspond with periods of high flow (i.e. sample set 3). It is likely that the variable responses to sedimentation reflects the different life cycle characteristics and stages present in the river during the study and therefore reflects natural temporal variability in the macroinvertebrate community structure. The abundance of sediment sensitive taxa demonstrated a similar pattern to that recorded for the taxonomic NMDS plots, with no differences in abundances recorded for sets 1, 3 and 9. These changes in sediment sensitive taxa may be driven by the life cycle of EPT taxa, which are particularly sensitive to fine sediment within the substrate (Conroy et al., 2016) and which were temporally variable in richness during this study. EPT richness below a threshold of 2 taxa in this study coincided with clear differences in community structure associated with the sediment treatment.

Given the study took place during summer; discharges were naturally low and favoured the deposition of fine sediments (Wood and Armitage, 1999). Consequently, the dominant taxa recorded during this period are more likely to display affinities to fine sediment such as the families of *Caenidae* and *Chironomidae* (Jowett, 1997; Dewson et al., 2007). The presence of later instars of EPT taxa during the summer months may be limited due to emergence patterns, but the majority

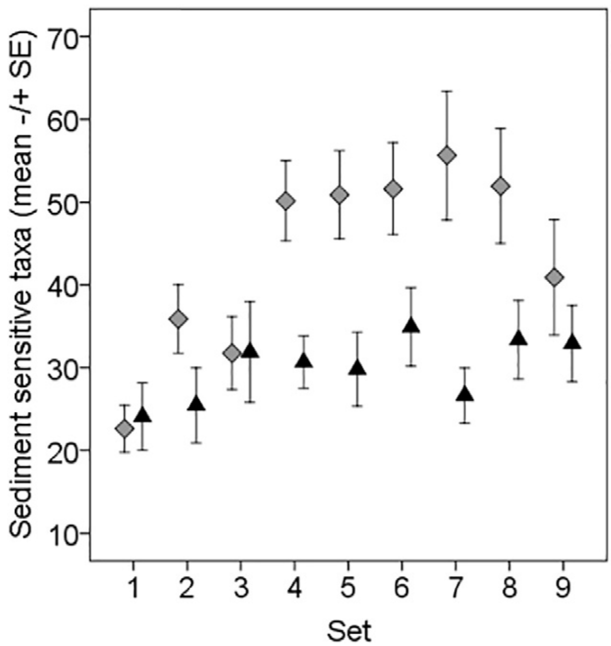

Fig. 4. Mean abundances (+/− 1 SE) of sediment sensitive taxa (as defined under the E-PSI index) over the nine sampling sets. Grey rhombus = clean substrates and; black triangle = sedimented substrates.

(excluding *Caenidae*) probably display a greater affinity for clean substrates (Sutherland et al., 2012) and may account for the community patterns recorded in this study. As such, the implications of fine sediment deposition will be most pronounced during summer months. It is therefore vital to consider within-year temporal variation and taxon life stages when assessing the implications of fine sediment deposition on aquatic communities (Johnson et al., 2012).

Overall significant differences were recorded for the abundances of taxa classified as sensitive to fine sediment (Turley et al., 2016). These results indicate that at the patch scale, removal of fine sediments may enhance habitat complexity and thereby increase the heterogeneity of instream communities. Micro-scale habitat characteristics are critical in the regulation of macroinvertebrate diversity (Pardo and Armitage, 1997; Lamouroux et al., 2004; Laini et al., 2014). Despite this, the majority of studies conducted on fine sedimentation impacts often take a reach-scale approach (e.g. Downes et al., 2006; Burdon et al., 2013) and therefore understanding the importance of variable micro-scale

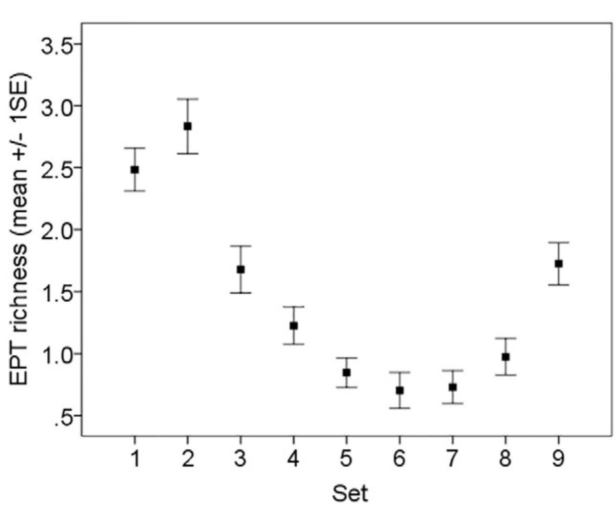

Fig. 5. Mean abundances (+/− 1 SE) of EPT taxa over the nine sampling sets.

habitat dynamics is limited. Within this study clean substrates supported a greater number of unique taxa (11) compared to sedimented substrates (2), highlighting the importance of micro-scale habitat differences for biodiversity.

Taxa richness, community abundance and EPT richness did not demonstrate any significant differences between sediment treatments. The documented effects of fine sediment on taxa richness and community abundance are not consistent in the literature with some studies documenting a reduction in taxa richness (Cline et al., 1982; Rabeni et al., 2005) or community abundance (Armstrong et al., 2005; Larsen et al., 2011) whilst others recorded no modification (Lenat et al., 1981; Kaller and Hartman, 2004; Downes et al., 2006); and in some instances abundances have been reported to increase (Matthaei et al., 2006). Streams that are characterised by low fine sediment content and support a greater proportion of fine sediment sensitive taxa, are likely to be more heavily affected. In contrast, rivers that are species poor may not display a marked response to an increase in fine sediment.

4.2. Taxon specific responses to fine sedimentation

A small number of associations were observed between individual taxa and fine sediment treatments. Sedimented substrates were characterised by significantly greater abundances of two taxa that typically burrow into fine substrates; *Dicranota* sp. and Oligochaeta (Lenat et al., 1979; Fitter and Manuel, 1986). Even at the order level, *Oligochaeta* are widely documented to be positively correlated with fine sediment (Richards et al., 1993; Waters, 1995; Angradi, 1999; Descloux et al., 2013); however, the experimental effects of fine sediment for *Dicranota* sp. have not been widely documented. The reduction of pore space in heavily sedimented and clogged substrates potentially favours taxa with small body sizes (Gayraud and Philippe, 2001; Duan et al., 2009; Xu et al., 2012). In marked contrast, two species demonstrated strong affinities for clean substrates; the Ephemeropteran species, *Habrophlebia fusca* which may be vulnerable to gill damage within fine bed material (Jones et al., 2012) and *Gammarus pulex*, which although common in rivers with fine sediment patches is a highly mobile taxon and may have actively sought clean sediments (Wood et al., 2010; Mathers and Wood, 2016).

4.3. Biological traits

Several previous studies have suggested that macroinvertebrate community trait profiles may alter as a function of habitat modifications; reflecting a filtering effect of taxa with traits sensitive to fine sediment deposition (Usseglio-Polatera et al., 2000; Larsen et al., 2011; Bona et al., 2015; Doretto et al., 2017). However, when the functional composition of macroinvertebrate communities was examined in this study, the effects of fine sediment were not as marked as those obtained using taxonomic community composition data. Differences between functional trait composition associated with sediment treatment were only observed on a very limited number of sampling occasions with trait profiles breaking down completely towards the latter end of the sampling period (encompassing the latter half of August and September), most likely associated with taxon lifecycles. Despite the absence of a clear community effect, a number of individual traits showed a significant response to fine sediment content.

Locomotion modalities were the most responsive to increased fine sediment loading with crawlers, swimmers and interstitial dwellers all demonstrating a reduction in occurrence within sedimented substrates. Habitat trait groups have been reported to display significant responses to sedimentation, with fine sediment having the potential to limit access to preferred habitats (Gayraud and Philippe, 2001; Rabeni et al., 2005). Interstitial pore space is an important determinant in macroinvertebrate colonisation and diversity, with fine sediment clogging limiting the ability of many taxa to access subsurface habitats, in particular larger organisms that require larger interstitial space (Larsen and

Ormerod, 2010b; Mathers et al., 2014). It is therefore not surprising that the number of interstitial dwellers in combination with the maximal size of organisms reduced within the sedimented columns (Buendia et al., 2013; Descloux et al., 2014; Milesi et al., 2016). Similarly, crawlers have been widely documented to be affected by increasing fine sediment content with some studies citing their reduced locomotion as a factor in their reduced abundance (Bo et al., 2007; Buendia et al., 2013) whilst others link their decline to negative effects on respiration modalities (Rabeni et al., 2005). In contrast, the habitat group of swimmers demonstrated variable responses to enhanced sediment loading, with some studies documenting a decrease in richness but no effect on density (Rabeni et al., 2005), whilst others saw a reduction in abundance (Larsen et al., 2011) or even a positive correlation (Buendia et al., 2013). Habitat complexity prior to sedimentation probably influences the magnitude of the effects recorded on the invertebrate assemblage. Rivers which are naturally more heterogeneous are likely to display greater effects in response to instream stressors such as fine sediment deposition.

Feeding modalities are often associated with fine sediment content, with increasing fine sediment loads affecting the quality of trophic resources and thereby affecting feeding activities (Jones et al., 2012). In contrast to the expectations of the wider literature, the only taxa that demonstrated a reduction in abundance to increased fine sediment content were those that displayed shredder feeding characteristics (Descloux et al., 2014; Doretto et al., 2016). Similarly, respiration modalities are often particularly sensitive to fine sediment with some respiratory structures being significantly impaired or damaged by fine particles (Lemley, 1982; Townsend et al., 2008). This study documented no significant associations with fine sediment content and respiratory structures which were supported by the wider sedimentation literature. Taxa which respire via plastron and spiracles demonstrated a reduction in abundance in marked contrast to results reported by Logan (2007) and Archaimbault et al. (2005). This biological response is primarily a function of increasing numbers of the *Diptera* within the genus *Dicranota* sp. and may highlight a limitation of biological trait analyses that only consider individual traits.

The application of biological traits in evaluating the effect of stressors has seen increasing recognition, with many studies proposing that the application of trait compositions may provide a better or comparable indicator for different types and combinations of instream stressors than traditional taxonomic based metrics (Menezes et al., 2010; Göthe et al., 2016). However, from the results reported here and in a number of other studies, it is clear that further research is required around the assignment of biological traits and caution should therefore be applied when undertaking such analyses (Buendia et al., 2013; Descloux et al., 2014). Further research is required to develop trait databases that have greater applicability to the ecosystems being assessed. Currently the only database available to European researchers is that by Tachet et al. (2010) developed in French streams. Although applicable to other European streams, the low taxonomic resolution of the database (family/genus) raises some questions regarding the wider application of such an approach without some local modifications as many families with multiple genus (e.g. *Baetidae* and *Chironomidae*) support highly variable taxonomic responses (Monk et al., 2012). Traits are also unlikely to act in isolation but rather a combination of traits will determine the response of an individual species to a stressor (Pilière et al., 2016). Consequently, in future research, traits should be assessed as interacting factors within a more fully developed mechanistic understanding of the observed effects of fine sediment for macroinvertebrates.

5. Conclusion

Understanding the mechanistic implications of fine sediment upon macroinvertebrate communities still remains a significant challenge. This study indicates that the effect of increased fine sediment loading upon macroinvertebrate assemblages is not temporally consistent with a number of sampling periods displaying no discernible effects of fine sediment loading. The implications of increased fine sediment loading are likely to be heavily dependent on the timing of sedimentation events relative to taxon life cycles. Future studies concerned with investigating the effects of fine sediment should do so with a greater awareness of the temporal dynamics of the communities they are studying. Despite the increasing application of biological trait composition within biomonitoring efforts, community trait profiles did not perform as consistently or strongly towards the effect of enhanced fine sediment loading as taxonomic approaches. Patch scale responses to fine sediment were however evident, with the two substrate treatments supporting distinct communities when taxonomic composition and individual trait modalities were considered. The results from this study indicate the importance of recognising micro-scale habitats within the context of maximising aquatic biodiversity. Further research is required to fully understand the seasonal effects of fine sediment deposition and dynamics on aquatic macroinvertebrate assemblage structure and function.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2017.04.096.

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Understanding the controls on deposited fine sediment in the streams of agricultural catchments

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HIGHLIGHTS

- Field data for deposited fine sediment in agricultural streams are presented.
- Stream power was found to be the most effective explanatory variable.
- The majority of stream beds were saturated with fine sediment.
- Below saturation, deposited fine sediment is related to sediment pressure.
- Target sediment loads need to include the ability of streams to transport sediment.

GRAPHICAL ABSTRACT

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ABSTRACT

Excessive sediment pressure on aquatic habitats is of global concern. A unique dataset, comprising instantaneous measurements of deposited fine sediment in 230 agricultural streams across England and Wales, was analysed in relation to 20 potential explanatory catchment and channel variables. The most effective explanatory variable for the amount of deposited sediment was found to be stream power, calculated for bankfull flow and used to index the capacity of the stream to transport sediment. Both stream power and velocity category were highly significant ($p \ll 0.001$), explaining some 57% variation in total fine sediment mass. Modelled sediment pressure, predominantly from agriculture, was marginally significant ($p < 0.05$) and explained a further 1% variation. The relationship was slightly stronger for erosional zones, providing 62% explanation overall. In the case of the deposited surface drape, stream power was again found to be the most effective explanatory variable ($p < 0.001$) but velocity category, baseflow index and modelled sediment pressure were all significant ($p < 0.01$); each provided an additional 2% explanation to an overall 50%. It is suggested that, in general, the study sites were transport-limited and the majority of stream beds were saturated by fine sediment. For sites below saturation, the upper envelope of measured fine sediment mass increased with modelled sediment pressure. The practical implications of these findings are that (i) targets for fine sediment loads need to take into account the ability of streams to transport/retain fine sediment, and (ii) where agricultural mitigation measures are implemented to reduce delivery of sediment, river management to mobilise/remove fines may also be needed in order to effect an improvement in ecological status in cases where streams are already saturated with fines and unlikely to self-cleanse.

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1. Introduction

Excessive sediment pressure on aquatic habitats has become of increasing concern for river systems around the world (Relyea et al., 2012). In particular, intensification of agriculture has increased fine sediment loading to rivers (Wilcock, 1986; Dearing et al., 1987; Owens and Walling, 2002; Walling et al., 2003a; Foster et al., 2011; Jones and Schilling, 2011), leading to high concentrations of suspended solids and, potentially, deposition of fine sediment. Evidence has also been accumulating, from both field survey and experiments, on the deleterious effects of excessive fine sediment on biota (Waters, 1995; Wood and Armitage, 1997; Matthei et al., 2006; Bilotta and Brazier, 2008; Larsen et al., 2011; Sutherland et al., 2012; Wagenhoff et al., 2012, 2013; Chapman et al., 2014). It is clear from this evidence that the impact of excessive fine sediment on biota is more often related to deposited rather than suspended material (Kemp et al., 2011; Jones et al., 2012a, 2012b, 2014). In the light of this, attempts have been made to identify target values for both deposited fine sediment and sediment loading (Cooper et al., 2008; Collins and Anthony, 2008; Bryce et al., 2010; Collins et al., 2011; Benoy et al., 2012). Yet, the relationship between deposited fine sediment and agricultural sediment pressure is still poorly understood.

Sediment pressure has been variously quantified by catchment or local/network riparian land use (Sutherland et al., 2010), runoff-weighted percentage land use (Wagenhoff et al., 2011) and modelled sediment load apportionment (Collins and Anthony, 2008). Catchment land use has been shown to be related to deposited fine sediment in specific cases of intensification of agriculture (e.g. Niyogi et al., 2007; Sutherland et al., 2010; Wagenhoff et al., 2011). However, at a strategic level, only the approach based on modelled sediment load has potential to link fine sediment deposition with current or future projected land management and, thus, provide information on the likely effectiveness of mitigation measures for fine sediment delivery to rivers in terms of sediment deposition and its biotic impact. The ability to make this link is fundamental to supporting national policies regarding the protection of water resources and ecological status.

Representative field sampling of deposited fine sediment in agricultural streams across England and Wales, carried out as part of a wider national scientific policy support project, provided a unique opportunity to explore the relationship between an instantaneous measurement of deposited fine sediment and sediment pressure. Sampling was specifically designed to cover both the range of agricultural sediment pressure and different biological river types across England and Wales (following Davy-Bowker et al., 2008). The impact on biota is covered elsewhere (Murphy et al., 2015). The aim of this paper is to analyse the sediment data in conjunction with a range of catchment and channel descriptors in order to investigate potential linkages between agricultural sediment pressure and deposited fine sediment in streams. In particular, it is hypothesized that the mass of deposited fine sediment is directly related to the amount of sediment delivered to the channel and inversely related to the capacity of the stream to transport fine sediment.

2. Approach and methods

The approach taken was a synoptic survey of streams in agricultural catchments across England and Wales. Sampling sites were selected from the 12,447 stream sites within the Environment Agency River Habitat Survey (RHS) database. Biological river types were based on the physical attributes of catchment geology, distance from source, altitude and slope; with boundary values loosely based on those associated with RIVPACS IV super end groups (Davy-Bowker et al., 2008). Screening was undertaken to eliminate any sites with a substantial influence from urban areas or sewage effluent (see below). All sites were upstream of any lakes and reservoirs and on independent watercourses; in cases with more than one candidate site per watercourse, the most downstream site meeting the screening requirements was selected. Full details regarding the site selection process are given in Murphy et al. (2015). Some 230 sites were sampled once in either spring or autumn between May 2010 and November 2011. Most samples were collected during low to medium flows as necessitated by the technique and no samples were collected during or immediately after peak flow events. From data on water width, depth and velocity category at the time of sampling, approximately 90% samples were collected when the flow was less than 10% of the estimated median annual flood, or approximately bankfull flow. An independent dataset (Anthony et al., 2012) of 55 similar sites, sampled in both autumn and spring by the same field team and in exactly the same manner between October 2009 and May 2011, was also available for model testing and to assess temporal variability.

2.1. Deposited fine sediment

Fine sediment deposited on, or in, the river substrate to a depth of about 10 cm was collected using the disturbance technique (Duerdoth et al., 2015 adapted from Collins and Walling, 2007a, 2007b). An open-ended, stainless steel cylinder (height 75 cm; diameter 48.5 cm) was carefully inserted into an undisturbed patch of stream bed to a depth of at least 10 cm, until an adequate seal with the substrate was achieved, and the depth of water within the cylinder was measured. To provide an instantaneous measure of the deposited surface drape, the water column was agitated vigorously for one minute using a metal pole, without touching the stream bed. This established a vortex that brought any fine sediment into suspension. This was then immediately sampled, while the water was still in vigorous motion, by plunging two inverted 50 ml tubes to the bottom of the cylinder which then filled as they were turned upright and brought to the surface. To sample the total (i.e. combined surface and sub-surface) deposited fine sediment, the stream bed was then disturbed to a depth of about 10 cm, vigorously agitated for one minute to suspend any subsurface fines and a second pair of 50 ml samples quickly taken. For each river reach sampled, four sampling locations were identified visually by the workers in the field. In broad terms, patches with a propensity to erode fine sediment (erosional) were defined as those higher velocity areas in or close to the thalweg, whereas patches with a propensity to deposit fine sediment (depositional) were in eddies or areas of lower flow velocity such as pools or backwaters. Two sets of samples were collected from erosional and two from depositional zones of the main channel, in order to characterise the reach-scale average (derived from all 4 samples) and provide an indication of within-reach variability.

The samples were refrigerated and kept in the dark until analysed. Deposited fine sediment was characterised in terms of mass, volatile solids (i.e. organic matter derived from loss on ignition) and particle size. Fine sediment mass and volatile solids were measured within one week of return to the laboratory using one of each pair of 50 ml tubes. The samples were passed through a 2 mm sieve, to remove leaves and twigs, prior to filtration using pre-ashed, washed and dried 90 mm Whatman Glass Microfibre GF/C filters (pore size 1.2 μm). The filtered samples were dried in a pre-heated oven at 105 °C overnight and ashed in a pre-heated muffle furnace at 500 °C for 30 min. Reach-average values of sediment mass were calculated using geometric means. Averaging the four samples provided an effective measure of deposited fine sediment at the reach scale (cf. Collins and Walling, 2007a, 2007b) which has been shown to be reliable across a wide range of river types (>60% boulders/cobbles to >60% sand and silt) and not affected by operator bias (Duerdoth et al., 2015). Measurement uncertainty, in terms of 95% confidence intervals, was estimated to be ± 0.27 and ± 0.32 logarithmic units (i.e. factors of 1.86 and 2.09) on the average total and surface deposited fine sediment, respectively (Duerdoth et al., 2015).

Absolute particle size (<1 mm) was analysed on the second 50 ml tube of each pair using a Malvern Mastersizer 2000. In most cases, the whole sample was analysed using either a HydroS (with pump/stir speed of 2700 rpm) or HydroG (with pump speed 1600 rpm and stir speed 700 rpm) dispersion unit, dependent on the amount of sediment in the sample. For very large amounts of sediment, samples were centrifuged at 4000 rpm for 15 min, the supernatant carefully decanted and the sediment thoroughly mixed before subsampling. In order to give the absolute particle size distribution of the whole sample, organic material was not removed. To aid disaggregation and dispersion, 5 ml of 5% sodium hexametaphosphate was added to each sample which was then shaken and left for a minimum of 1 h before analysis. The sample was then passed through a 1 mm sieve into the dispersion unit where maximum ultrasound was applied for 3 min and switched off for 1 min prior to measurement.

For each of the sampled sites, land cover, modelled sediment pressure and other catchment and channel descriptors were derived as follows.

2.2. Land cover

Land cover data for 2007 was derived for each of the sites in ARC-GIS Version 9.3.1 using the 25 m raster dataset LCM2007 (Morton et al., 2011) and digital catchment boundaries based on a 50 m digital terrain model (Morris and Flavin, 1990). The LCM2007 dataset was developed from satellite images and digital cartography and gives land cover information based on the UK Biodiversity Action Plan Broad Habitats. It has 23 classes. These were amalgamated into three classes considered to be most relevant to different agricultural use (i.e. arable and horticulture, improved grassland, and unimproved grassland/upland), as described in Table 1. In the case of improved grassland, land cover classes 6 and 7 (neutral and calcareous grassland, respectively) have been included with class 4 (designated improved grassland) as these have similar spectral properties and so may not be distinguishable; in practice, land cover classes 6 and 7 are only minor components, making up less than 4% of the total area in all but three of the selected catchments.

2.3. Sediment pressure

Agricultural sediment delivery to streams was modelled using a national pressure layer generated by a new policy-support framework based on updates and refinements to the process-based Phosphorus and Sediment Yield CHaracterisation In Catchments (PSYCHIC) model (Collins et al., 2007, 2009a, 2009b; Davison et al., 2008; Stromqvist et al., 2008) and the June agricultural census returns for 2010 as model input for crop areas and livestock numbers. This is a generic model based on national datasets relating to climate, soils and farm types which is designed to capture the variation in sediment pressure across England and Wales. The original PSYCHIC framework has been shown to perform satisfactorily at field (Collins et al., 2009a) and national (Collins et al., 2009b) scale. The agricultural sediment pressure modelling framework used in this work has been tested and shown to perform satisfactorily at a range of scales including plot, field, catchment (Collins et al., 2012a) and national (Zhang et al., 2014) scale. The calculation of cross-sector sediment pressures is fully described in Collins et al. (2009a). Sediment pressure from urban sources was calculated on the basis of published data for event mean concentrations following Mitchell et al. (2001) and Mitchell (2005). Inputs from sewage treatment works were based on consented discharges and a correction for the relationship between observed and consented suspended solids concentrations. Sediment pressure from bank erosion was calculated as a function of the duration of excess bank shear stress and channel density, calibrated against the results from sediment fingerprinting studies (Collins and Anthony, 2008; Collins et al., 2009a). The modelled cross-sector data were used to ensure that no site had urban inputs >2 kg/ha/yr or STW inputs >0.5 kg/ha/yr, thereby permitting an assessment of the potential relationship between agricultural fine sediment loss and instantaneous measurements of deposited fine sediment on stream beds.

Table 1
Catchment and channel descriptors.

Descriptor	Source/derivation
Arable (%)	% area in LCM2007 class 3 (arable and horticulture)[a]
Improved grassland (%)	% area in LCM2007 classes 4, 6 and 7 (improved, neutral and calcareous grassland)[a]
Unimproved grass and upland (%)	% area in LCM2007 classes 5, 10, 11, 12 and 13 (rough grassland, heather, heather grassland, bog and montane habitats)[a]
Sediment pressure (T/yr)	Derived from updated PSYCHIC model (see text)
Sediment yield (T/km^2/yr)	Derived from sediment pressure and catchment area
Catchment area (km^2)	Digital terrain model (50 m resolution)
Altitude (m)	RHS database from maps[b]
Distance to source (km)	RHS database from maps[b]
Strahler stream order	RHS database from maps[b]
Channel slope (m/km)	RHS database from maps[b]
MSUB (phi units)	Mean substratum size derived from field measurement at time of sampling using RIVPACS protocol[c]
Bankfull width (m)	RHS database from field survey[b]
Water width (m)	Field measurement at time of sampling (RIVPACS protocol)[c]
Water depth (m)	Field measurement at time of sampling (RIVPACS protocol)[c]
Velocity category	Field measurement at time of sampling (RIVPACS protocol)[c] 1:≤10 cm/s; 2:> 10 to ≤25 cm/s; 3:> 25 to ≤50 cm/s; 4:> 50 to ≤100 cm/s; 5: >100 cm/s
Habitat Modification Score class	RHS database from field survey[b]
Median annual flood (m^3/s)	Flood Estimation Handbook method using digital data (see text)
Stream power (W/m)	Derived from median annual flood and channel slope
Unit stream power (W/m^2)	Derived from stream power and bankfull width
Baseflow index	Estimated from hydrology of soil types[d]

[a] Morton et al. (2011).
[b] Environment Agency (2003).
[c] Murray-Bligh et al. (1997).
[d] Boorman et al. (1995).

2.4. Other catchment and channel descriptors

In addition to the land cover statistics and modelled sediment pressure for each of the sampled sites, a range of catchment and channel descriptors were available from maps or associated databases (Table 1). They included those RIVPACS channel descriptors (substrate size, water width, water depth and velocity category) collected during the field campaigns, thus characterising hydromorphological conditions at the time of sampling, and descriptors from the RHS database.

In addition, stream power has been used to index the capacity of a stream to transport sediment (Bagnold, 1966; Knighton, 1999; Gurnell et al., 2010). It is well-known that most of the annual load of suspended sediment is carried during high flows so stream power was calculated using the median annual flood (similar in return period to bankfull flow) which can be estimated from catchment characteristics. A revised unbiased equation for the median annual flood, based on a study of 602 rural catchments across the UK, is given by Kjeldsen and Jones (2010) as:

$$Q_{MED} = 8.3062\, AREA^{0.8510}\, 0.1536^{(1000/SAAR)}\, FARL^{3.4451}\, 0.0460^{BFIHOST2}$$

where Q_{MED} is median annual flood (m^3/s), $AREA$ is catchment area (km^2), $SAAR$ is standard average annual rainfall 1961–90 (mm), $FARL$ is an index of flood attenuation due to reservoirs and lakes, and $BFIHOST2$ is the square of the baseflow index derived from Hydrology of Soil Types (HOST) data (Boorman et al., 1995).

Stream power and specific, or unit, stream power (Bagnold, 1966) are then given by:

$$\Omega = \rho g\, Q_{MED}\, S$$

$$\omega = \Omega / W_{BF}$$

where Ω is stream power (W/m), ρ is density of water (kg/m^3), g is acceleration due to gravity (m/s^2), Q_{MED} is median annual flood (m^3/s), S is channel slope (m/m), ω is specific or unit stream power (W/m^2), W_{BF} is bankfull width (m). Both channel slope and bankfull width were taken from the RHS database.

Flow regime is also relevant to fine sediment deposition in that it indicates the overall balance between potentially depositing and flushing flows. This may be effectively represented by the baseflow index (BFI) or proportion of the flow which occurs as baseflow. Low values of BFI represent flashy responsive catchments, while high values represent slowly-responding groundwater-fed catchments with a propensity for excessive deposition of fine sediment (Sear et al., 1999). BFI was estimated directly from the proportion of HOST soil types in the catchment. The HOST classification of soils (Boorman et al., 1995) is based on conceptual models of the hydrological processes taking place in the soil and, where appropriate, the underlying geology. These models take into account the physical properties of the soil, permeability of the underlying geology and depth of the water table. BFI coefficients for each of the soil classes were derived from measured BFI for 575 catchments across the UK using bounded multiple regression analysis by Boorman et al. (1995); the overall standard error of the estimate across all soil classes is quoted as 0.09.

2.5. Statistical analysis

Analysis was carried out in the R language. The amount of deposited fine sediment, as well as many of the variables included in the analysis, were log-normally distributed. Consequently, a logarithmic transformation was applied to all continuous variables. This implies that the model developed to explain the deposited fine sediment will be multiplicative in form which seemed appropriate. Categorical variables were treated as factors. The Habitat Modification Score (HMS) class (an indicator of anthropomorphic alteration of the river channel and available from the River Habitat Survey database) was subsequently dropped from the analysis as individual subscores could be interpreted as either enhancing or reducing deposition of fines, sometimes dependent on whether samples were upstream or downstream of a particular feature, leading to inconsistency of impact. Preliminary regression tree analysis suggested that interaction terms were not important.

3. Results

The sampled sites were strongly biased towards the north and west of England and Wales (Fig. 1). This was due to the process of site screening to ensure that the sediment pressure was mostly derived from agriculture as described by the cross-sector model. Missing catchment or channel characteristics meant that 26 sites were dropped from the analysis. Modelled sediment pressure, expressed as sediment yield, ranged from 1.4 to 190 T/km^2/yr., with a median value of 28 T/km^2/yr. The majority of these values were well above empirical target values proposed for the sediment yields of different river types (Cooper et al., 2008) and alternative targets derived from palaeolimnological reconstruction to represent modern background sediment delivery to river channels, prior to post-war agricultural intensification (Foster et al., 2011). Thus, it is highly plausible that most of the sites were heavily impacted by agricultural sediment (cf. Collins et al., 2012b).

3.1. Deposited fine sediment

The reach-averaged instantaneous mass of fine sediment in the surface drape ranged from 6 to 4562 g/m^2 with a median value of 181 g/m^2; the reach-averaged mass of total fine sediment (i.e. surface plus subsurface down to circa 10 cm depth) ranged from 8 to 69,664 g/m^2 with a median value of 906 g/m^2 (Table 2). Volatile solids (i.e. organic fraction determined by loss on ignition) ranged from 2 to 497 g/m^2 in the surface drape and 4 to 3492 g/m^2 in the total. The median percentage volatile solids was 16% for the surface drape and

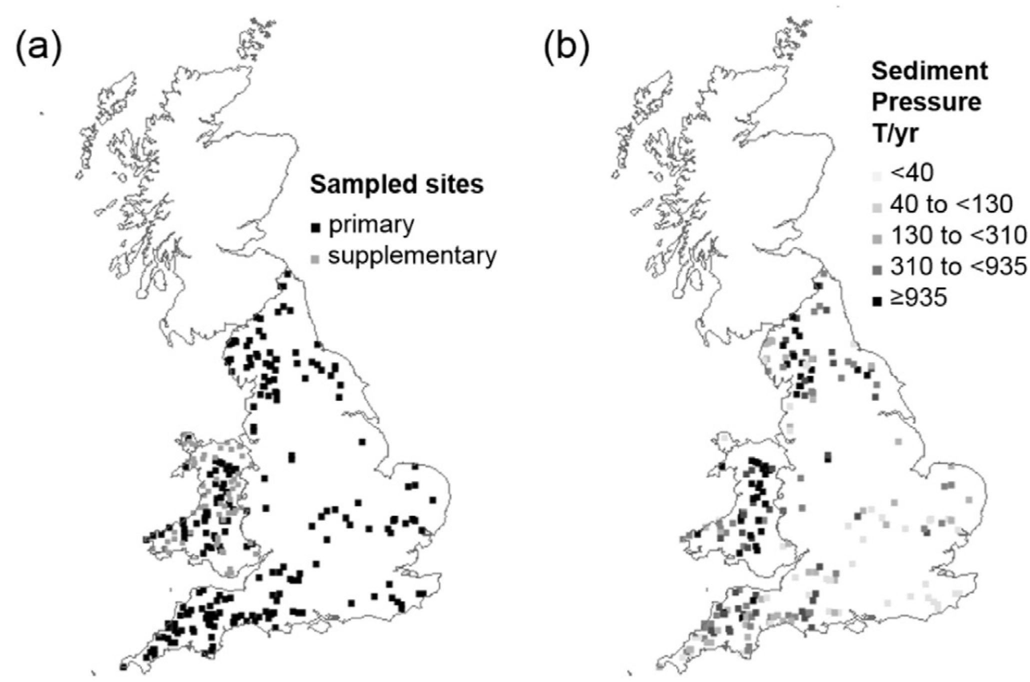

Fig. 1. (a) Location of sampled sites; (b) sediment pressure class based on quintiles from an updated version of the PSYCHIC model using agricultural data for 2010.

Table 2
Selected percentiles for reach-averaged instantaneous measures of sediment mass and particle size for surface drape and total.

%ile	Sediment mass g/m^2	Volatile solids g/m^2	Volatile solids %	Median grain size μm	Span grain size[a]	Sand % by volume	Silt % by volume	Clay % by volume
Surface drape: reach-averaged values primary sites								
10	25.58	6.48	9.54	15.95	3.80	16.04	47.62	6.97
25	58.04	10.99	12.62	19.31	4.28	20.29	55.84	8.95
50	180.86	25.17	16.44	25.44	5.07	26.10	61.90	11.81
75	454.13	60.37	22.17	35.09	6.00	34.70	66.33	13.96
90	988.22	132.52	34.45	45.09	6.97	42.32	69.44	16.27
Surface drape: reach-averaged values supplementary sites								
10	35.91	10.34	9.51	14.64	4.06	14.89	58.07	7.59
25	83.62	15.69	13.15	18.53	4.45	18.06	60.73	9.53
50	196.73	33.97	18.39	23.23	5.15	23.81	64.74	11.44
75	383.31	51.99	24.41	27.19	5.81	27.45	67.76	14.73
90	1074.82	125.80	35.92	33.18	6.64	32.42	70.59	17.20
Total (surface and subsurface to circa 10 cm): reach-averaged values primary sites								
10	107.47	16.04	6.39	16.45	4.11	18.37	42.05	6.22
25	301.89	33.22	8.51	20.43	4.74	23.01	49.64	8.40
50	906.01	82.82	11.12	27.21	5.80	30.36	57.88	11.12
75	2452.09	241.54	14.91	40.13	7.22	39.73	63.54	14.10
90	7720.38	550.33	19.35	64.61	8.51	49.52	67.21	16.66
Total (surface and subsurface to circa 10 cm): reach-averaged values supplementary sites								
10	175.79	20.41	6.83	15.57	4.44	17.11	52.06	7.10
25	397.46	45.84	8.98	19.13	5.06	20.75	58.43	8.62
50	961.42	103.01	12.06	23.70	6.08	24.97	63.17	10.76
75	2187.51	181.45	15.99	33.00	6.98	33.10	65.85	13.85
90	7567.31	573.01	20.50	39.16	8.03	38.70	69.16	16.62

[a] Span of grain size given by $(D_{90} - D_{10}) / D_{50}$ where D_i is the absolute grain size with i% finer by volume.

11% for the total, with the surface drape having a higher percentage content of volatile matter, as might be expected. There was close correlation between surface and total sediment mass (Spearman rank correlation $\rho = 0.92$; $p \ll 0.001$).

The reach-averaged median absolute particle size (Table 2) varied between 10 and 176 μm in the surface drape; 95% sites had a median grain size in the silt range (i.e. between 4 and <63 μm). The median grain size of the total sediment was, in general, slightly coarser with 89% sites in the silt range. The span of the grain size distribution of most samples was broad; with a number of samples having a bimodal distribution. The reach-averaged percentage of clay sizes (<4 μm) was always less than 22%, but the percentage sand-sized material (≥63 μm and <1 mm) ranged between 5 and 70% in the surface drape and between 10 and 81% in the total sediment (Fig. 2). As with the sediment mass variables, there was a close correlation between measures of absolute particle size in the surface drape and total sediment.

3.2. Temporal variability

The primary sites were sampled only once, with 73% sites being sampled in autumn 2010 or spring 2011. The sites in the supplementary dataset were each sampled twice – first in autumn and then in spring of the following year – and these sites were used to assess the influence of temporal variation in the deposited fine sediment which may be related to the timing of the sampling with respect to the flow regime. In general, the deposited fine sediment in the supplementary sites had a similar distribution of sediment mass and sediment characteristics to those of the primary sites. However, they did not include sites with extremely low sediment mass. There was also a tendency for more volatile solids and a slightly finer calibre of material (Table 2). All the supplementary sites were located in Wales and, as none of the sampled streams had flow data, the pattern of daily mean flows on the River Teifi at Llanfair in south-west Wales is used to illustrate the possible variation in river flows during the various sampling periods (Fig. 3).

Short-term temporal variability was assessed in two ways. First, the difference in logged values of sediment mass and volatile solids from autumn to spring was compared to the 95% confidence intervals derived from the uncertainty study of Duerdoth et al. (2015). For the total sediment, the observed difference in the reach-scale sediment mass for 50 of the 55 (91%) sites and in volatile solids for 48 of the 55 sites lay within the measurement error. For the surface drape, observed differences were greater but, for both the sediment mass and the volatile solids, observed differences in over 82% sites still lay within the measurement error. Those sites with significant changes in measured values (i.e. differences greater than measurement error) showed both loss and gain of sediment in both the total and the surface

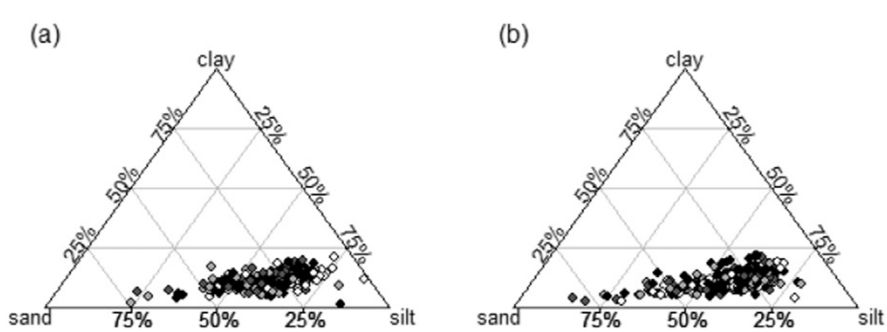

Fig. 2. Ternary diagrams giving percentage sand, silt and clay in (a) reach-averaged surface and (b) reach-averaged total bed sediments (grey scale indicates the number of samples on which the reach average is based from 4 (black) to 1 (white)).

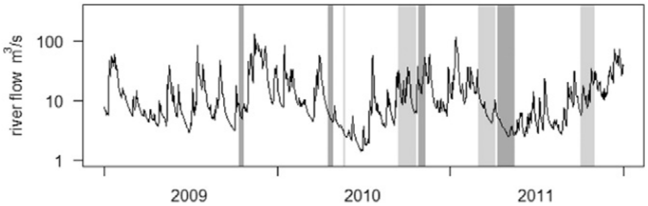

Fig. 3. Sampling periods overlain on mean daily flows (note logarithmic scale) for the River Teifi at Llanfair, south-west Wales. Light grey bars relate to primary sites; dark grey bars to the supplementary dataset.

Table 3
Significant relationships between deposited fine sediment and land cover.

Regression model	Adjusted R^2	Residual standard error	n[a]
Reach-averaged total sediment			
$\log TS = 2.718 + 0.0118\,AH$	0.308	0.479	163
$\log TS = 3.238 - 0.0113\,UGU$	0.365	0.501	194
Reach-averaged surface sediment			
$\log SS = 2.077 + 0.0096\,AH$	0.257	0.441	163
$\log SS = 2.552 - 0.0114\,UGU$	0.420	0.453	194

where *TS* is average sediment mass in surface drape and subsurface to a depth of approximately 10 cm (g/m^2), *SS* is average surface sediment mass (g/m^2), *AH* is percentage catchment area in LCM2007 class 3 (arable and horticulture) and *UGU* is percentage catchment area in LCM2007 classes 5, 8, 10–13 (unimproved grassland and upland).

[a] Number of catchments (zero % land cover omitted from relationships).

drape even though all comparisons were between samples taken in autumn and the following spring, after a relatively wet winter (Fig. 3). A second assessment of change was provided by looking at the correlation between pairs of measurements (i.e. measurement in autumn correlated with the equivalent measurement in spring). In all cases, the correlation was highly significant (total: sediment mass $\rho = 0.75$, volatile solids $\rho = 0.71$; surface: sediment mass $\rho = 0.67$, volatile solids $\rho = 0.66$; $p < 0.001$). Thus, it may be argued that taking single instantaneous samples may add scatter but it is unlikely to

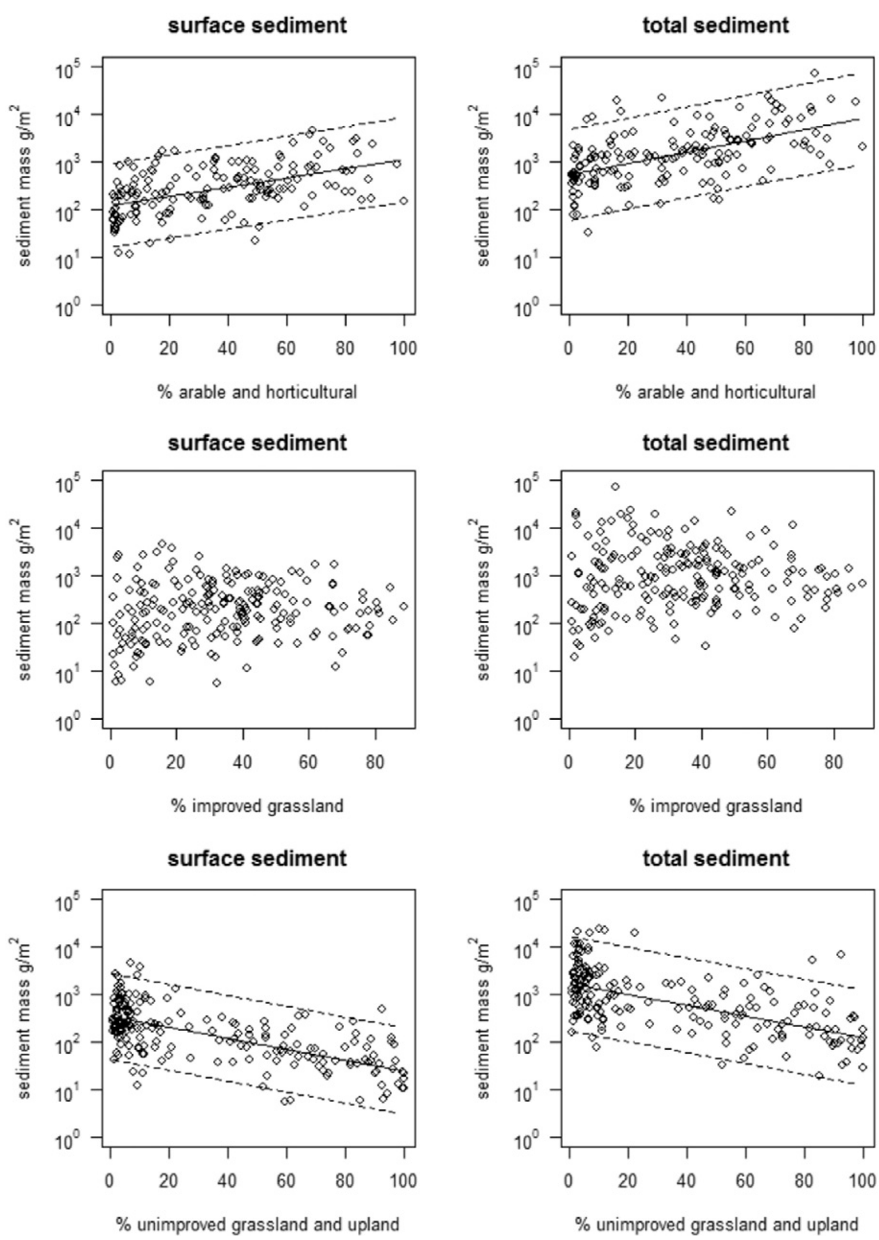

Fig. 4. Deposited fine sediment and catchment land cover: significant regression lines ($p < 0.001$) and 95% prediction intervals shown by solid and dashed lines, respectively.

Table 4
Spearman cross-correlation between reach-averaged mass of deposited fine sediment and potential explanatory variables (values with significance level $p < 0.001$ based on t test where $t = \rho\sqrt{[(n-2)/(1-\rho^2)]}$ with $(n-2)$ degrees of freedom (Siegel, 1956); only sites with no missing data used $n = 204$).

	Total sediment mass kg/m^2	Surface drape kg/m^2	Arable %	Improved grassland %	Unimproved/ upland %	Sediment pressure T/yr	Sediment yield T/km^2/yr	Catchment area km^2	Altitude m	Distance to source km	Strahler stream order	Channel slope m/km	MSUB phi	Bankfull width m	Water width m	Velocity category	HMS class	Median annual flood m^3/s	Stream power W/m
Surface drape kg/m^2	0.90																		
Arable %	0.68	0.67																	
Improved grassland %																			
Unimproved/ upland %	−0.59	−0.60	−0.84	−0.37															
Sediment pressure T/yr	−0.46	−0.40	−0.57		0.60														
Sediment yield T/km^2/yr	−0.62	−0.60	−0.69		0.64	0.74													
Catchment area km^2			−0.22		0.31	0.80	0.23												
Altitude m	−0.53	−0.52	−0.70		0.67	0.51	0.69												
Distance to source km	−0.28	−0.23	−0.37		0.44	0.84	0.36	0.93	0.24										
Strahler stream order	−0.34	−0.30	−0.48		0.48	0.77	0.42	0.77	0.34	0.78									
Channel slope m/km	−0.43	−0.45	−0.41		0.32		0.33	−0.47	0.43	−0.37									
MSUB phi	0.76	0.72	0.73		−0.65	−0.56	−0.60	−0.30	−0.58	−0.44	−0.48	−0.37							
Bankfull width m	−0.32	−0.26	−0.40		0.41	0.65	0.35	0.63	0.35	0.65	0.65		−0.50						
Water width m	−0.43	−0.36	−0.58		0.59	0.81	0.48	0.76	0.43	0.78	0.74		−0.59	0.70					
Velocity category	−0.52	−0.45	−0.49		0.45	0.44	0.39	0.28	0.40	0.33	0.32		−0.58	0.35	0.54				
HMS class	0.27	0.26	0.38		−0.30	−0.34	−0.34		−0.28	−0.23	−0.33		0.33		−0.34	−0.17			
Median annual flood m^3/s	−0.54	−0.48	−0.66		0.62	0.90	0.61	0.78	0.48	0.84	0.84		−0.66	0.69	0.86	0.48	−0.39		
Stream power W/m	−0.73	−0.68	−0.79		0.73	0.71	0.71	0.41	0.69	0.52	0.61	0.48	−0.80	0.54	0.68	0.51	−0.43	0.80	
Unit stream power W/m^2	−0.69	−0.67	−0.73		0.66	0.52	0.66		0.65	0.31	0.41	0.61	−0.69		0.49	0.42	−0.42	0.61	0.90
Baseflow index	0.43	0.51	0.57	0.31	−0.52	−0.47	−0.55	−0.22	−0.48	−0.35	−0.41		0.49	−0.26	−0.41	−0.24	0.26	−0.56	−0.57

fundamentally change the relationships found. It is assumed that this finding from the supplementary dataset applies to the single instantaneous measurements from the primary sites.

3.3. Relationship to land cover

Deposited fine sediment mass in both the surface drape alone and the subsurface to a depth of approximately 10 cm was significantly ($p < 0.001$) related to land cover (Fig. 4). In particular, sediment mass was positively related to the percentage of the catchment (above zero) of arable and horticultural land and negatively related to unimproved grassland and upland. There was no relationship with improved grassland, and amalgamating this class with either of the other two simply degraded those relationships. While these results were highly significant, there was a large degree of scatter, with arable land cover explaining only 25 to 31% of the total variance in deposited fine sediment (Table 3).

3.4. Relationship to other variables

Initial exploration of the available data showed a very high degree of cross-correlation amongst the selected catchment and channel descriptors (Table 4). Many of the high correlations simply revealed where different variables were indexing the same attribute e.g. catchment scale appears in catchment area, channel width, river discharge, stream power and modelled sediment pressure. Land cover variables were consistently highly correlated with other catchment descriptors — in particular, altitude, median annual flood and stream power; arable and horticultural land cover was the mirror image of unimproved grassland and upland. This implies that land cover, at this scale of analysis, may simply be a reflection of the fact that arable agriculture is found in the drier, low altitude parts of England and Wales while grassland is found in the wetter, upland areas. Percentage arable land cover was also inversely related to sediment pressure, despite its positive relation to deposited fine sediment.

In seeking to explain the mass of deposited fine sediment on the channel bed, it is therefore vital to understand how it varies with other catchment and channel descriptors. The highest correlation found was with channel substrate (*MSUB*) itself — a visual assessment which included the percentage of fines but which is not designed to address the issue of siltation, i.e. infiltration of fines into a gravel substrate or thin layers of silt covering coarser substrates (Murray-Bligh et al., 1997). In particular, the relationship with *MSUB* was found to be curvilinear, flattening off at a value of around 1200 g/m^2 for the surface sediment and 10,000 g/m^2 for the total (Fig. 5). Stream power showed the second highest correlation with deposited fine sediment, implying that the capacity of a stream to transport sediment is fundamental,

Table 5
River types based on hydromorphology (following Orr et al., 2008).

River type	River type Orr et al., 2008	Strahler stream order	Unit stream power Wm^{-2}	Slope %	No. sites
1	1/2	1 and 2	<20	<2.5	30
2	3/4	1 and 2	<20	>2.5	0
3	5/6	1 and 2	>20	<7.5	65
4	7/8	1 and 2	>20	>7.5	2
5	9	3 and 4	<50	–	25
6	10	3 and 4	>50	–	85
7	11	5	–	–	1

although strongly linked to many other catchment descriptors including some of those used to model sediment pressure. The negative relationship between deposited fine sediment and modelled sediment pressure (Table 4) is counter-intuitive and implies the importance of other factors in mediating this relationship.

3.5. Hydromorphological controls on substrate composition

The capacity of a stream to transport sediment may be characterised by its hydromorphology. Accordingly, the river typology developed by Orr et al. (2008) was applied. No data were available which indicated floodplain extent so there was no discrimination between some river types. This is not a serious limitation as the focus here is on relatively small streams. Based on stream order, specific stream power and slope, the sampled sites fell into six categories (Table 5). There were no sites in type 3/4 which are small streams with lower stream power but steeper slope and only one site with a stream order of 5.

Substrate (*MSUB*) varied significantly between hydromorphological river types. Ignoring river types with few sites, type 1 (low stream power and low slope) had significantly finer substrate than other types and type 6 (high stream power) significantly coarser substrate (Tukey HSD test; $p < 0.01$). Deposited fine sediment also varied with river type (Fig. 6). For the surface drape, there were significant differences (AOV; $p \ll 0.001$) in sediment mass; type 1 rivers had more fine sediment than types 3, 4 and 6, and type 6 rivers had less fine sediment than types 1, 3 and 5. Neither % volatile solids nor % sand-sized material in the surface drape differed significantly across river types. In the case of the total sediment (surface drape plus depth to approximately 10 cm), both mass of sediment and % sand-sized material showed significant differences between river types but only to the extent that type 1 had higher sediment mass and higher % sand-sized material than types 3 and 6. There was no significant difference in % volatile solids. The pattern of differences in fine sediment across hydromorphological types emphasises both the higher sediment mass found in lower order streams and the importance of unit stream

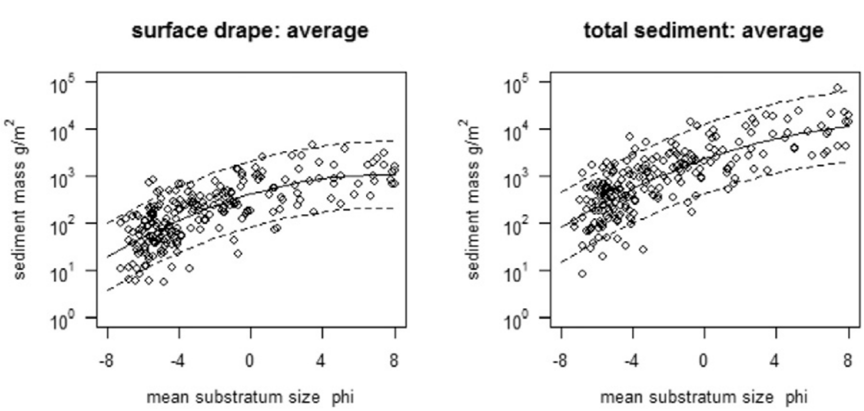

Fig. 5. Relationship between reach-averaged measured fine sediment and mean substratum size derived from visual assessment following protocol for RIVPACS environmental variables (Murray-Bligh et al., 1997); best fit polynomial regression lines and 90% prediction intervals shown.

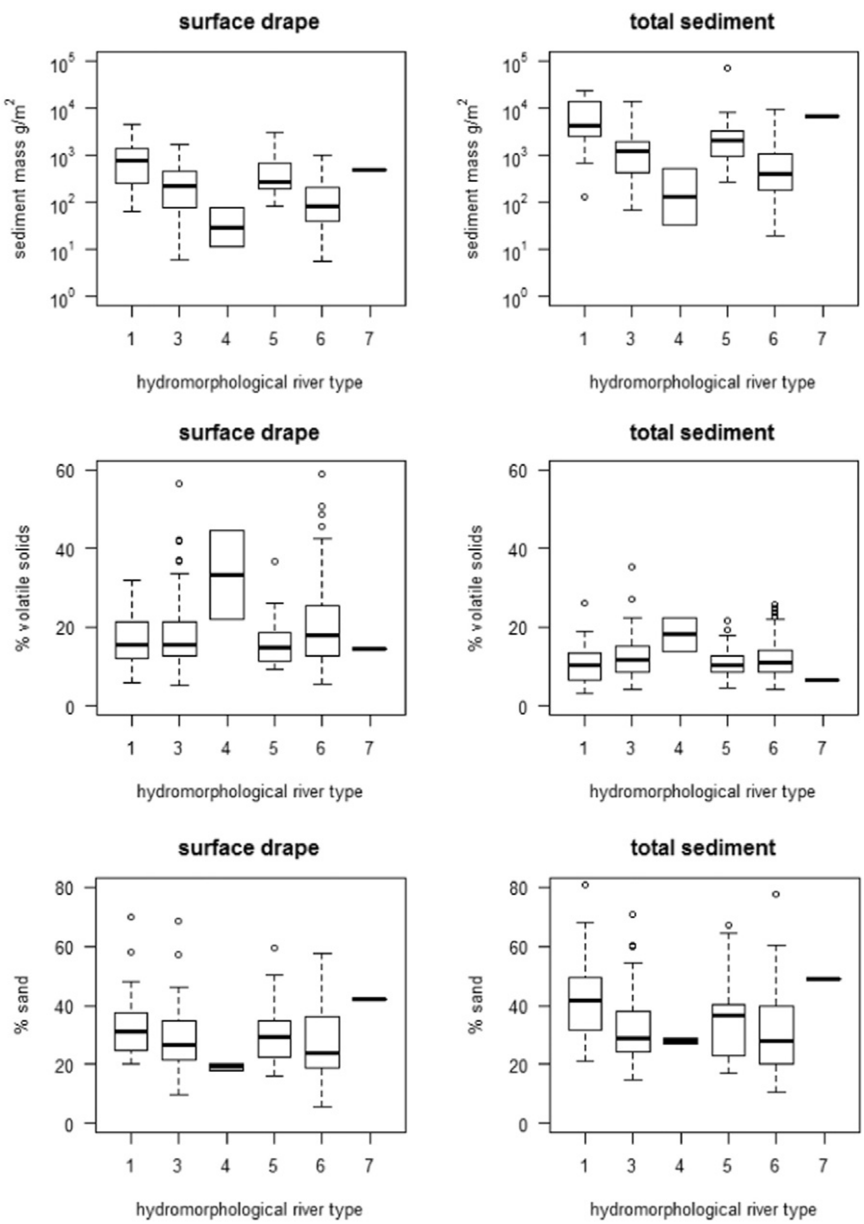

Fig. 6. Deposited fine sediment characteristics by hydromorphological river type; see Table 5 for definition of river types following Orr et al. (2008).

power — specifically, the link between low unit stream power and larger mass of deposited fine sediment.

3.6. Relationship of deposited fine sediment to modelled sediment pressure

To understand the link between deposited fine sediment and modelled sediment pressure, it was hypothesized that the mass of deposited fine sediment was (i) inversely related to the capacity of the stream to transport fine sediment, (ii) directly related to the amount of sediment delivered to the channel system, (iii) mediated by channel geometry, and (iv) influenced by flow regime, insofar as this describes the balance between potentially depositing and flushing flows, or the potential, in ground-water dominated systems, for fine sediment to be delivered to the channel during times of low flow. The measured sediment mass at any one site may also have been influenced by the time since the last flood event but it was not possible to index this dynamic temporal variation by the available national-scale data considered here. Given the degree of cross-correlation between variables (Table 4), model identification proceeded by selecting, in turn, alternative descriptors of transport capacity with modelled sediment pressure and other potential explanatory variables. The primary sites (Fig. 1) were used to derive the models; the supplementary sites (Fig. 1) were used for model assessment.

3.6.1. Total sediment

The most effective linear models for describing the reach-averaged total sediment mass are given in Table 6. Each of these models satisfied the diagnostics for constancy of variance and normality of residuals, and each of the retained terms was significant ($p < 0.05$). If categorical variables were included, the number of factors has been simplified such that individual parameter values were more than one standard error apart. Based on the Akaike Information Criterion (*AIC*), the first two models given for total sediment mass were not distinguishable from each other (relative likelihood given by exp. $(AIC_{min} - AIC_i) / 2 = 0.64$; Burnham and Anderson, 2002). The third alternative, based on specific stream power, was a poorer fit. Only the regression model based on stream power, calculated using the estimated median annual flood or approximately bankfull flow, included the modelled

Table 6
Best-fit linear models for explaining instantaneous data on deposited fine sediment.

Regression model	Adjusted R^2	Akaike information criterion	Residual standard error
Average total sediment			
$\log TS = 4.714 - 0.614 \log(\Omega) + 0.128 \log(TL) - 0.456\ (vc = 2) - 0.624\ (vc > 2)$	0.578	242.0	0.428
$\log TS = 4.379 - 0.473 \log(Q_{MED}) - 0.658 \log(S) - 0.472\ (vc = 2) - 0.639\ (vc > 2)$	0.580	241.1	0.427
$\log TS = 4.535 - 0.544 \log(\omega) - 0.477\ (vc = 2) - 0.734\ (vc > 2)$	0.553	253.0	0.441
Average total sediment — erosional zones			
$\log ETS = 4.622 - 0.690 \log(\Omega) + 0.147 \log(TL) - 0.525\ (vc = 2) - 0.752\ (vc > 2)$	0.617	265.9	0.454
$\log ETS = 4.255 - 0.526 \log(\Omega) - 0.741 \log(S) - 0.543\ (vc = 2) - 0.770\ (vc > 2)$	0.619	264.8	0.452
$\log ETS = 4.416 - 0.602 \log(\omega) - 0.549\ (vc = 2) - 0.878\ (vc > 2)$	0.585	281.4	0.472
Average total sediment — depositional zones			
$\log DTS = 4.922 - 0.492 \log(\Omega) - 0.428\ (vc \geq 2)$	0.402	317.1	0.516
$\log DTS = 4.703 - 0.350 \log(Q_{MED}) - 0.551 \log(S) + 0.704 \log(BFI) - 0.477\ (vc \geq 2)$	0.417	313.8	0.510
$\log DTS = 4.751 - 0.404 \log(\omega) + 0.669 \log(BFI) - 0.418\ (vc = 2) - 0.602\ (vc > 2)$	0.408	317.1	0.514
Average surface sediment			
$\log SS = 3.750 - 0.520 \log(\Omega) + 0.164 \log(TL) + 0.736 \log(BFI) - 0.344\ (vc \geq 2)$	0.500	234.4	0.420
Average surface sediment — erosional zones			
$\log ESS = 3.520 - 0.655 \log(\Omega) + 0.185 \log(TL) - 0.447\ (vc \geq 2)$	0.483	284.3	0.476
$\log ESS = 3.377 - 0.383 \log(Q_{MED}) - 0.641 \log(S) + 0.599 \log(BFI) - 0.484\ (vc \geq 2)$	0.486	284.1	0.474
$\log ESS = 3.353 - 0.533 \log(\omega) - 0.432\ (vc = 2) - 0.587\ (vc > 2)$	0.453	295.7	0.489
Average surface sediment — depositional zones			
$\log DSS = 3.885 - 0.375 \log(\Omega) + 0.949 \log(BFI)$	0.343	319.8	0.520
$\log DSS = 3.587 - 0.472 \log(\omega) - 0.161\ (vc = 3) - 0.376\ (vc > 3)$	0.333	324.1	0.524

where *TS*, *ETS* and *DTS* are averaged sediment mass (surface and subsurface to a depth of approximately 10 cm) for total, erosional and depositional zones respectively (g/m^2), *SS*, *ESS* and *DSS* are averaged surface sediment mass for total, erosional and depositional zones respectively (g/m^2), Ω is stream power (W/m), *TL* is modelled sediment pressure (T/yr), *vc* is velocity category, Q_{MED} is median annual flood (m^3/s), *S* is channel slope (m/km) and ω is specific stream power (W/m^2).

agricultural sediment pressure. In this model (Fig. 7a), total fine bed sediment had a highly significant relationship with stream power ($p \ll 0.001$) and velocity category ($p \ll 0.001$). Velocity category was taken as a very broad indication of the relative turbulence intensity of the flowing water, assuming that measurements were taken at roughly similar flow stages (low to medium flows rather than in spate as necessitated by the deployment of the disturbance technique used for sediment sampling). As turbulence intensity controls the ease with which sediment is maintained in suspension, it was expected that higher velocity categories would be associated with smaller amounts of deposited fine sediment as shown here. Only the two lowest categories were distinguishable from the rest of the data. The residual relationship between reach-averaged total sediment mass and modelled agricultural sediment pressure, although positive, was weak (Fig. 7b). This may be partly due to the fact that some of the variables used to calculate stream power are also instrumental in the modelling of sediment load. Analysis showed that 15% of the variance in modelled sediment load was not explained by these variables with catchment area contributing some 71% of the total variance in modelled sediment load but only 49% of the total variance in stream power. The predicted versus measured values of total sediment mass (Fig. 7c) gives an indication of the overall model fit for the primary sites; residual standard error was considerably higher than the measurement error (Duerdoth et al., 2015).

A similar analysis was undertaken using the mass of total sediment in erosional and depositional zones of the main channel separately. The relationships for erosional zones were similar to those for the reach average, although they were slightly stronger (Table 6), implying that these zones may be more indicative of modelled sediment pressure. In the case of depositional zones, the fitted models explained much less of the variability in total fine sediment and diagnostics revealed some pattern in the plot of residuals versus fitted values. Modelled sediment pressure was not a significant variable and the baseflow index, included in two of the relationships, was only marginally significant. Using the mass of non-volatile solids or the mass of the non-volatile silt-clay size fraction (assuming equivalence of fraction by volume and by mass) did not improve the relationship with modelled sediment pressure.

3.6.2. Surface drape

For the reach-averaged surface sediment mass, there was only one regression model which satisfied the diagnostics for acceptability and explained some 50% of the variation in the measured fine sediment mass (Table 6). Again, the most effective explanatory variable was stream power, calculated from the estimated median annual flood, (Fig. 8a) but four other variables were also significant: velocity category ($p = 0.0006$), baseflow index ($p = 0.004$) and modelled agricultural sediment pressure ($p = 0.007$). Each of these variables added about 2% explanation to the variation in surface fine sediment mass. Only the lowest velocity category was distinguishable from the others; with more fine sediment being associated with the lowest velocity category, as expected. There was a positive relationship with baseflow index as again might be expected; large amounts of fine sediment were associated with a high baseflow index indicative of steady seasonal variation in flow. A high baseflow index is associated with relatively few large flow events which might flush out fine sediment, and there is the potential for sediment delivery, from local impermeable areas or autochthonous production by instream biota, during times of low flow (Sear et al., 2008). There was also a highly significant positive residual relationship with modelled agricultural sediment pressure (Fig. 8b). The overall model for the primary sites (Fig. 8c) had a residual standard error higher than the measurement error (Duerdoth et al., 2015).

Separate analyses for erosional and depositional zones were less strong than the reach-averaged values for the surface drape (Table 6). The relationship with the baseflow index was less clear and, in depositional areas, the surface sediment mass showed no significant relationship with modelled sediment pressure. Again, the relationships were not improved by using the mass of non-volatile solids or the mass of the non-volatile silt–clay fraction.

3.6.3. Independent model assessment

The dataset relating to the supplementary sites (Fig. 1) was used as an independent assessment of the fitted model for the reach-averaged deposited fine sediment. The total sediment mass showed a somewhat wider scatter of values compared with the original dataset (Fig. 9a).

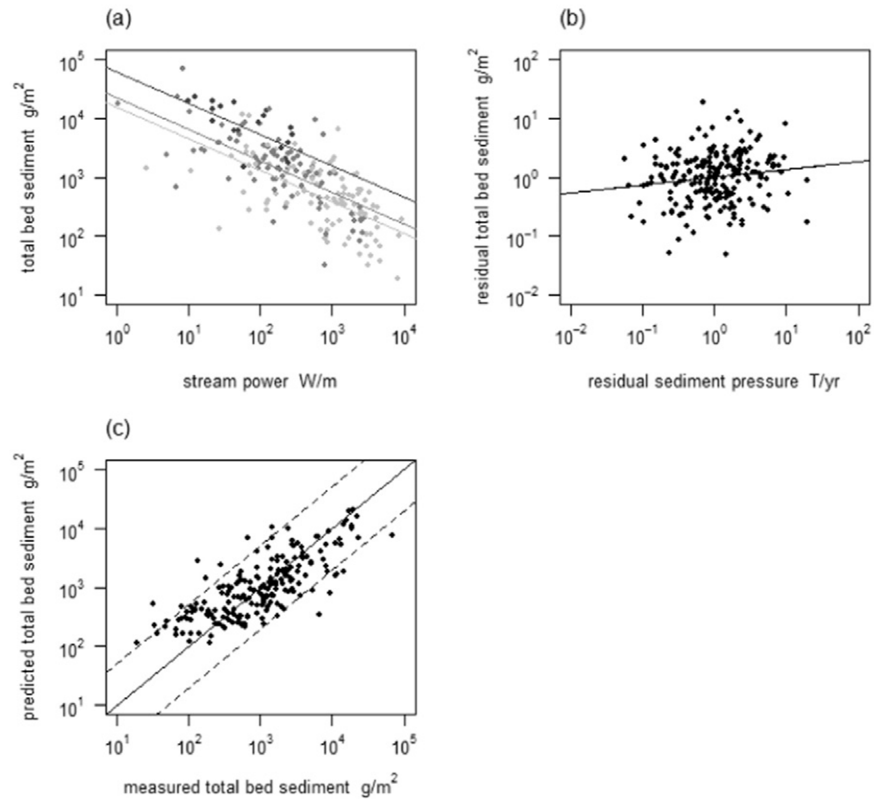

Fig. 7. Regression analysis for total fine sediment mass (primary sites): (a) relationship with stream power and velocity category (black: $vc = 1$; dark grey: $vc = 2$; light grey: $vc \geq 3$); (b) residual relationship with modelled sediment pressure, predominantly from agriculture; (c) predicted versus measured total fine sediment mass showing 1:1 line and 90% confidence intervals.

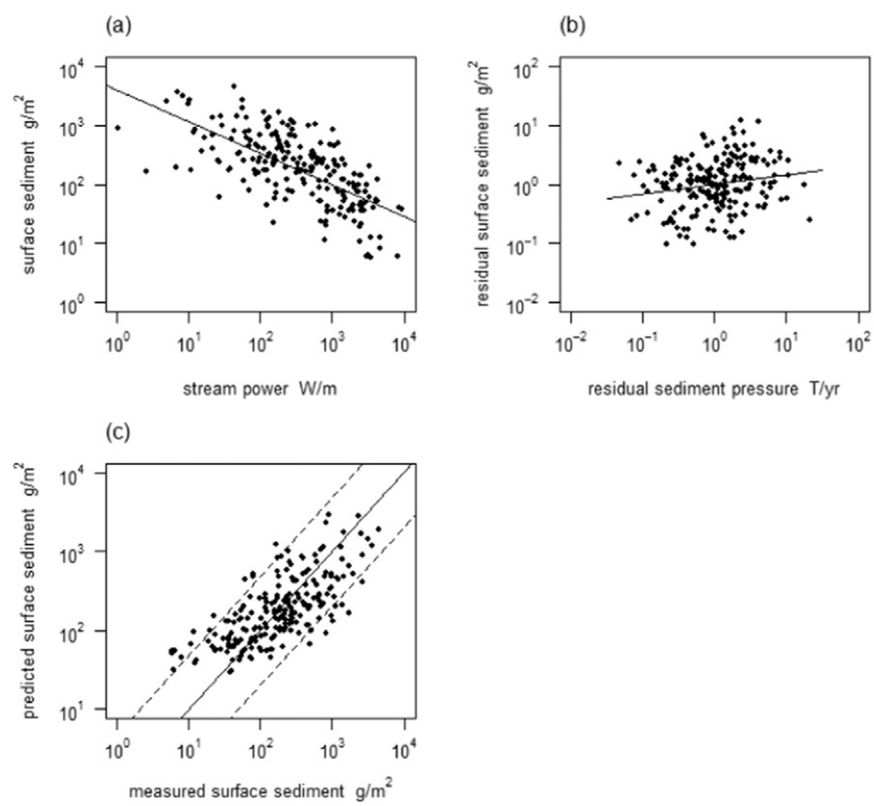

Fig. 8. Regression analysis for surface fine sediment mass (primary sites): (a) relationship with stream power; (b) residual relationship with modelled sediment pressure, predominantly from agriculture, taking account of stream power, velocity category and baseflow index (see Table 6); (c) predicted versus measured surface fine sediment mass showing 1:1 line and 90% confidence intervals.

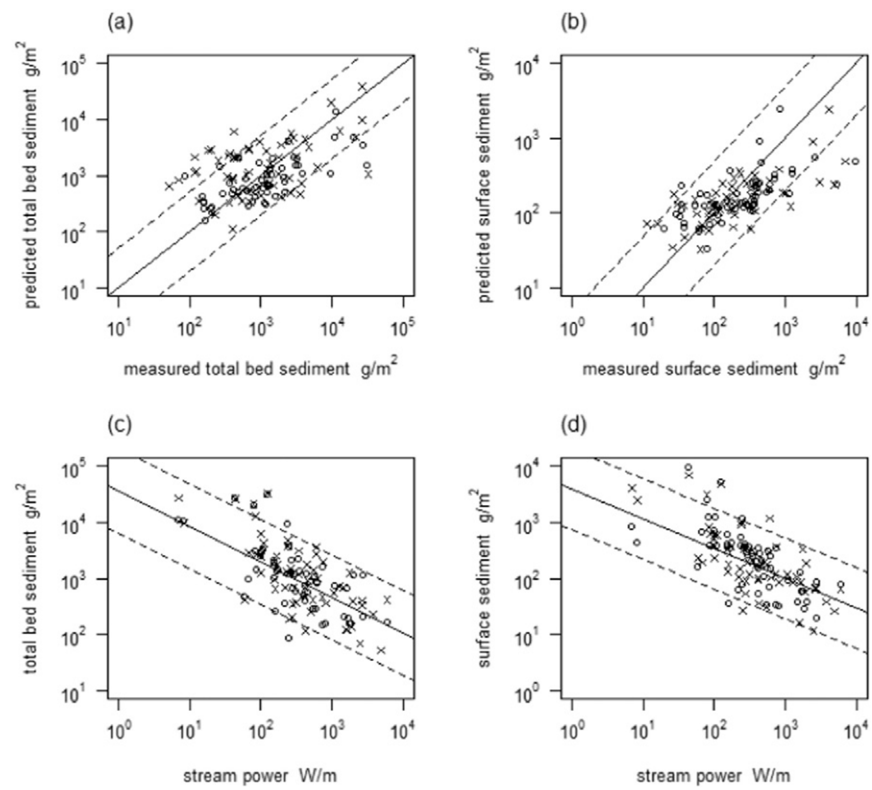

Fig. 9. Assessment of regression relationships using independent dataset from supplementary sites (measurements taken in autumn × and spring ○): (a) measured and predicted reach-averaged total bed sediment; (b) measured and predicted reach-averaged surface sediment; (c) relationship between total bed sediment and stream power; (d) relationship between surface sediment and stream power. In all cases, relationship from analysis of primary dataset with 90% prediction intervals is shown.

In particular, several of the autumn measurements fell outside the 90% confidence band, with the model overestimating the amount of deposited fine sediment. Most of the spring measurements fell within the 90% confidence band but here there was a tendency for the model to underestimate the measured values. By contrast, the reach-averaged surface sediment mass showed a similar spread of values compared with the original dataset (Fig. 9b). However, there were a few outliers where the model seriously underestimated very high values of measured deposited sediment. These were equally present in the autumn and spring samples. For the supplementary sites, the relationship between measured deposited fine sediment mass and stream power showed a similar fit to that of the primary sites for both the surface and total sediment, with little or no discrimination between seasons (Fig. 9c, d).

4. Discussion

The data presented in this paper provide improved spatial coverage in the quantification of instantaneous fine sediment storage within streams across England and Wales and offer a unique baseline snapshot of substrate condition for assessment of future change at the sampled sites. Previously published data for the UK has mostly focused on large rivers with moorland headwaters (Owens et al., 1999; Walling et al., 1998) and lowland groundwater-dominated rivers (Collins and Walling, 2007b, 2007c), albeit that these more spatially constrained datasets provide better temporal coverage (typically two years of monthly or every other month sampling).

The data presented also extend the characterisation of deposited fine sediment by including both non-volatile solids and measures of absolute particle size. The percentage of volatile solids is an important measure for linking to biota as this relates to availability of nutrients through decomposition and a food source for aquatic organisms. Critically, decomposition of organic matter can lead to reduced interstitial oxygen concentration, a key stressor on aquatic organisms (Jones et al., 2012a, 2012b; Sear et al., 2014), and crucially important for nutrient transformation pathways and the production of ammonia (e.g. Pretty et al., 2006; Trimmer et al., 2009). The organic component of deposited fine sediment is thus a critical, yet with notable exceptions (e.g. Marttila and Kløve, 2014, 2015) often overlooked, determinant of biological response to fine sediment pressure (Collins et al., 2009c; Murphy et al., 2015). Indeed, Von Bertrab et al. (2013) go further to suggest that the chemical composition of deposited sediment is more important to benthic macroinvertebrate assemblages than the amount of sediment. The percentage organic matter and associated sediment oxygen demand are also recognised as important parameters for fish egg survival (Olsson and Persson, 1988; Greig et al., 2005, 2007; Sear et al., 2014; Sear et al., 2016).

Data on the absolute particle size of fines (<1 mm) indicate that, although the silt/clay size fraction is most associated with agricultural runoff, there was a large variation in the percentage sand-sized particles present. This is an interesting finding in the context of the clogging of gravel substrates. Sand can more easily bridge pore spaces within gravels such that finer and, critically, organic material is more easily trapped (Warren et al., 2009), thus reducing flow through the gravel and the exchange of oxygen-rich waters. For river management, it is therefore important to understand the source of the sand-sized material and its transport regime (Collins et al., 2009c), in addition to the more usual source apportionment of the finer size fractions (e.g. Walling et al., 2003b; Collins et al., 2012c, 2012d). The relatively large amounts of sand-sized particles are consistent with previously published findings (Milan et al., 2000; Julien and Bergeron, 2006). Intuitively, on the basis of limited transport distances, eroding channel banks may be a key contributor to the sand-sized particles, thus, driving important process linkages in the river substrate that impact on aquatic ecology.

4.1. Relationship with land cover

A number of studies have found strong positive relationships between deposited fine sediment and percentage of land use under agriculture in small to medium catchments (Table 7). However, those studies which have reported a high correlation between fine sediment and land use are generally those where sites range from near-pristine to highly impacted, where sediment pressure from agriculture is high, e.g. potato production (Sutherland et al., 2010) or intensive pasture (Niyogi et al., 2007), and where the range of geomorphological variation is relatively small. In our study, while there is a significant relationship between deposited fine sediment and % arable and horticultural land, the latter is highly correlated with other catchment descriptors and negatively correlated with modelled sediment pressure, suggesting a more complex linkage to deposited fine sediment. Indeed, both Anlauf and Moffitt (2010) and Sutherland et al. (2010) report that variation in fines was almost equally explained by *either* percentage agriculture *or* stream slope. Hence, it is important to develop a more process-based understanding of what controls the amount of deposited fine sediment sequestered in stream beds.

4.2. Dominant drivers and relationship to modelled sediment pressure

In our study, the most effective explanatory variable for the amount of deposited fine sediment was found to be stream power, calculated from the estimated median annual flood or approximately bankfull flow. This is a measure of the ability of a stream to transport sediment, but it is also correlated with many other factors. Other variables which had a statistically significant, but small, contribution were stream velocity category, modelled agricultural sediment pressure and, in the case of channel bed surface deposition, flow regime indexed by BFI. The identified model structure (Table 6) accorded with expectations and explained 50–60% of the variation in the measured deposited fine sediment.

Other studies have also consistently identified stream slope (a contributor to stream power) to be a dominant geomorphic factor (Walters et al., 2003; Anlauf and Moffitt, 2010; Sutherland et al., 2010; Relyea et al., 2012). Anlauf and Moffitt (2010) also found slow water habitat to be a significant predictor of deposited fine sediment alongside the percentage of agricultural land in the catchment. Stream order has also previously been identified as an important contributory factor, which suggests a need to understand how the balance between sediment supply and transport capacity changes downstream. For example, Relyea et al. (2012) reported that first order streams had more fine sediment than all other Strahler orders, and that 4th and 5th order streams had less fine sediment that lower orders. Similarly, Wagenhoff et al. (2011) found positive relationships between suspendable inorganic sediment (SIS) and % catchment runoff from pasture, an indication of sediment delivery, for all stream orders except the lowest in their study (third order). A similar tendency was seen in our data, suggesting that it is the lower order streams which are more likely to be impacted by deposited fine sediment; perhaps partly as a result of the strong coupling between low-order streams and their catchment.

The spatial scale of any analysis is fundamental to understanding the controls on fine sediment deposition, as is due recognition of the co-variation within the dataset. Despite sampling agricultural streams across a gradient of modelled sediment pressure, this was not found to be a key driver of deposited fine sediment. One reason for this was the substantial variation in catchment hydrogeomorphology across the sites. This is a driver of *both* sediment pressure and in-stream transport, as indexed by stream power at approximately bankfull flow. Sites with high modelled agricultural sediment pressure also had high stream power and relatively small amounts of deposition, implying that these streams could carry much of the delivered sediment. Sites with low modelled sediment pressure had low stream power and large amounts of deposited fine sediment, implying that these streams were limited in their transport capacity with respect to even relatively low sediment pressure. Clearly these linkages need to be interpreted in the context of stream power being a function of other physical factors (e.g. slope), correlated with other variables including land use, and the longer-term temporal basis of the modelled agricultural sediment pressure. Despite these limitations, the findings have important implications with respect to setting sediment load targets to avoid excessive deposition as it suggests that, at least for small catchments, such targets should be dependent on the transport capacity of the receiving channel. The approach to target-setting based on measured in-stream sediment loads developed by Cooper et al. (2008) partly takes this into account by default. As a result, Cooper et al. proposed a much more stringent target for chalk streams than other river types. However, Cooper et al.'s empirical approach cannot distinguish those streams with low sediment load due to limited sediment supply from those with a low transport capacity. Thus, it is clear that target-setting for sediment loads demands a much more robust approach taking into account sediment delivery, transport capacity, bed mobility and biological sensitivity (Sear et al., 2008; Collins et al., 2011; Bilotta et al., 2012).

4.3. Another potential explanation

Another aspect of the relationship between deposited fine sediment and modelled agricultural sediment pressure can be explored by considering the capacity of the substrate to sequester fine sediment. It is clear that different substrates can accommodate different amounts of fines dependent on their pore space and ease of ingress. Wooster et al. (2008) defined the saturated fine sediment fraction (FSF) as a function of the geometric standard deviation of the grain size of both the substrate framework and the fine sediment matrix, and their relative grain sizes. A rough conversion of the saturated FSF into mass of fine sediment per unit area can be achieved using our measurement

Table 7
Published relationships between deposited fine sediment and land use.

Source	Measure of fine sediment	Measure of land use	R^2 (%)	No. sites	Location
Walser and Bart (1999)	Sediment index based on fine sediment depth	% agricultural land	43	14	Chattahoochee River Georgia, USA
Niyogi et al. (2007)	Mass of suspendable inorganic sediment (depth 5 cm)	% pasture land	59	21	Otago Province New Zealand
Sutherland et al. (2010)	% fines <2 mm by mass from shovel cores	% land under potato production	67	15	New Brunswick Canada
Anlauf and Moffitt (2010)	% bed area classed as fines <2 mm	% agricultural land	75	56	Salmon River Idaho, USA
Wagenhoff et al. (2011)	Mass of suspendable inorganic sediment (depth 5 cm)	% catchment runoff from pasture	27	43	Southland Province New Zealand
This study	Mass of total suspendable sediment (depth ca. 10 cm)	% arable and horticultural land	31	163[a]	England and Wales

[a] excludes catchments with no arable land cover.

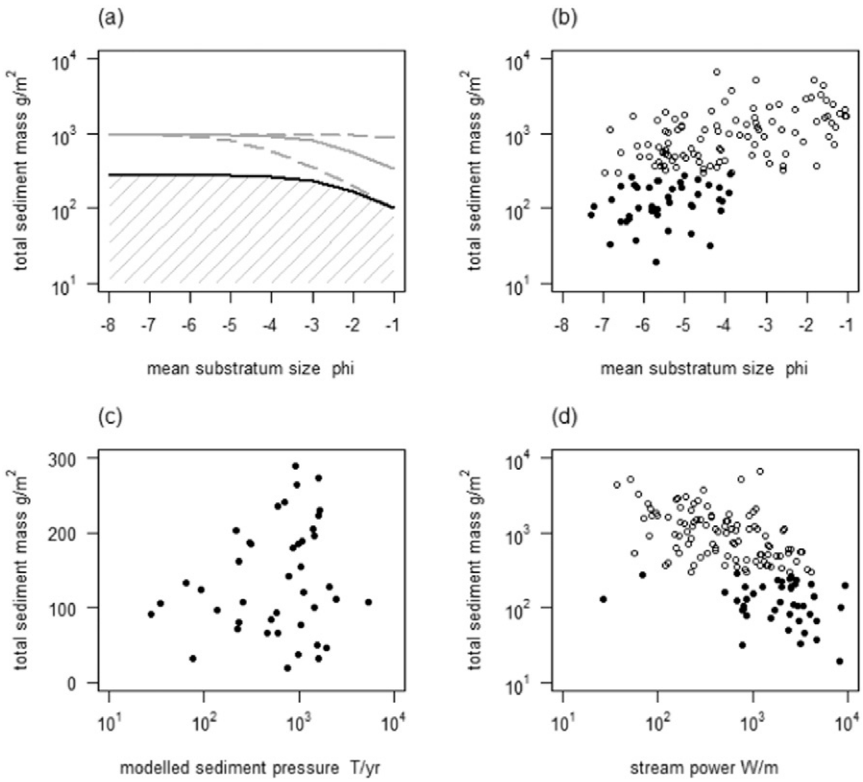

Fig. 10. Analysis of unsaturated substrate for sites with *MSUB* ≤ −1: (a) calculated saturation following Wooster et al. (2008): solid grey line shows saturation level for silt-sized fines in uniform substrate; dashed grey lines show how this varies with the maximum and minimum D_{50} of fines measured in this study; solid black line shows saturation level for silt-sized fines in highly non-uniform substrate, shading below this identifies most likely unsaturated substrate; (b) measured total sediment mass versus mean substratum size with solid circles indicating those sites likely to be unsaturated; (c) measured total sediment mass against modelled sediment pressure for sites likely to be unsaturated by fines; (d) measured total sediment mass versus stream power with solid circles indicating those sites likely to be unsaturated.

depth of approximately 10 cm and an assumed particle density of fine sediment. For the purposes of this argument, a particle density of 2485 kg/m^3 has been assumed. For the coarser range of mean substratum size (2 to 256 mm i.e. coarse sand to cobbles), the calculated mass of fine sediment at saturation varied between about 100 and 1000 g/m^2 (Fig. 10a) dependent on the assumed grain size of the fines and the uniformity of the substrate. Assuming that the fine material is silt-sized (0.063 mm) and that the substrate is highly non-uniform (geometric standard deviation 4), the shading in Fig. 10a indicates the substrate which was most likely to be below saturation.

By comparing this with the measured total sediment mass for a given mean substratum size (Fig. 10b), it appeared highly likely that the majority of the sampled sites were saturated with fines. This may help to explain the weak relationship with modelled sediment pressure, although other potential factors may be at play here, including the much longer temporal basis of the modelled sediment pressure. Based on the analysis above, there were 42 sites with a mean substratum size coarser than −3 phi units (i.e. >8 mm) and measured total fine sediment mass less than 300 g/m^2 which were unlikely to be saturated with fines. The scatterplot of measured total fine sediment mass and modelled sediment pressure for these sites had a wedge-shaped distribution (Fig. 10c). The upper limit of deposited fines clearly increased with the modelled sediment pressure. Below the upper limit, smaller amounts of deposited sediment were then perhaps a reflection of the temporal dynamics of the siltation process, such as the sequence of recent flow events leading to disturbance or washout of fines and the local rate of siltation coupled with the elapsed time since disturbance. Thus, this subset of sites appeared not only to be unsaturated with respect to deposited fines but also supply-limited; the dominant driver for the envelope curve was modelled sediment pressure, predominantly from agriculture, and there was no relationship between deposited fine sediment and stream power (Fig. 10d).

The hypothesis that the majority of the sampled sites may be saturated with deposited fines requires further work — particularly with respect to field testing and proper evaluation of the parameters required in the model proposed by Wooster et al. (2008). However, the possibility of splitting sites into saturated and unsaturated substrates does provide a useful new perspective for understanding the controls on deposited fine sediment in agricultural streams. It was only in unsaturated sites that modelled sediment pressure, predominantly from agriculture, seemed to dictate the amount of deposited fine sediment. This has important implications with respect to the implementation of agricultural mitigation measures to reduce sediment pressure in that, if most agricultural streams are saturated with fines, then simply reducing sediment delivery may have little immediate impact on deposited stream sediment. Additional river management may be needed to mobilise or extract the existing fines, especially in cases where bed material is not naturally mobilised during bankfull or larger events.

Traditionally chalk stream management has included regular gravel cleaning (Shackle et al., 1999) and there have been a number of recent studies which have explored the effectiveness of substrate restoration by either cleaning or addition of clean gravels (Merz and Setka, 2004; Meyer et al., 2008; Geist and Sternecker, 2013; Pulg et al., 2013). In these studies, improvements to physical habitat, in terms of both fine sediment content and compaction of the substrate; hyporheic water quality, including increased oxygen supply and reduced concentrations of nitrite and ammonium; and biota have all been reported. However, the length of time over which improvement in habitat was maintained varied from 5 months to 5 years. Presumably, this is a function of fine sediment delivery and reinforces the need to implement mitigation measures to reduce sediment pressure in tandem with river channel management (Greig et al., 2005). Another important consideration is the potential for negative impacts in downstream sites; for example, Geist and Sternecker (2013) reported significantly increased sediment deposition for 1 km downstream of a restored site. An understanding of the controls on siltation and how these change downstream is, therefore, vital to effective holistic management of river systems.

5. Conclusion

Deposited fine sediment was characterised in 230 streams, representative of different biological stream types, across a gradient of modelled agricultural sediment pressure, thus providing a systematic survey of deposited fine sediment across England and Wales. The data offer a unique snapshot of substrate condition, across a wider range of river types than hitherto reported, for the assessment of biotic impact and future change.

Deposited fine sediment was found to be predominantly related to stream power, calculated from the estimated median annual flood, rather than modelled sediment pressure, which, for the measured sites, is largely from agriculture. These results are consistent with previously published studies in so far as they relate to small streams of low Strahler stream order which are impacted by agriculture and have a high variation in their hydrogeomorphology — a driver of *both* sediment pressure and in-stream transport. Thus, it is suggested that the majority of the sites were essentially transport-limited and an analysis, in terms of substrate capacity to hold fine sediment, implied that most of the sites were saturated with respect to fine sediment. Below the level of saturation, there was some indication of a positive relationship between the maximum amount of deposited fine sediment and modelled sediment pressure which provided an upper envelope for those sites which may be considered to be supply-limited. Further work is needed to develop and test this idea in the field.

There are two important implications of these findings:

- future proposed targets for sediment loads need to take into account channel hydromorphology — specifically, the ability of streams to transport/retain fine sediment;
- river management to mobilise/remove fines from the bed should be considered in conjunction with mitigation measures for reducing delivery of fine sediments for those streams identified as being already saturated with fines and unlikely to self-cleanse. In this case, due care will need to be exercised with respect to potential downstream impacts.

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Mapping the combined risk of agricultural fine sediment input and accumulation for riverine ecosystems across England and Wales

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ABSTRACT

Fine sediment inputs from agricultural sources are a potential threat to freshwater ecosystems and may impact on the ability of EU members' states to achieve environmental targets under the Water Framework Directive (WFD).

An index (the Agricultural Sediment Risk index or ASR) representing the risk of agricultural fine sediment accumulation in rivers was produced using estimates of sediment inputs from the process-based PSYCHIC model and predictions of fine sediment accumulation using River Habitat Survey data. The ASR was mapped across the entire river network of England and Wales.

The ASR map and index were combined with a national dataset of fisheries surveys using logistic regression to test its relevance to freshwater biota. The ASR was strongly associated with a group of species sensitive to fine sediment inputs including salmon and trout. Another group of species including roach and perch showed a positive association with low levels of agricultural sediment inputs potentially due to their impacts on predators and competitors.

The proposed approach demonstrates how existing national monitoring data and sediment pressure models can be combined to produce an assessment of risk to aquatic ecosystems from agricultural fine sediment sources at a national scale that can be used alongside WFD classification tools to identify potential causative pressures and design remedial actions.

1. Introduction

With increasing environmental pressures on rivers and their ecosystems, there is a need for simple, robust tools to support environmental management decision-making (Bainbridge, 2014). In Europe, the Water Framework Directive (WFD) requires member states to bring rivers to Good Ecological Status (GES) between 2015 and 2027 by reviewing existing activities and undertaking targeted remedial action (European Union, 2000).

Agriculture is considered a significant pressure on aquatic ecosystem health through the elevated inputs of nutrients, pesticides, herbicides and sediment and their impact on natural populations of fish, invertebrates, macrophytes and diatoms (Collins et al., 2011; Duerdoth et al., 2015; Gayraud et al., 2002; Jones et al., 2012a, 2014; Kemp et al., 2011). Fine sediment from an agricultural origin currently represents the majority of total fine-grained sediment delivered to watercourses across England and Wales, with an estimated 72–76% of all fine sediment considered to originate from this source (Collins et al., 2009a,b; Zhang et al., 2014).

Fine sediment (defined here as inorganic and organic particles of less than 2 mm in diameter) are known to have both positive and negative impacts on instream ecosystems whether directly (e.g. smothering and clogging) or indirectly (e.g. as vectors for contaminants). They can have direct impacts on fish species by clogging gills, reducing oxygen availability to incubating embryo, increasing stress levels, reducing visibility, carrying pollutants and modifying the morphological structure of habitats (Collins et al., 2011; Kemp et al., 2011; Kjelland et al., 2015). They can also have indirect impact on fish behaviour, feeding, swimming ability and reproduction thereby imposing longer term effects on population structure and resilience (Kjelland et al., 2015). Fine sediment also affects

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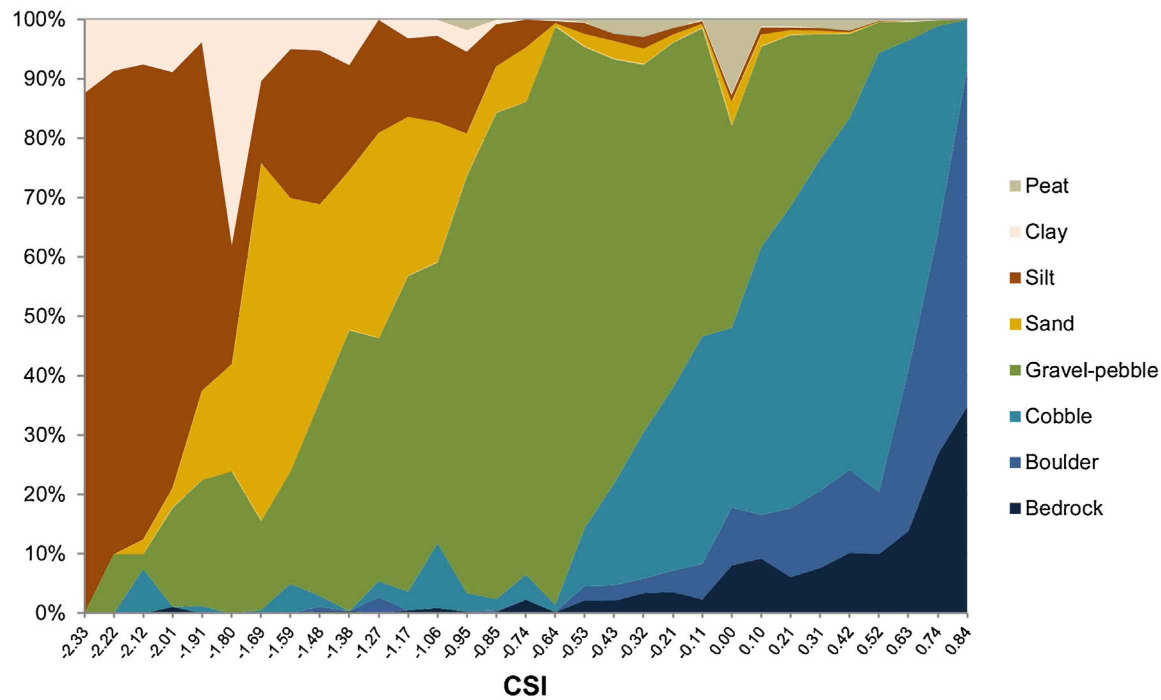

Fig. 1. Channel Substrate Index. RHS sites were grouped into 31 bins based on their CSI index value. The graph displays, for each bin, the average occurrence of 8 channel substrate types. Fine sediment (sand, silt, and clay) are dominant at the lower end of the scale and are gradually replaced by coarser sediments as CSI increases.

macro invertebrates via accumulation on and within the river substrate (Jones et al., 2011; Wood and Armitage, 1997), and through increased concentrations within the water column (Gayraud et al., 2002). Channel sediment size is a key element explaining aquatic macrophyte distribution (Gurnell et al., 2010). Fine sediment and macrophytes interact in complex ways. Fine sediment deposition on river margin favours the settlement and growth of emergent vegetation whose leaves, roots and shoots locally reduce flow velocities leading to further sediment entrapment and accumulation (Clarke and Wharton, 2001; Jones et al., 2012a; Sand-jensen, 1998 Sand-jensen, 1998). Fine sediment and macrophyte interaction encourages channel recovery in widened streams through the development of marginal benches and banks and subsequent reductions in channel width (Gurnell, 2014).

The diffuse nature of sediment inputs makes fine sediment management problematic, especially at catchment scale (Collins et al., 2011). The presence and accumulation of fine sediment in streams is dependent on a series of factors, including: precipitation (intensity and total), land management practice (e.g. tillage), the presence of pathways to rivers, channel morphology, channel modifications, impounding structures, flow regime, sediment transported from upstream, and instream vegetation communities (Bilotta et al., 2008; Collins et al., 2009b, 2011). The complex interaction of all these factors makes it difficult to predict accurately where and how much fine sediment will accumulate in a water body and more importantly its origin. As a result, there are no detailed (<10 km^2) spatial data characterising fine sediment accumulation across rivers, either globally or nationally.

The effective management of fine sediment is also limited by the structure and nature of existing decision-making. Organisations responsible for policy development, environmental management and the implementation of European directives are subject to continued resource cuts in the face of ongoing economic challenges, meaning that national scale monitoring is constantly being rationalised, thereby increasing the need for robust modelling approaches to support strategic decision-making (Collins and McGonigle, 2008; Naura, 2014). On this basis, there is a need to develop simple modelling tools for predicting agricultural sediment levels in rivers that can be easily applied to fine sediment management by regulatory bodies, and that permit strategic extrapolation in the context of the limited availability of data and knowledge on fine sediment origin and delivery (Bainbridge, 2014; Collins and McGonigle, 2008; Collins et al., 2009c).

One approach that has been widely used in environmental organisations is risk assessment. Risk assessment is one means of identifying potential levels of threats posed by contaminants based on data, models or expert opinion (Fairman et al., 1999). Risk levels can easily be represented in the form of maps and communicated to all stakeholders (Zerger, 2002). In the absence of specific or accurate data sources and knowledge, risk assessment may provide a meaningful way of supporting decision-making using existing resources (Jones, 2001). To the users, the relative simplicity and openness of outputs and derivation process may bring clarity and transparency and foster trust.

In this paper, we develop a risk-based approach towards assessing the likelihood that accumulated fine sediments on the river bed are of agricultural origins and we test the resulting fine sediment index on existing biological monitoring data. We choose fine sediment accumulation rather than concentration within the stream, for the following reasons: (a) data on fine sediment accumulation on the stream bed are more widely available and relatively simple to measure; (b) accumulation represents both the concentration of fine sediment in the water column and the deposition rate of entrained sediment, and; (c) it has been shown to be a major cause of change in biological communities (Jones et al., 2012a, 2014, 2012b).

The risk of fine sediment accumulation was assessed by combining a map of fine sediment distribution produced with spatially explicit predictive models based on existing River Habitat Survey (RHS) data (Naura et al., 2016), with a map of agricultural fine sediment inputs derived from the sediment module of the process-based ADAS Pollutant Transport (APT) framework (Collins et al., 2012b; Davison et al., 2008; Zhang et al., 2014). The correlation between the final risk map and aquatic biota was tested statisti-

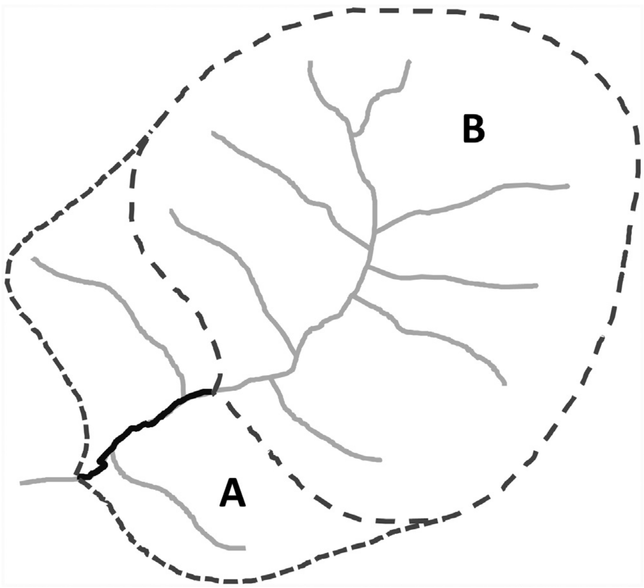

Fig. 2. Derivation of agricultural fine sediment inputs and river length for the calculation of Agricultural Sediment Load as part of the ASL equation for a 500 m section (in bold). A- sub-catchment area directly feeding into the example 500 m reach; B sub-catchment area upstream generating fine sediment also entering the example 500 m reach in sub-catchment A. The network length in sub-catchments A and B correspond to LRN and CRN, respectively in the ASL equation. The agricultural fine sediment inputs terms, LS and CS, are derived using the respective sub-catchment boundaries for A and B.

cally using the Environment Agency (EA) National Fish Population Database (NFPD) and predictions of natural fish populations using the Fisheries Classification Scheme (FCS) (Wyatt, 2003). Further validation could be undertaken in the future using national scale invertebrate or macrophyte datasets. These additional datasets were not available to this project.

2. Material and methods

To produce the agricultural fine sediment risk map, two indices were derived: the Fine Sediment Accumulation index (FSA) which represents the extent of fine sediment (i.e. sand, silt, and clay) on the river bed, and the Agricultural Sediment Load index (ASL) which provides an estimate of the amount of fine sediment from agricultural origin delivered to individual reaches through run-off from agricultural land and channel network transport.

2.1. Fine sediment accumulation index

Fine sediment accumulation was mapped using an index of sediment size called the Channel Substrate Index (CSI) derived as part of prior research using RHS data (Naura et al., 2016). RHS is a standard methodology for hydromorphology assessment under the WFD that has been implemented at more than 25,000 sites across the UK since 1994. During a River Habitat Survey, a visual estimate of the dominant channel substrate is recorded at a series of 10 equally spaced transects along a 500 m reach (Raven et al., 1997). Each site can be described according to the relative occurrence of nine substrate types across 10 transects. The CSI was derived using Correspondence Analysis on 2680 semi-natural RHS sites (i.e. sites with few or no in-channel bank structures/modifications) and represents average channel substrate size along a continuous scale from fine to coarse sediment (Fig. 1). The CSI index was modelled against a series of GIS attributes representing gradients of geomorphological change (e.g. slope, geology) using a geostatistical technique called regression kriging (Webster and Oliver, 2007). The resulting model was applied to every 500 m section on the 1:50,000 river network across England and Wales to produce a national map of river substrate sediment size distribution (Naura et al., 2016). The Fine Sediment Accumulation (FSA) index was created by partitioning CSI values into 5 categories to reflect the likelihood of fine sediment occurrence and their extent. Partitioning was undertaken by manually splitting the CSI scale based on the relative occurrence of sand, silt and clay in each category.

2.2. Agricultural sediment load index

The ASL index was derived using estimates of fine sediment delivered to rivers across England and Wales using the APT model (Collins et al., 2012a; Zhang et al., 2014).

APT builds upon the widely used and validated PSYCHIC (Phosphorus and Sediment Yield CHaracterisation In Catchments) model (Collins and Anthony, 2008; Collins et al., 2009a,b; Collins et al., 2014a,b; Collins et al., 2007; Comber et al., 2013; Davison et al., 2008; Strömqvist et al., 2008) for agricultural emissions to rivers. APT simulates fine sediment loss from agricultural land and woodland and estimates the load delivered to watercourses. It operates at a daily time step and can output at a 1 km^2 spatial resolution. APT simulates sediment losses at field scale, with a WFD water body represented as a large number of fields which are then subject to landscape scale retention factors to estimate delivery of mobilised fine sediment from agricultural land to the river network. Critically, field drainage as a sediment delivery pathway is represented, as well as surface runoff. The APT model uses as input three types of data; daily weather, physical attributes of the land, and crop and livestock management data. The daily weather data was interpolated for each WFD water body from existing UK Meteorological Office records using an inverse distance weighting function in the IRRIGUIDE tool (Bailey and Spackman, 1996). During the simulations, a WFD water body is represented by a small number of major soil types taken from the NSRI Natmap Soils Database. Other physical data required as input include slope and altitude, plus field boundary features (based on the Countryside Survey; Hornung, 1998) which are a key control on agricultural land-to-river connectivity. Crop areas were based on the 2010 June Agricultural Survey completed by farmers in England and Wales, which has been mapped to a 1 km grid using the approach described in Comber et al. (2013). APT models crops as either part of a three year rotation, or (primarily for permanent grassland) as continuous cropping. The primary benefit of this approach is that it allows the simulations to include the effects of crop management in previous years. APT runs encompassed a 20-year period (1991–2010) and annual average agricultural fine sediment losses over this period per WFD water body were calculated for inclusion in the approach detailed by this paper.

To be able to produce estimates of agricultural fine sediment entering the river network at any given point, it was necessary to derive catchment boundaries for every 500 m point on the river network. Catchment areas were derived by burning the Centre of Ecology & Hydrology (CEH) 1:50,000 digitised river network into the 50 m SAR Digital Elevation Model (DEM) and building a reconditioned DEM using the AGREE (Hellweger, 1997) reconditioning tool in *ArcHydro* (Maidment, 2002). Because of inconsistencies between the DEM and river network, a substantial number of points failed to generate valid catchment areas. The number of failures was reduced by running them through the same delineation process but using a flow direction grid built from a non-stream burnt DEM.

An estimate of the amount of agricultural fine sediment delivered to individual 500 m reaches was derived using a combination of the local agricultural sediment input value for that 500 m reach plus an assessment of sediment transported into the reach but orig-

Fig. 3. Distribution of 9406 fisheries electro-fishing survey sites from the 2000–2005 EA monitoring programme.

inating from the upstream catchment. The Agricultural Sediment Load (ASL) metric was calculated using the following equation:

$$\mathrm{ASL} = (\mathrm{LS}/\mathrm{LRN} \times 500) + (\mathrm{CS}/\mathrm{CRN} \times 500)$$

where LS represents the agricultural fine sediment load entering a given 500 m reach; LRN is the length of river network in metres within the catchment area feeding into the 500 m reach; CS represents the amount of fine sediment delivered to the catchment upstream of the 500 m reach, and CRN is the length of river network in the catchment upstream in metres (Fig. 2).

The ASL thus represents the sum of two predicted estimates of fine sediment load delivered to rivers, standardised to a 500 m section. The first value considers local sources of fine sediment feeding into the reach of interest and will account both for run-off and for transported sediment from any tributary that may enter the river section in question. The second value deals with sediment transported from upstream. It represents the quantity of sediment delivered to an average 500 m reach in the upstream catchment. Both quantities act as an estimate of the amount of agricultural fine sediment delivered each year to individual 500 m reaches.

2.3. Agricultural sediment risk

Agricultural Sediment Risk categories (ASR) were defined using a matrix combining the FSA and ASL indices to represent increasing risk of agricultural fine sediment accumulation in-channel and their potential impacts on biota. The ASL index was split into 5 categories based on the distribution quintiles derived from the range of sediment inputs generated using APT. The matrix was drawn using the combined expertise of all authors. On this basis, a map of ASR was produced for the entire river network.

2.4. Link to biota

The relevance of the ASR map to aquatic biota was assessed against Environment Agency (EA) single-run fish density estimates

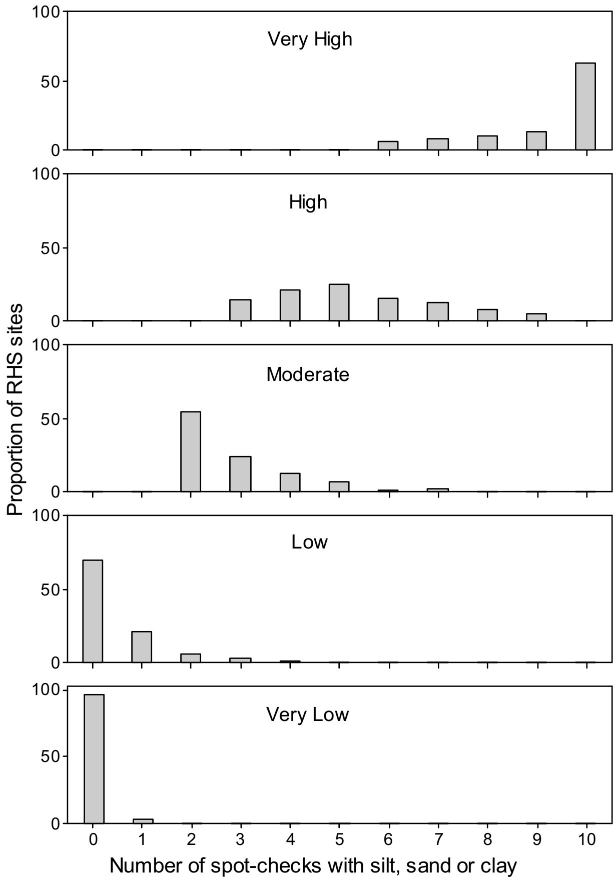

Fig. 4. Proportion of RHS sites within five FSA risk categories with sand, silt or clay as dominant channel substrates across 10 transects.

from 9406 electro-fishing surveys between 2000 and 2005 alongside prediction of occurrence at reference condition from the FCS (Fig. 3). FCS predictions were used to select sites with high habitat suitability (i.e. sites with a predicted likelihood at reference condition greater than 60%). Fish presence/absence was modelled against ASR using logistic regression. ASR was treated as a factor and each level was tested against the lowest available control level, generally ASR level I or 2. Overall factor significance was tested using chi-square statistics and individual factor levels were tested against the control using Z-statistics for associated odds-ratios. Odds-ratios provided the direction of change with an odds-ratio greater than one signifying a positive impact on fish occurrence and an odds-ratio less than one a negative impact. Odds-ratios not significantly different from one indicated no observable impact of ASR on fish.

3. Results

3.1. Fine sediment accumulation index

Partitioning of the CSI yielded 5 categories with increasing occurrence of fine sediment (Fig. 4). More than 60% of sites in the 'very high' FSA class (CSI < −1.56) had fine sediment dominant at 10 transects; 80% of sites with 'high' FSA (−1.56 < CSI < −1.02) were dominated by fines at 5 or more transects. The 'moderate' FSA class (−1.02 < CSI < −0.8) contained a majority of sites (80%) with 3 or 4 transects with fine sediment dominant whereas 'low' FSA (−0.8 < CSI < 0.29) sites had between 0 and 2 transects with fines dominant. The 'very low' category (CSI > 0.29) represented sites with no or little fine sediment.

3.2. Agricultural sediment load

ASL estimates were produced for most 500 m sections following DEM processing. The final number of invalid catchment delineations for individual sections was 55,224 out of a total of 342,586 (16.1%). Most invalid catchments were located in hydrometric areas with missing data, and in low gradient areas where low relief associated with complex grid like river channels made catchment delineation unreliable.

The FSA map (Fig. 5) showed a split between upland and lowland areas with high levels of fine sediment observed in East Anglia, Lincolnshire, Kent, Sussex and also large cities such as Manchester, Liverpool, Birmingham and London whereas the uplands in Wales, Cornwall and the Lake District showed low levels of agricultural fine sediment accumulation.

The map of ASL (Fig. 6) shows high sediment supply from agricultural sources in Norfolk, Suffolk and parts of Lincolnshire where agricultural field drains are present. In contrast to the previous map, the uplands of Wales, the south-west and the north-west display high levels of sediment supply reflecting higher levels of soil erosion and run-off from steeper slopes driven by higher rainfall totals, compared to those received in eastern areas of England.

3.3. Risk based matrix

The ASR matrix (Table 1) was designed in a symmetrical way to give equal importance to the ASL and FSA indices in determining integrated risk. The 'very high' and 'high' ASR categories combine high levels of fine sediment accumulation in river channels with high supply from agricultural land use. Sites belonging to those categories are likely to feature large amounts of accumulated fine sediment from agricultural origins. The "low" ASR categories represent sites with little fine sediment accumulation or sites with fine sediment dominant but with low contributions from agricultural land use. High levels of ASR are predicted for East Anglia, Lincolnshire and Kent in the east of England, as well as Merseyside and Manchester in the northwest of England area and around some big cities with the exception of London (Fig. 7).

3.4. Correlation between agricultural sediment risk and fisheries data

Out of 23 fish species used in conjunction with the ASR assessments, seven did not have enough data to enable analysis (Table 2). Overall fish species prevalence varied from 1% (carp) to 70% (trout) and the number of sites with high habitat suitability at reference condition ranged from 0 to 6227. The remaining 16 fish species could be split into 3 groups according to the direction and strength of correlative relationships.

The first group of eight species shared a sensitivity to agricultural fine sediment. It included salmonids, eels and some cyprinids (bleak, gudgeon, pike and bullhead). These species were found to have a negative relationship with ASR. Trout and salmon displayed the strongest relationships with very low odds-ratios at nearly all levels of ASR. A gradual increase in impact typified by decreasing odds-ratio values with increasing ASR was discernible for salmon and trout. Trout had the strongest response to ASR with low odds-ratios at ASR 2 and 3. Odds-ratios for salmon were somewhat higher and significantly dropped at ASR categories 4 and 5.

ASR also had significant or near significant overall impact on bleak, gudgeon, bullhead and eel. Pairwise comparisons showed significant impacts for high or very high levels of ASR. Results for Pike were altogether less clear. Although ASR had an overall high level of significance, pairwise comparisons yielded contradictory results, with ASR category 4 being significantly different from ASR category 1, but no difference could be observed between ASR cat-

Table 1
Agricultural Sediment Risk matrix combining FSA and ASL categories. The boundaries for ASL categories are shown in tonnes per year.

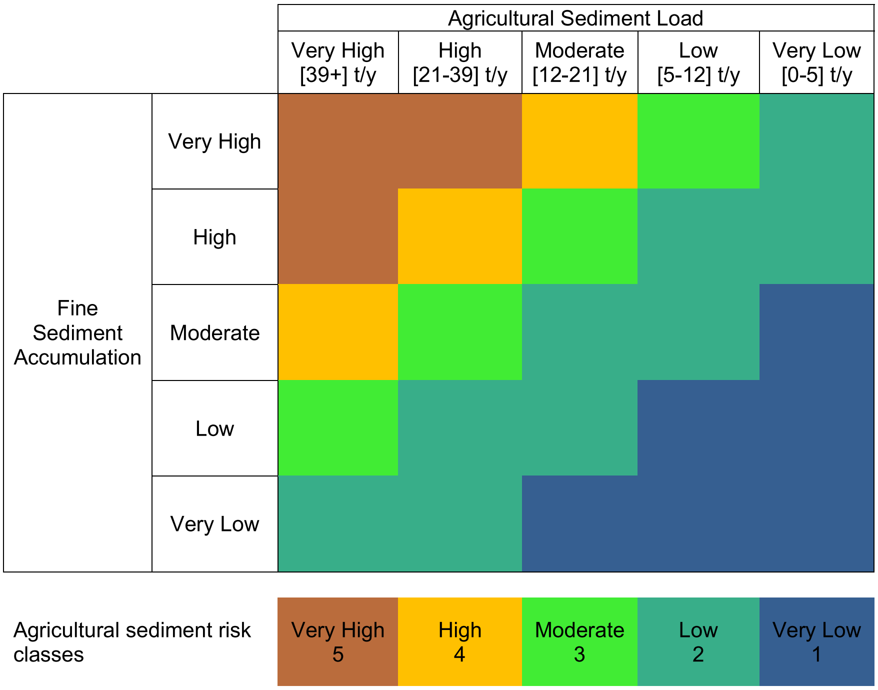

Table 2
Test of 23 fish species occurrence against ASR for sites with high habitat suitability at reference condition using logistic regression. P_v = Species prevalence; N_{60} = number of sites with probability of occurrence at reference condition less than 60%; NS = not significant; NED = not enough data; significance levels symbols: * p < 0.1; ** p < 0.05; *** p < 0.01.

Species	P_v	N_{60}	ASR factor significance	Pairwise comparisons to control. Odds-ratio and significance level			
				ASR2	ASR3	ASR4	ASR5
Trout (Salmo trutta)	70%	6627	χ^2 = 435***	0.66***	0.27***	0.09***	0.10***
Salmon (Salmo salar)	32%	3557	χ^2 = 33.3***	0.81**	0.87	0.19***	0.07***
Bleak (Alburnus alburnus)	6%	119	χ^2 = 18.2***	control	0.53	0.08***	0.10***
Gudgeon (Gobio gobio)	23%	1298	χ^2 = 16.7***	1.35	1.03	1.21	0.62**
Eel (Anguilla Anguilla)	39%	2589	χ^2 = 9.1*	0.85	0.89	1.22	0.67**
Bullhead (Cottus gobio)	54%	5137	χ^2 = 7.9*	1.1	1	0.92	0.77*
Pike (Esox Lucius)	23%	1360	χ^2 = 18.5***	1.24	1.44	0.58**	1.01
Grayling (Thymallus thymallus)	7%	101	χ^2 = 1.1	0.75	0.53	N/A	N/A
Roach (Rutilus rutilus)	31%	2130	χ^2 = 18.8***	1.87***	1.32	1.40***	1.06**
Perch (Perca fluviatilis)	25%	806	χ^2 = 14***	2.25***	1.74**	1.28	1.07
Stone Loach (Barbatula barbatula)	39%	2462	χ^2 = 8.3*	1.33***	1.29**	1.17	1.18
Chub (Leuciscus cephalus)	28%	2109	χ^2 = 11.3**	1.09	1.44**	1.19	1.67***
Minnow (Phoxinus phoxinus)	35%	2408	χ^2 = 9.1*	0.95	1.1	1.65***	0.95
Stickleback (Gasterosteus aculeatus)	13%	404	χ^2 = 6.2	0.89	1.29	2.05	0.99
Spined Loach (Cobitis taenia)	1%	42	χ^2 = 2.7	control	4.5	1	1.6
Dace (Leuciscus leuciscus)	24%	1320	χ^2 = 44.6***	1.61**	0.64**	2.06**	0.79
Bream (Abramis brama)	7%	28	NED				
Barble (Barbus barbus)	4%	3	NED				
Ruffe (Gymnocephalus cernuus)	4%	0	NED				
Lamprey (Lampetra planeri)	12%	0	NED				
Rudd (Scardinius erythrophthalmus)	4%	0	NED				
Carp (Cyprinus carpio)	1%	0	NED				
Tench (Tinca tinca)	5%	0	NED				

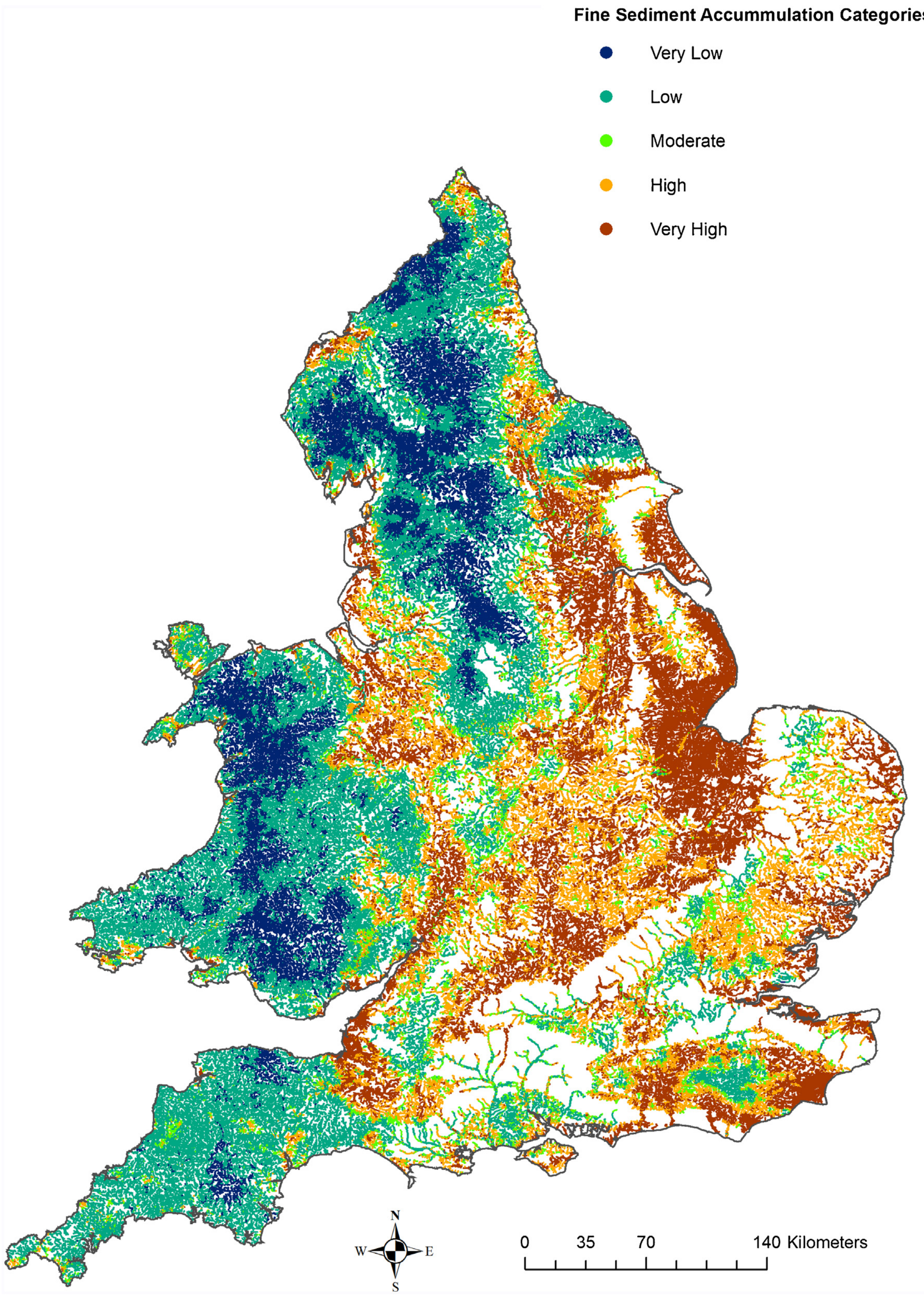

Fig. 5. FSA category distribution across England and Wales.

egory 5 and the control. Grayling had too small a sample size to enable meaningful analysis and comparison although the odds-ratios suggested a potential negative impact of agricultural fine sediment on species occurrence.

The second group of seven fish species displayed significant positive relationships to ASR. Increasing risk was thereby associated with increasing likelihood of finding the species. Roach, perch and stone loach displayed the highest levels of association with FSA. The odds-ratios, however, decreased with increasing ASR, which sug-

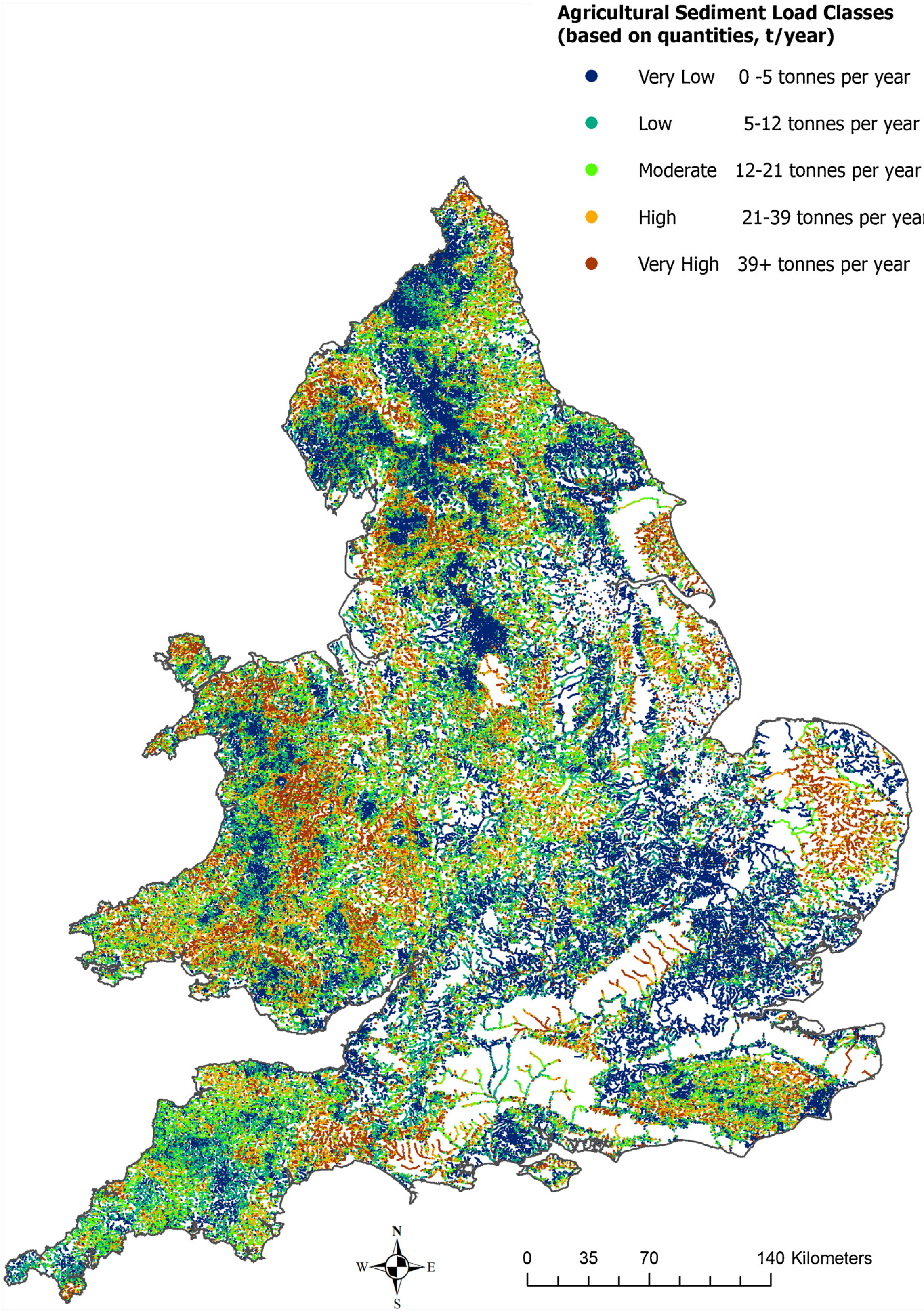

Fig. 6. ASL categories across England and Wales.

gests that agricultural sediment may benefit these particular fish species at low levels but its impacts may change as ASR increases. Chub and minnows displayed inconsistent patterns across the scale despite reaching overall significance. These results suggest that ASR may not directly benefit these species but may have an indirect effect through its impact on competing species. Sticklebacks and Spined Loach both failed to reach significance for all tests despite showing overall positive relationships to ASR.

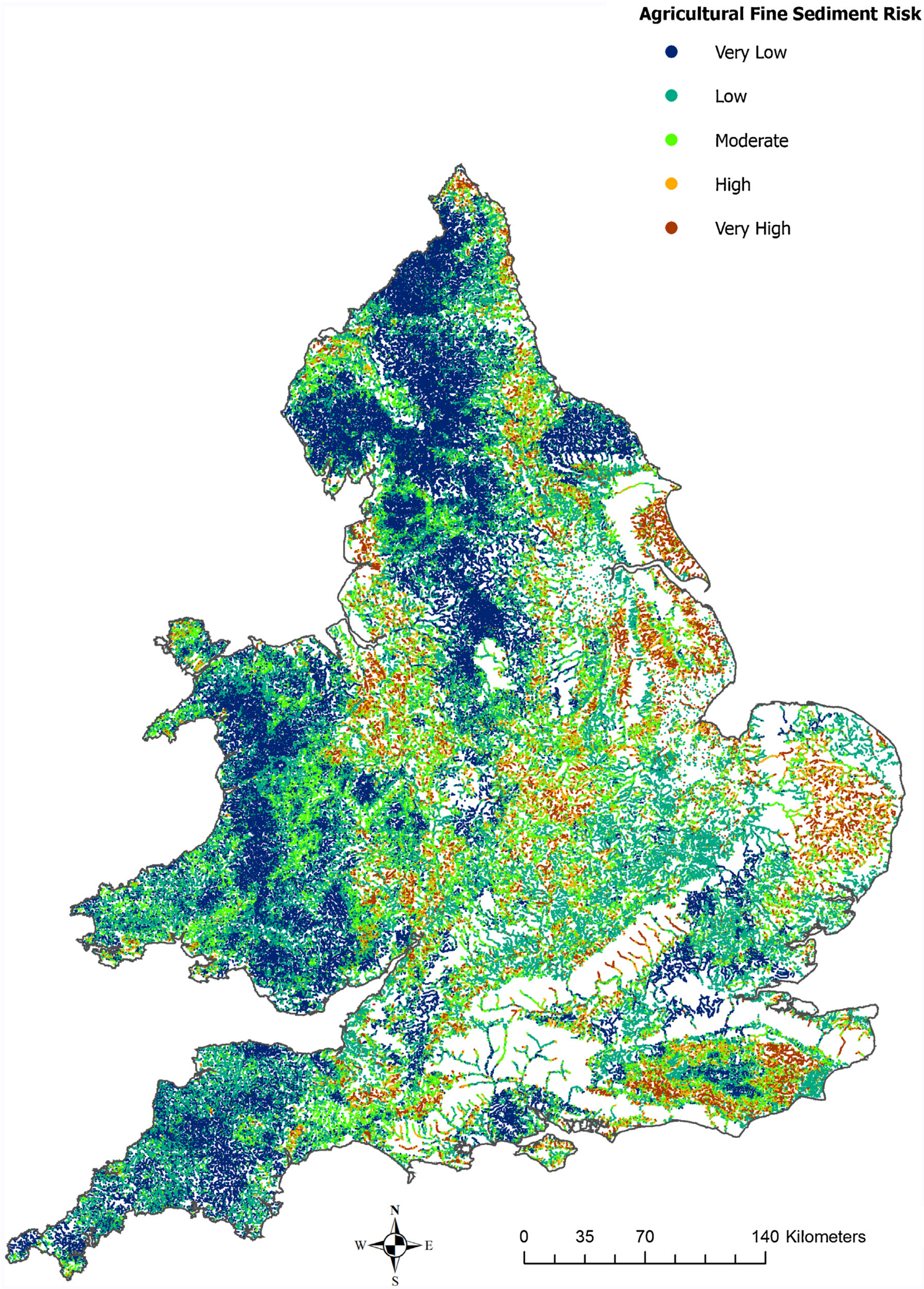

Fig. 7. ASR categories across England and Wales.

The last group contained only one species (Dace) and was characterised by no clear pattern of relationship between ASR and species occurrence or density. Individual pairwise differences yielded conflicting results with ASR having a positive impact on Dace occurrence at both low and high levels of ASR and a negative impact at moderate risk.

4. Discussion

4.1. Fine sediment risk mapping

The CSI provided a useful means of mapping fine sediment accumulation across the entire river network of England and Wales by concentrating on its finer fractions (i.e. sand, silt and clay) and creating an index representing its relative occurrence. The resulting FSA categories were good indicators of fine sediment accumulation potential with, at one end of the scale, sites that tend to retain fine sediment across most of their length, and at the other end, sites that are free of fine sediment accumulation.

The FSA gives an indication of substrate coverage but does not reflect the actual quantity of accumulated fine sediment over the reach. Future quality checks could be made to determine the strength of the relationship between the estimates of FSA and available fine sediment storage data (Collins and Walling, 2007a,b; Walling and Amos, 1999). Such comparisons would be assisted by the fact the modelled sediment pressure layer represents typical conditions over a twenty year period (1991–2010) as opposed to a specific modelled year.

The sediment pressure modelling was generated using the latest policy-support national scale framework for fine sediment loss from agriculture. It is, however important to note that some difficulties were experienced in deriving meaningful catchment areas for 16.1% of 500 m reaches across the river network of England and Wales. These reaches were consistently in lowland areas where sedimentation impacts on biota are likely to be detectable. The pressure modelling generates predictions of agricultural fine sediment loads delivered to the river channel network but does not include any subsequent routing and storage or remobilisation. Ongoing work is developing improved representation of current practice by farmers (e.g. on field drain maintenance, implementation of sediment control measures) and this new understanding will need to be combined with the framework reported here to update national scale understanding of sediment pressure from agriculture. The national pressure layer used herein includes crop areas and livestock numbers but not on-farm implementation of mitigation measures for erosion and sediment delivery control such as those supported by agri-environment schemes.

The ASL calculation provided an estimate of the amount of fine sediment delivered to a reach from both local inputs and from upstream sources. As there was no absolute definition of what constitutes a 'high' or 'low' agricultural sediment load, the use of quintiles based on the overall predicted range of agricultural fine sediment delivery to rivers enabled an unbiased classification of ASL and introduced an element of proportionality. Future work could make use of estimates of sediment delivery under lower-intensity pre-World War II agriculture derived from palaeo-records such as those recently proposed (Collins et al., 2012a; Foster et al., 2011) to assess the impact of current agricultural land use and practise on ASR.

The risk matrix attempted to combine indices of agricultural fine sediment load and accumulation so as to reflect the likelihood that mobilised fine sediment delivered to river channels across England and Wales is from agricultural origin and is likely to be stored in the channel network.

The risk maps produced for England and Wales show how agricultural fine sediment delivery to the river network can be high (Fig. 6) in the upland areas of England and Wales, but that the overall risk is reduced by the transfer of this material through the river network (Fig. 7). Areas at high-moderate risk from agriculturally derived fine sediment are shown to be largely in low gradient rivers where the combination of high delivery from the farmed landscape coincides with high accumulation in the river channel. It is important to note that these areas may naturally have channels

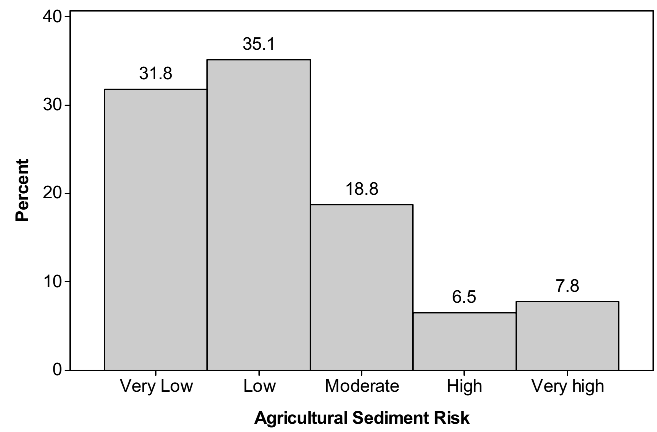

Fig. 8. Proportion of the 1/50,000 river network in England and Wales falling within each ASR category.

dominated by fine sediment because of local hydraulics and sediment supply from upstream sources (Church, 2002). As a result, future work will aim to assess how current land use and farming practise have potentially increased the accumulation of fines in river substrates relative to natural background levels. Such additional work could also be expanded further to project the potential impacts of both climate and land use change forecasts. Reducing inputs from agriculture will not significantly affect sediment accumulation and local biota in areas where agriculture is not the dominant source of fines delivered to river channels. Although it is possible to overestimate the importance of sediment from agricultural sources using the ASR map, statistics derived from the ASL map showed that agriculture appears to be the main source of fine sediment for the majority of rivers. In England and Wales, 58% of the river network sediment sources were overwhelmingly agricultural in nature (80–100% agricultural) and an additional 19% had high levels of agricultural inputs (from 60 to 80% agricultural). Previous work by Collins and Anthony (2008), Collins et al. (2009a,b) and Zhang et al. (2014) consistently identified agriculture as the dominant source of fine sediment delivered to the river channel network in the majority of water bodies across England and Wales. When considering the whole river network, however, the proportion of river reaches falling into the high ASR categories are relatively small with only 6.5% and 7.8% of 500 m sections having 'high' or 'very high' risk from agricultural sediment input (Fig. 8). This compares with a majority of river reaches falling into the 'very low (31.8%) or "low" (35.1%) categories. In spite of the high proportion of fine sediment that originates from agricultural origins, we found a minority of river reaches with high in-channel accumulation risks associated. Such information provides an additional data layer for supporting the spatial targeting of sediment remediation measures.

4.2. Link to fish species

Salmon and trout were the species most correlated with ASR. Sediment infiltration within gravels used for spawning is known to severely reduce salmon and trout egg survival (Sear, 2010). Salmon and trout are also sensitive to pollution by phosphates, pesticides and herbicides potentially carried on the surface of clay-sized particles (Kemp et al., 2011).

Bleak, gudgeon and eel showed responses to high sediment risk levels although with less of a marked trend than for salmonids. In their literature review, Kemp et al. (2011) could not find any reference to potential threats to egg survival of Bleak and Gudgeon resulting from fine sediment although eggs are deposited on gravel, which makes them potentially susceptible to fine sediment accumulation and smothering. The negative relationship of eels to ASR

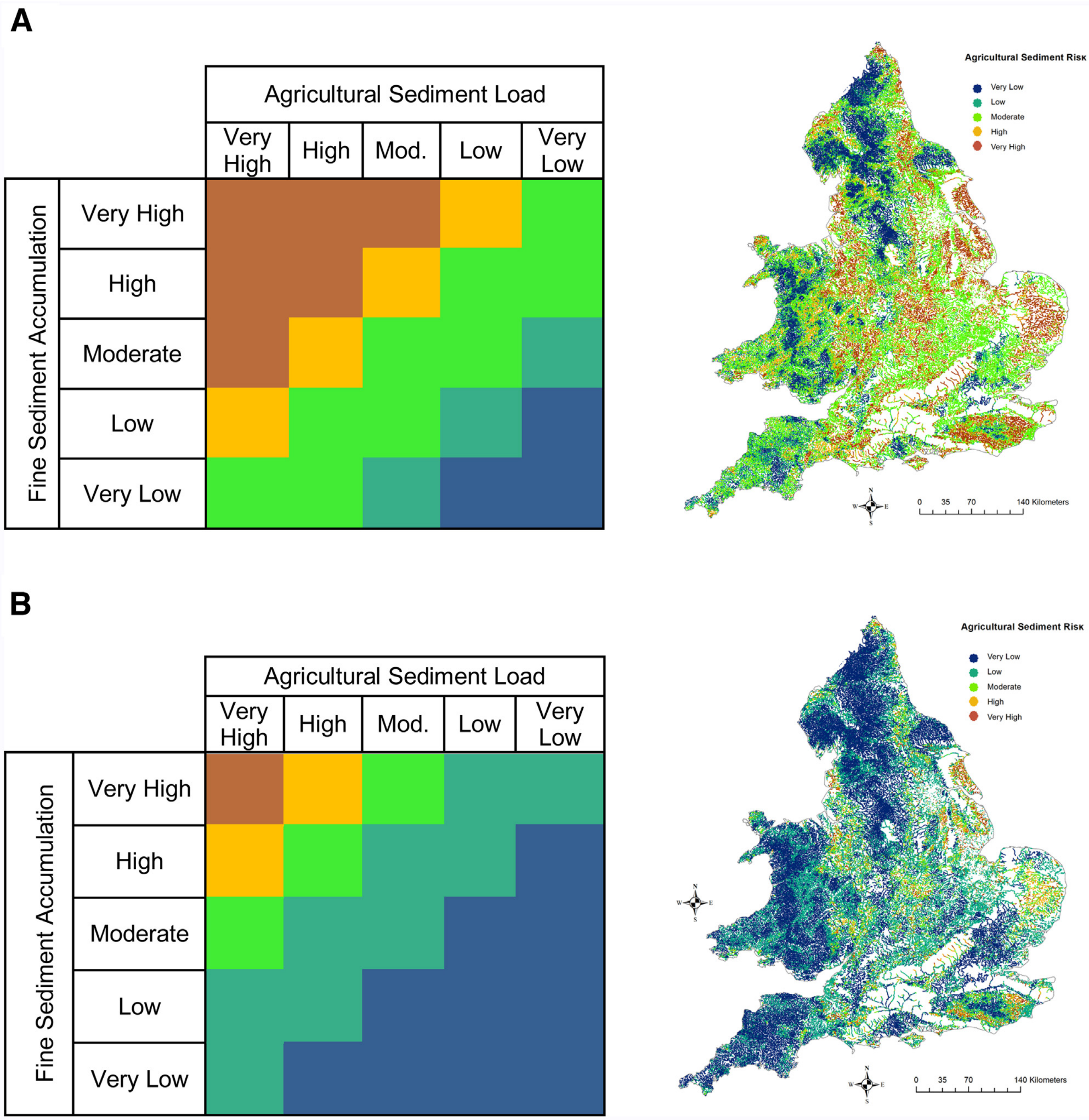

Fig. 9. A- ASR matrix and map illustrating a precautionary approach towards sediment management. B- ASR matrix and map illustrating a more stringent approach towards defining risk associated with elevated agricultural fine sediment.

is more puzzling as eels are known to prefer muddy habitats and do not incubate in freshwater (Maitland and Campbell, 1992). ASR may have an impact on their foraging ability and invertebrate food but this requires further investigation and analysis.

Grayling eggs have been reported as being sensitive to fine sediment. Although odds-ratios suggested a potential negative impact, they failed to reach significance as sample sizes were low.

Bullhead was loosely correlated with increased ASR. Although bullhead requires coarse substrate to reproduce and fairly clean water, eggs are laid on the underside of a stone excavated by the male. Fine sediment impacts may therefore be mitigated by nest building choice and spawning strategy. High density of sediment may impact on fish eggs by reducing spawning site availability and affecting egg survival by adhering to the surface.

Pike rely on vegetation to spawn and eggs are therefore unlikely to be affected by fine sediment accumulation unless sediment in suspension carries pollutants or sticks to the eggs. The level of significance of ASR, despite being high for overall effect was inconsistent between factor levels and may be the consequence of inherent uncertainties in the data used by this study.

ASR was positively correlated to a group of seven species who seemed to benefit from increasing agricultural sediment accumulation. The reasons behind these relationships are not clear although there may be a link to indirect effects on competitors and predators. The case of roach and perch was interesting as it showed a notable decrease in sediment impact with increasing ASR which suggests an indirect effect. Young roach rely on the presence of mud that they ingest to feed (Maitland and Campbell, 1992). They are therefore more likely to be found in places where fine sediment occurs. But roach are also typical prey for pike, trout and perch (as well as riverine birds). A relatively small increase in sediment input may therefore impact on trout and reduce predation through increased turbidity. As sediment load increases, local habitats and vegetation get gradually smothered and roach may suffer from an absence of cover to avoid other predators such as pike and perch, and a shortage of more nutritious food such as molluscs and invertebrates.

Perch do not rely on clean substrate for spawning. Their eggs are laid in shallow water around plants or other submerged objects. Perch feed on a wide range of prey, from invertebrates, molluscs to other fish species. Like pike, they can effectively detect and capture prey in the absence of visibility. They are therefore unlikely to be affected by elevated turbidity. Perch diets and feeding habits are similar to that of trout. The presence of fine sediment impacting trout populations may therefore give perch an opportunity to colonise adjacent habitats and survive in higher numbers.

Chub generally prefer diverse habitat with coarse and fine substrate. They spawn on vegetation, stones and gravel with a preference for weed. No mention of adverse impacts of fine sediment on chub eggs was found by Kemp et al. (2011) but it was suggested that their spawning habits may make them vulnerable. Evidence from the analysis here does not seem to support this. However, the positive relationships observed could also be the result of the adverse impact on competing species or random factors (e.g. variability/uncertainty introduced by survey techniques).

Minnows and Stone Loach displayed slight correlation with ASR. As for chub, the results are slightly counterintuitive as they lay their eggs over gravel and should be more susceptible to elevated fine sediment inputs. Minnows are also sensitive to low oxygen levels and pollution. They share a similar diet to trout and they are thought to potentially compete when both species are present (Maitland and Campbell, 1992). Considering their sensitivity to pollution, it is not clear why the analysis presented herein suggests that high levels of ASR benefit minnows.

Sticklebacks showed no significant preference for fine sediment although the males build their nests with particles of silt and sand. As the male sticklebacks fan their eggs during incubation, they are less likely to be affected by changes in sedimentation.

The case of Dace is confusing as it shows greater probability of occurrence at low and high levels of ASR and lower probabilities at moderate levels. Dace like to live in fast flowing rivers and rely on clean gravel for spawning. Silt has been shown to impact on egg survival (Kemp et al., 2011); therefore it is not clear why dace should benefit from agricultural fine sediment inputs.

4.3. Management implications

Despite its limitations outlined briefly above, the approach presented herein shows its potential usefulness as a management tool. Sediment accumulation can effectively be predicted using RHS data and sites potentially at risk from agricultural sources can be identified. Used in combination with WFD classification tools, it could help practitioners identify sites at risk of failing GES because of agricultural sediment inputs. The ASL and FSA maps could also help identify whether high sediment accumulation is due to high sediment delivery to river channels or to the retentive capacity of streams receiving the fine sediment delivered from agricultural sources. This could, in turn, be used to help inform management and ensure that actions taken on the ground reflect an understanding of the problem at hand.

Fine sediment accumulation results from both natural and human modifications to the land surface and river network. Thus an important element of any risk based approach is to account for natural accumulation in order to reveal excess accumulation resulting from human modifications. Future work will attempt to model natural sediment accumulation and delivery to streams using data from unimpacted RHS sites and national scale estimates of modern background delivery to rivers (Foster et al., 2011).

The ASR risk matrix is somewhat arbitrary and represents the authors' consensus on risks of agricultural fine sediment accumulation. From a management viewpoint, different matrices may be derived depending on the level of caution that environmental managers and regulators wish to exert when dealing with the specific issue of elevated agricultural fine sediment inputs to rivers and streams. As an example, two additional matrices and corresponding risk maps were produced to reflect a more precautionary approach towards managing ASR to biota (Fig. 9A), and one that is more stringent in its definition of risk (Fig. 9B). The resulting maps (Fig. 9A & B) show observable changes in the distribution of ASR with high risk sites being far more prevalent for the precautionary approach (26% of river reaches at 'high' or 'very high risk' compared to 6% for the more conservative approach).

For the purpose of this study, the analyses concentrated on fine sediment from agricultural origin but the approach could equally be applied to other sources of sediment such as sewage treatment works, urban areas and bank erosion using the national scale modelled layers reported in Collins et al. (2009a,b) and Zhang et al. (2014).

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Effects of Suspended Sediments on Aquatic Ecosystems

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Effects of Suspended Sediments on Aquatic Ecosystems

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Abstract.—Resource managers need to predict effects of pollution episodes on aquatic biota, and suspended sediment is an important variable in considerations of freshwater quality. Despite considerable research, there is little agreement on environmental effects of suspended sediment as a function of concentration and duration of exposure. More than 70 papers on the effects of inorganic suspended sediments on freshwater and marine fish and other organisms were reviewed to compile a data base on such effects. Regression analysis indicates that concentration alone is a relatively poor indicator of suspended sediment effects ($r^2 = 0.14$, NS). The product of sediment concentration (mg/L) and duration of exposure (h) is a better indicator of effects ($r^2 = 0.64$, $P < 0.01$). An index of pollution intensity (stress index) is calculated by taking the natural logarithm of the product of concentration and duration. The stress index provides a convenient tool for predicting effects for a pollution episode of known intensity. Aquatic biota respond to both the concentration of suspended sediments and duration of exposure, much as they do for other environmental contaminants. Researchers should, therefore, not only report concentration of suspended sediment but also duration of exposure of aquatic biota to suspended sediments.

The effects of suspended sediments on fish and aquatic life have been studied intensively. The available information on suspended sediment effects has been collated and analyzed in numerous reviews of the literature (Cordone and Kelly 1961; Petticord 1980; Alabaster and Lloyd 1982). However, although these reviews are both detailed and synoptic, they have not established general principles characterizing environmental effects of suspended sediments.

In this paper, we review the available literature in an attempt to identify factors that contribute to effects of suspended sediments on fish and aquatic life. This information should provide researchers with guidance on which data ought to be collected to develop a verified model of the environmental effects of suspended sediment. Experience with environmental toxicants suggests that severity of effects is related not only to concentration of a substance, but also to duration of exposure. In addition, frequency of pollution episodes, ambient water quality, species and life history stage affected, and the presence of disease organisms and other environmental toxicants may all affect the toxicity of a substance. Much of the reported work on effects of inert suspended sediments fails to include information other than concentration and an organism's response. Apparently, many researchers in this field assume that effects are dependent only on concentration, or that the time frame (although not explicitly stated) is implied (e.g., the time required for eggs to develop into fry). We analyzed the information available to determine which model provides better predictive power, the implicit concentration–response model currently in use or a concentration–duration response model similar to those currently used to assess the effects of toxicants.

Data-Base Development

Our search for a relationship between the magnitude of suspended sediment pollution and severity of effect involved collation and analysis of relevant data scattered throughout the literature. Researchers have reported a diverse assortment of effects. For the purpose of this assessment, effects were grouped into one of three categories:

(1) *Lethal effects.*—Lethal effects kill individual fish, cause population reductions, or damage the capacity of the ecosystem to produce fish. This category also includes reductions in population size that are believed to be caused by sublethal or behavioral effects.
(2) *Sublethal effects.*—Sublethal effects injure the tissues or physiology of the organism, but are not severe enough to cause death.
(3) *Behavioral effects.*—Behavioral effects change

TABLE 1.—Ranking of effects of suspended sediments on fish and aquatic life.

Rank	Description of effect
14	>80 to 100% mortality
13	>60 to 80% mortality
12	>40 to 60% mortality, severe habitat degradation
11	>20 to 40% mortality
10	0 to 20% mortality
9	Reduction in growth rates
8	Physiological stress and histological changes
7	Moderate habitat degradation
6	Poor condition of organism
5	Impaired homing
4	Reduction in feeding rates
3	Avoidance response, abandonment of cover
2	Alarm reaction, avoidance reaction
1	Increased coughing rate

activity patterns or alter the kinds of activity usually associated with an organism in an unperturbed environment.

Subsequently, effects were ranked according to severity of the effect on fish and aquatic life, as outlined in Table 1.

Although many articles deal with inert sediments and fisheries, we included in this analysis only those containing information on concentration of sediment in the water, length of time the organism was exposed to that sediment, and the nature of the effect. Many potentially useful articles lacked one or more pieces of essential information and were therefore excluded. In a few instances, missing information was supplied by the author of the original article or from a second published source.

Estimates of concentration and duration, or both, were used in some instances, but only when there were sufficient additional details in the original publication, or elsewhere, to do so with reasonable certainty. Many publications that provided no explicit measure of time of exposure did include a sufficiently detailed account of the context and circumstances of the pollution episode to permit useful estimates of exposure duration. In some instances, when information on the concentration of sediment in the water was not reported, information from authoritative sources other than the original reference was used. Typically, these outside sources provided correlations that permitted the conversion of turbidity measurements into concentrations of suspended sediment. In other instances, authors provided additional information in the form of personal communications. The rationale for each estimate of time and concentration are contained in Newcombe (1986).

Effects on Salmonid Fishes

There is a substantial body of knowledge about effects of suspended sediments on salmonid fishes. Previously published reviews (Cordone and Kelly 1961; Sorensen et al. 1977; Langer 1980; Alabaster and Lloyd 1982) indicate that salmonid fisheries can be affected by inert sediment (1) acting directly on free-living fish, either by killing them or by reducing their growth rate or resistance to disease, or both; (2) interfering with the development of eggs and larvae; (3) modifying natural movements and migrations of fish; (4) reducing the abundance of food organisms available to the fish; and (5) reducing the efficiency of methods used for catching fish. Tables 2–4 summarize the literature pertaining to lethal, sublethal, and behavioral responses of salmonid fishes to suspended sediment.

Effects of Aquatic Invertebrates

Benthic invertebrates in streams can be affected by elevated levels of suspended sediment in several ways. First, many benthic invertebrates are grazers and depend on periphyton for food. Any change in suspended sediment concentration that adversely affects algal growth, biomass, or species composition can adversely affect secondary production. Other invertebrates are filter feeders. Increases in suspended sediment levels tend to clog feeding structures, reduce feeding efficiency, and therefore reduce growth rates or stress or kill these organisms (Hynes 1970). Second, invertebrates that inhabit exposed streambed substrates are subject to scouring, which can damage exposed respiratory organs or make the organism more susceptible to predation through dislodgment (Langer 1980). Table 5 is a compilation of information on effects of suspended sediment on aquatic invertebrates. These data suggest that aquatic invertebrates are at least as sensitive to high levels of suspended sediment as salmonid fishes, and perhaps more so.

Effects on Periphyton

Effects of suspended sediment on algae are likely primarily related to its effect on light penetration. However, high levels of suspended sediment in conjunction with high flow rates can scour algae off streambed substrates and thereby reduce periphyton biomass (Alabaster and Lloyd 1982). In addition, increases in nutrients or toxic compounds, or both, adsorbed on suspended sediments can alter growth rates and biomass of algae.

TABLE 2.—Summary of data (in situ observations) on exposures to suspended sediment that resulted in lethal responses in salmonid fishes. Within species groups, stress indices are arranged in increasing order. For exposure, C = concentration (mg/L) and D = duration (h).

Species[a]	Exposure C	Exposure D	Stress index ($\log_e \cdot [C \times D]$)	Effect	Rank of effect	Source
			Arctic grayling			
Arctic grayling	25	24	6.397	6% mortality of sac fry	10	Reynolds et al. (1988)
	23	48	7.007	14% mortality of sac fry	10	Reynolds et al. (1988)
	65	24	7.352	15% mortality of sac fry	10	Reynolds et al. (1988)
	22	72	7.368	15% mortality of sac fry	10	Reynolds et al. (1988)
	20	96	7.560	13% mortality of sac fry	10	Reynolds et al. (1988)
	143	48	8.834	26% mortality of sac fry	11	Reynolds et al. (1988)
	185	72	9.497	41% mortality of sac fry	12	Reynolds et al. (1988)
	230	96	10.002	47% mortality of sac fry	12	Reynolds et al. (1988)
	20,000	96	14.468	10% mortality of age-0 fish	10	McLeay et al. (1987)
	100,000	96	16.077	20% mortality of age-0 fish	10	McLeay et al. (1987)
			Salmons			
Chinook salmon	488	96	10.755	50% mortality of smolts (high T°C)	12	Stober et al. (1981)
Coho salmon	509	96	10.797	50% mortality of smolts (high T°C)	12	Stober et al. (1981)
Chinook and sockeye salmon	1,400[b]	36	10.827	10% mortality of juveniles	10	Newcomb and Flagg (1983)
Coho salmon	1,200	96	11.654	50% mortality of juveniles	12	Noggle (1978)
	1,217	96	11.668	50% mortality of presmolts (high T°C)	12	Stober et al. (1981)
Chinook and sockeye salmon	207,000[b]	1	12.240	100% mortality of juveniles	14	Newcomb and Flagg (1983)
	9,400	36	12.732	50% mortality of juveniles	12	Newcomb and Flagg (1983)
Chum salmon	97	3,912[b]	12.847	77% mortality of eggs and alevins	13	Langer (1980)
	111	3,912[b]	12.981	90% mortality of eggs and alevins	14	Langer (1980)
Chinook and sockeye salmon	82,000	6	13.106	60% mortality of juveniles	12	Newcomb and Flagg (1983)
Coho salmon	18,672	96	14.400	50% mortality of presmolts	12	Stober et al. (1981)
Chinook salmon	19,364	96	14.436	50% mortality of smolts	12	Stober et al. (1981)
Chum salmon	28,000	96	14.804	50% mortality of juveniles	12	Smith (1939)
Coho salmon	28,184	96	14.811	50% mortality of smolts	12	Stober et al. (1981)
	29,580	96	14.859	50% mortality of smolts	12	Stober et al. (1981)
	35,000[b]	96	15.027	50% mortality of juveniles	12	Noggle (1978)
Chinook and sockeye salmon	39,400	36	15.145	90% mortality of juveniles	14	Newcomb and Flagg (1983)
Chum salmon	55,000	96	15.479	50% mortality of juveniles	12	Smith (1939)
			Whitefish			
Whitefish	16,613	96[b]	14.282	50% mortality of juveniles	12	Lawrence and Scherer (1974)
			Trouts			
Rainbow trout	200[c]	24	8.476	5% mortality of fry	10	Herbert and Richards (1963)
	7	1,152	8.995	17% reduction in egg-to-fry survival	10	Slaney et al. (1977b)
	21	1,152	10.094	62% reduction in egg-to-fry survival	13	Slaney et al. (1977b)
	200[c]	168	10.422	8% mortality of fry	10	Herbert and Richards (1963)
	90	456	10.622	5% mortality of subadults	10	Herbert and Merkens (1961)

TABLE 2.—Continued.

Species[a]	Exposure C	Exposure D	Stress index ($\log_e$ [$C \times D$])	Effect	Rank of effect	Source
	68	720[b]	10.799	25% reduction in population size	11	Peters (1967)
	37	1,440	10.883	46% reduction in egg-to-fry survival	12	Slaney et al. (1977b)
	47	1,152	10.889	100% mortality of incubating eggs	14	Slaney et al. (1977b)
	57	1,440	11.315	23% reduction in egg-to-fry survival	11	Slaney et al. (1977b)
	270[d]	456	11.721	10–35% mortality of subadults	11	Herbert and Merkens (1961)
	270[e]	456	11.721	80% mortality of subadults	13	Herbert and Merkens (1961)
	101	1,440	11.888	98% mortality of eggs (high metals and NH_3 levels)	14	Turnpenny and Williams (1980)
Brown trout	110	1,440	11.973	98% mortality of eggs	14	Scullion and Edwards (1980)
Rainbow and brown trout	300	720[b]	12.283	97% reduction in population size	14	Peters (1967)
Rainbow trout	1,000–2,500	144	12.437	100% mortality of eggs	14	Campbell (1954)
	157	1,728	12.511	100% mortality of eggs	14	Shaw and Maga (1943)
	810[d]	456	12.820	5–80% mortality of subadults	13	Herbert and Merkens (1961)
	810[e]	456	12.820	80–85% mortality of subadults	14	Herbert and Merkens (1961)
	200[c]	2,352	13.061	50% mortality of fry	12	Herbert and Richards (1963)
	1,000–2,500	480	13.641	57% mortality of fingerlings	12	Campbell (1954)
	4,250	588	14.731	50% mortality (life stage not specified)	12	Herbert and Wakeford (1962)
	160,000	24	15.161	100% mortality (life stage not specified)	14	D. W. Herbert, personal communication in Alabaster and Lloyd (1982)
	49,000	96	15.363	50% mortality of juveniles	12	Lawrence and Scherer (1974)
	1,000–6,000	1,440[b]	15.432	85% reduction in population size	14	Herbert and Merkens (1961)
Brown trout	1,040	8,670	16.024	85% reduction in population size	14	Herbert et al. (1961)
	5,838	8,670	17.750	85% reduction in population size	14	Herbert et al. (1961)

[a] Scientific names: Arctic grayling, *Thymallus arcticus*; chinook salmon, *Oncorhynchus tshawytscha*; coho salmon, *O. kisutch*; sockeye salmon, *O. nerka*; chum salmon, *O. keta*; whitefish, *Coregonus* sp.; rainbow trout, *Oncorhynchus mykiss*; brown trout, *Salmo trutta*.
[b] Estimated.
[c] Wood fiber.
[d] Kaolin.
[e] Diatomaceous earth.

Models of Suspended Sediment Effects

The literature on suspended sediment effects on fish and aquatic life is dominated by the tacit assumption that the implicit concentration–response model applies. This model suggests that if the concentration of suspended sediment is known, then the response of aquatic biota can be predicted. However, for environmental toxicants (copper and ammonia, for example), it is known that responses are dependent not only on concentration but also on duration of exposure. It is our hypothesis that the concentration–duration response model, commonly applied to contaminants, also applies to suspended sediment effects.

To test this hypothesis, the information collated from the scientific literature was ranked by severity of effect and plotted against suspended sediment concentration (Figure 1) and intensity (concentration times duration of exposure; Figure 2). The results indicated that the natural logarithm of the concentration of suspended sediment was

TABLE 3.—Summary of data on exposures to suspended sediment that resulted in sublethal responses in salmonid fishes. Within species groups, stress indices are in increasing order. For exposure, C = concentration (mg/L) and D = duration (h).

Species[a]	Exposure C	Exposure D	Stress index ($\log_e \cdot [C \times D]$)	Effect	Rank of effect	Source
				Arctic grayling		
Arctic grayling	100	1	4.605	Reduction in feeding rate	4	McLeay et al. (1984)
	100	1,008	11.521	6% reduction in growth rate	9	McLeay et al. (1984)
	300	1,008	12.620	Physiological stress	8	McLeay et al. (1987)
	300	1,008	12.620	10% reduction in growth rate	9	McLeay et al. (1987)
	1,000	1,008	13.823	33% reduction in growth rate	9	McLeay et al. (1987)
				Salmons		
Coho salmon	14	1	2.639	Reduction in feeding efficiency	4	Berg and Northcote (1985)
	100	1[b]	4.605	45% reduction in feeding rate	4	Noggle (1978)
	250	1[b]	5.521	90% reduction in feeding rate	4	Noggle (1978)
	300	1[b]	5.704	Feeding ceased	4	Noggle (1978)
	53.5	12	6.465	Physiological stress, changes in behavior	8	Berg (1983)
Chinook salmon	1.5–2.0[c]	1,440	7.832	Gill hyperplasia, poor condition of fry	8	Anderson, U.S. Fish and Wildlife Service, personal communication
	6[c]	1,440	9.064	Reduction in growth rate	9	MacKinlay et al. (1987)
	75	168[b]	9.441	Harm to quality of habitat	7	Slaney et al. (1977a)
	84[d]	336	10.248	Reduction in growth rate	9	Sigler et al. (1984)
	1,547	96	11.908	Histological damage to gills	8	Noggle (1978)
				Trouts		
Cutthroat trout	35	2	4.248	Feeding ceased, cover sought	4	Bachmann (1958)
Rainbow trout	500	9	8.412	Physiological ill effects	8	Redding and Schreck (1980)
	171	96	9.706	Histological damage	8	Goldes (1983)
Steelhead	84[d]	336	10.248	Reduction in growth rate	9	Sigler et al. (1984)
Rainbow trout	50[e]	960[b]	10.779	Reduction in growth rate	9	Herbert and Richards (1963)
	50[f]	960[b]	10.779	Reduction in growth rate	9	Herbert and Richards (1963)
Trout	270	312[b]	11.341	Histological damage to gills	8	Herbert and Merkens (1961)
Rainbow trout	50[c]	1,848	11.434	Reduction in growth rate	9	Sykora et al. (1972)
	5,000–300,000	168	13.641–17.736	Fish survived, but gill epithelium harmed	8	Slanina (1962)
Brook trout	12[c]	5,880	11.164	Reduction in growth rate, reduced condition	9	Sykora et al. (1972)
	100[c]	1,176[b]	11.675	Reduction in growth rate	9	Sykora et al. (1972)
	24[c]	5,280	11.736	Reduction in growth rate	9	Sykora et al. (1972)

[a] Scientific names: cutthroat trout, *Oncorhynchus clarki*; steelhead = anadromous rainbow trout; brook trout, *Salvelinus fontinalis*.
[b] Estimated.
[c] Lime-neutralized iron hydroxide.
[d] Fire clay.
[e] Coal dust.
[f] Wood fiber.

poorly correlated with the ranked response of aquatic biota (r^2 = 0.14, NS). Regression of the natural logarithm of suspended sediment intensity against ranked response was more strongly correlated ($r^2 = 0.64$, $P < 0.01$). This analysis suggests that suspended sediment effects on aquatic ecosystems can be better predicted with a concentration–duration response model developed from the available information.

Stress Index

Pollution episodes reported in the primary literature span a wide range of suspended sediment concentrations and exposure times. The range of the product of these two variables (concentration and duration of exposure) is even larger, spanning many orders of magnitude. To compress this range and provide numbers of manageable size, the natural logarithm of the product was taken as an index of severity, which we refer to as a stress index.

The considerable variability among data in the literature limits our ability to test the stress index for predicting precise responses of aquatic biota to exposures to suspended sediment. Variables in the data include, but are certainly not limited to, species, life history stage and physiological con-

TABLE 4.—Summary of data on exposures to suspended sediment that resulted in behavioral responses in salmonid fishes. Within species groups, stress indices are in increasing order. For exposure, C = concentration (mg/L) and D = duration (h).

Species	Exposure C	Exposure D	Stress index ($\log_e$ [$C \times D$])	Effect	Rank of effect	Source
				Arctic grayling		
Arctic grayling	100[a]	1	2.303	Avoidance response	3	Suchanek et al. (1984a), Suchanek et al. (1984b)
				Salmons		
Coho salmon	54	0.02	0.077	Alarm reaction	2	Berg (1983)
	88	0.02	0.565	Alarm reaction	2	Bisson and Bilby (1982)
	4.3[b]	1	1.447	Avoidance response	3	Updegraff and Sykora (1976)
	88	0.08	1.952	Avoidance response	3	Bisson and Bilby (1982)
	25	4	4.605	Sport fishing declines	4	Phillips (1970)
Salmon	8	24	5.257	Sport fishing declines	4	A. H. Townsend, unpublished, cited in Lloyd (1985)
Chinook salmon	650	1	6.477	Homing performance disrupted	5	Whitman et al. (1982)
Coho salmon	6,000[a]	0	8.700	Avoidance response	3	Noggle (1978)
				Whitefish		
Whitefish	0.7	1	−0.416	Overhead cover abandoned	3	Lawrence and Scherer (1974)
				Trouts		
Rainbow trout	100[a]	1	2.303	Avoidance response	3	Suchanek et al. (1984a), Suchanek et al. (1984b)
	100[c]	0.25	3.219	Coughing rate increased	1	Hughes (1975)
	250[d]	0.25	4.135	Coughing rate increased	1	Hughes (1975)
	66	1	4.190	Avoidance response	3	Lawrence and Scherer (1974)
Trout	8	24[a]	5.257	Sport fishing declines	4	A. H. Townsend, unpublished, cited in Lloyd (1985)
Rainbow trout	665	1[a]	6.500	Overhead cover abandoned	3	Lawrence and Scherer (1974)
Brook trout	4.5	168[a]	6.628	Overhead cover abandoned	3	Gradall and Swenson (1982)

[a] Estimated.
[b] Lime-neutralized iron hydroxide.
[c] Coal dust.
[d] Wood fiber.

dition of the organism affected, water temperature, dissolved oxygen concentration, particle size distribution and chemical composition of the sediment, and presence of other contaminants. Information on the degree to which these variables exacerbate or ameliorate the effects of suspended sediments is incomplete. Therefore, it is not yet possible to formulate generalizations about the nature or magnitude of their effects. Wide diversity and lack of precision in descriptions of organism responses represent another stumbling block in analyzing the available data, necessitating the response ranking used in this analysis. In many cases the effect reported was not necessarily the most serious effect on the organism at a given concentration and duration of exposure. Also, the duration of exposure reported does not necessarily represent the threshold for adverse effects. For example, reduction in the feeding rate of Arctic grayling was reported at 100 mg total suspended sediment/L for 1 h (McLeay et al. 1984). However, the same effect was reported for an 840-h exposure (McLeay et al. 1984). It is likely that the longer exposure would have more severe effects on the organism, but both effects were ranked the same.

Conclusions

Resource managers need information that relates the magnitude of pollution episodes to effects on aquatic ecosystems so the effects of various development schemes (e.g., coal and placer mining proposals) can be evaluated.

The implicit concentration–response model of suspended sediment effects currently in use provides little predictive ability. The dose concentration–duration response model (e.g., dose measured as pollution intensity) proposed in this paper provides better results. The stress index provides a convenient tool for assessing the severity of environmental effects when there is insufficient time or resources to complete a detailed environmental assessment. The predictive ability of this tool will

TABLE 5.—Summary of data on the effects of suspended sediment on aquatic invertebrates.

Taxon	Exposure C	Exposure D	Stress index ($\log_e$ [$C \times D$])	Effect	Rank of effect	Source
Zooplankton	24[a]	0.15	1.281	Reduced capacity to assimilate food	4	McCabe and O'Brien (1983)
Benthic invertebrates	8	2.5	2.996	Lethal: increased rate of drift	10	Rosenberg and Wiens (1978)
Macro invertebrates	53–92	24[a]	7.462	Lethal: reduction in population size	10	Gammon (1970)
Benthic invertebrates	1,700	2	8.132	Lethal: alteration in community structure and drift patterns	10	Fairchild et al. (1987)
Zoobenthos	10–15	720[a]	9.105	Lethal: reduction in standing crop	10	Rosenberg and Snow (1977)
Benthic invertebrates	8	1,440	9.352	Lethal: up to 50% reduction in standing crop	12	Rosenberg and Wiens (1978)
Cladocera	82–392	72[a]	9.745	Lethal: survival and reproduction harmed	12	Robertson (1957); from Alabaster and Lloyd (1982)
Benthic fauna	29	720[a]	9.947	Lethal: populations of Trichoptera, Ephemeroptera, Crustacea, and Mollusca, disappear	14	M.P. Vivier, personal communication in Alabaster and Lloyd (1982)
Benthic invertebrates	16	1,440	10.045	Lethal: reduction in standing crop	12	Slaney et al. (1977b)
Cladocera and Copepoda	300–500	72	10.268	Lethal: gills and gut clogged	14	Stephan (1953) cited in Alabaster and Lloyd (1982)
Benthic invertebrates	32	1,440	10.738	Lethal: reduction in standing crop	12	Slaney et al. (1977b)
Zoobenthos	>100	672[a]	11.115	Lethal: reduction in standing crop	12	Rosenberg and Snow (1977)
Benthic invertebrates	62	2,400	11.910	Lethal: 77% reduction in population size	13	Wagener and LaPerriere (1985)
	77	2,400	12.127	Lethal: 53% reduction in population size	12	Wagener and LaPerriere (1985)
Bottom fauna	261–390	720[a]	12.365	Lethal: reduction in population size	12	Tebo (1955)
Benthic invertebrates	390	720[a]	12.545	Lethal: reduction in population size	12	Tebo (1955)
	278	2,400	13.411	Lethal: 80% reduction in population size	13	Wagener and LaPerriere (1985)
Stream invertebrates	130[b]	8,760	13.945	Lethal: 40% reduction in species diversity	14	Nuttall and Bielby (1973)
Benthic invertebrates	743	2,400	14.394	Lethal: 85% reduction in population size	14	Wagener and LaPerriere (1985)
	5,108	2,400	16.322	Lethal: 94% reduction in population size	14	Wagener and LaPerriere (1985)
Stream invertebrates	25,000[b]	8,760	19.204	Lethal: reduction or elimination of populations	14	Nuttall and Bielby (1973)

[a] Estimated.
[b] China clay.

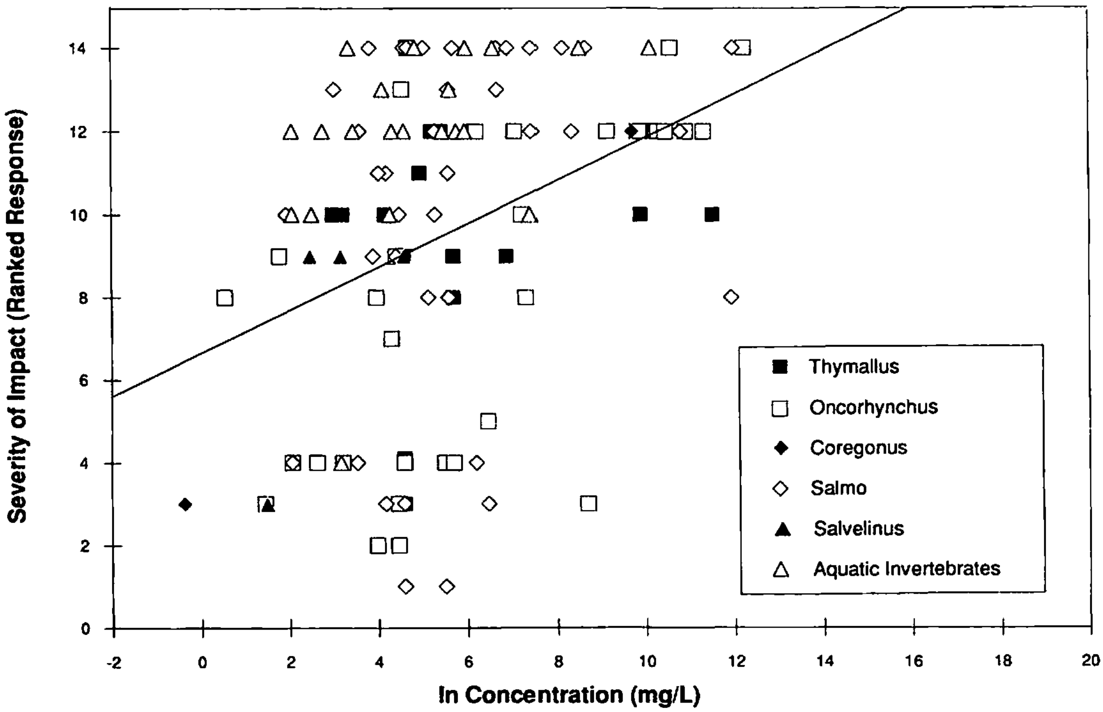

FIGURE 1.—Relationship between $\log_e$ (ln) of suspended sediment concentration and severity of effects on salmonid fishes and aquatic invertebrates. Severity of effect = 0.524 $\log_e$ concentration + 6.738; $r^2 = 0.141$, $N = 120$.

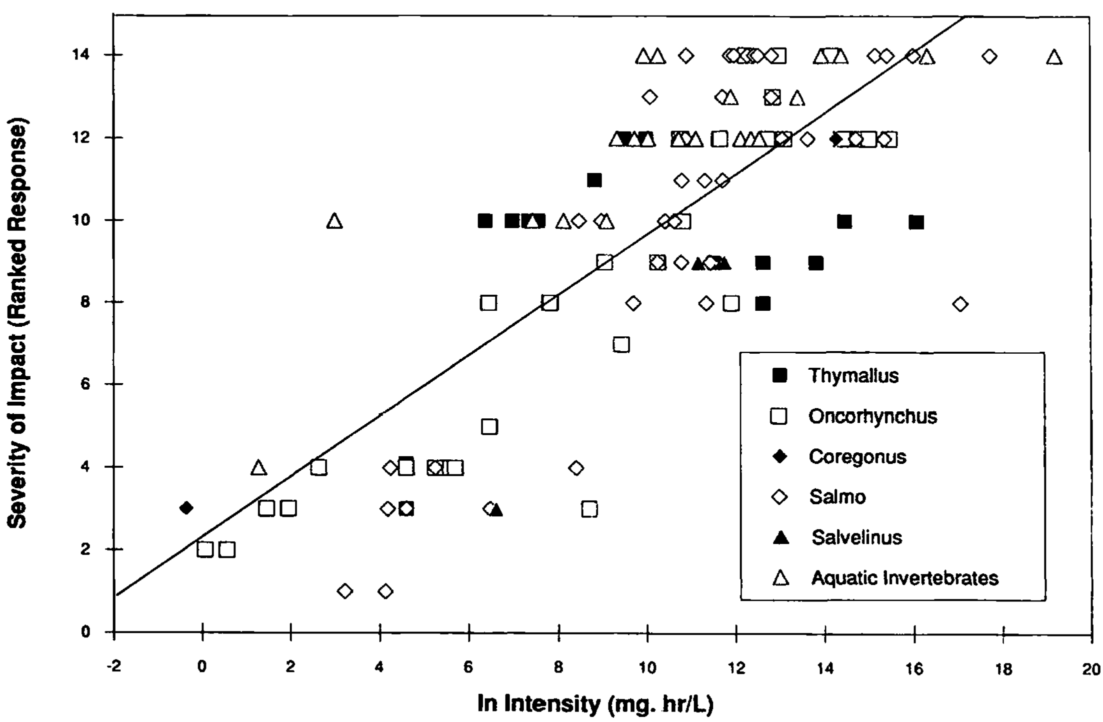

FIGURE 2.—Relationship between $\log_e$ (ln) of suspended sediment intensity and severity of effects on salmonid fishes and aquatic invertebrates. Severity of effect = 0.738 $\log_e$ intensity + 2.179; $r^2 = 0.638$, $N = 120$. Intensity is concentration (mg/L) times duration of exposure (h).

improve as more and better information on effects of suspended sediment on aquatic biota become available.

Future research in this field ought to be reported in terms of concentration of suspended sediment, duration of exposure, and response. In this way our ability to predict the environmental effects of pollution events will be improved. In addition, studies ought to concentrate on dissociating the effects of exposures to suspended sediment from the confounding effects of other variables.

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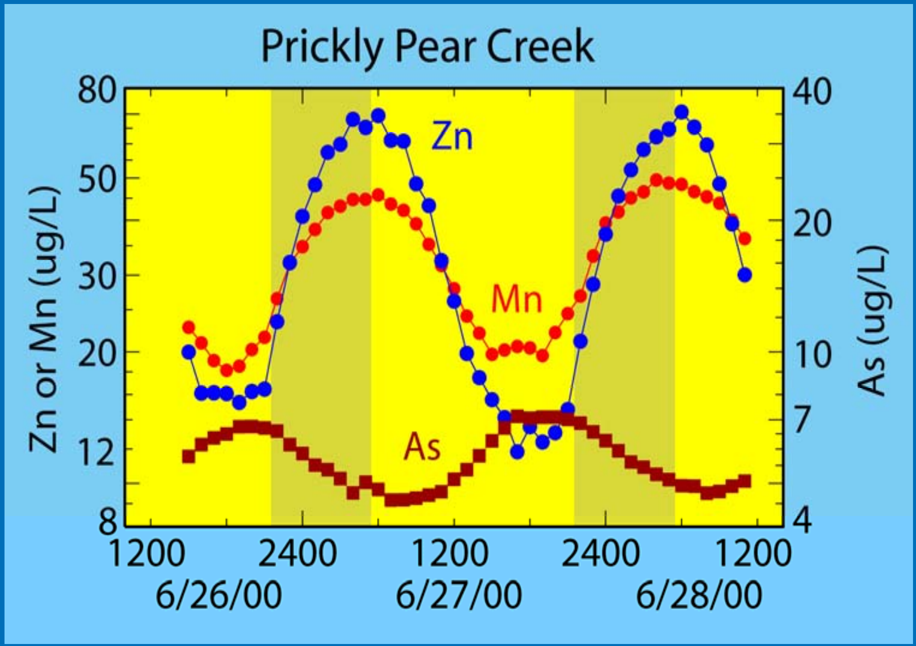
Fascinating Biogeochemistry: How Diel Cycling Complicates Surface-Water Monitoring



David Nimick
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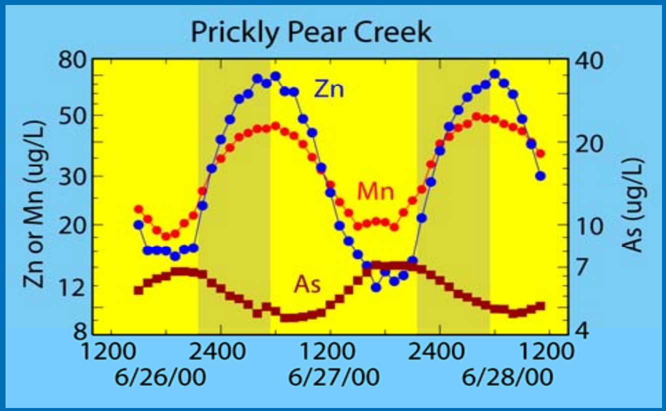
U.S. Department of the Interior
U.S. Geological Survey

Definitions

- **Periodicity of 24 hours:**



Diel *or* Diurnal Cycles

- **Activity:**

Diurnal

Nocturnal

USGS

Session I1: Effects of Diel Cycling on Stream Conditions Thursday 10:00-11:30 AM

- Pamela Reilly: Diel Cycles in Major and Trace Elements in Streams: Anthropogenic Effects on, and Additions to, Natural Cycles
- Richard Inouye: Diel Variation of Sediment Load in a 5th Order River in SE Idaho—Temporal Variation and Impacts on Load Estimates
- Briant Kimball: Diel Cycles Confound Synoptic Sampling in a Metal-Contaminated Stream
- Alba Argerich: Effects of Daily Fluctuations in Streamflow on Stream Metabolic Activity Calculations
- POSTER 13B. Pamela Reilly: Diel Biogeochemical Processes and Their Effects on Sample Design and Trend Analysis: A Study Looking at Diurnal Arsenic Cycling in a NJ Stream

Outline

- What is Diel Cycling?
- Diel Cycling Mechanisms
- Examples of Diel Cycles
  - Field parameters
  - Other common cycles
  - Nutrients
  - Metals
- Implications for Monitoring Water Quality
  - Examples (How you can get into trouble!)
  - Monitoring guidelines (How to stay out of trouble!)
  - Instrumentation

Madison River, Montana

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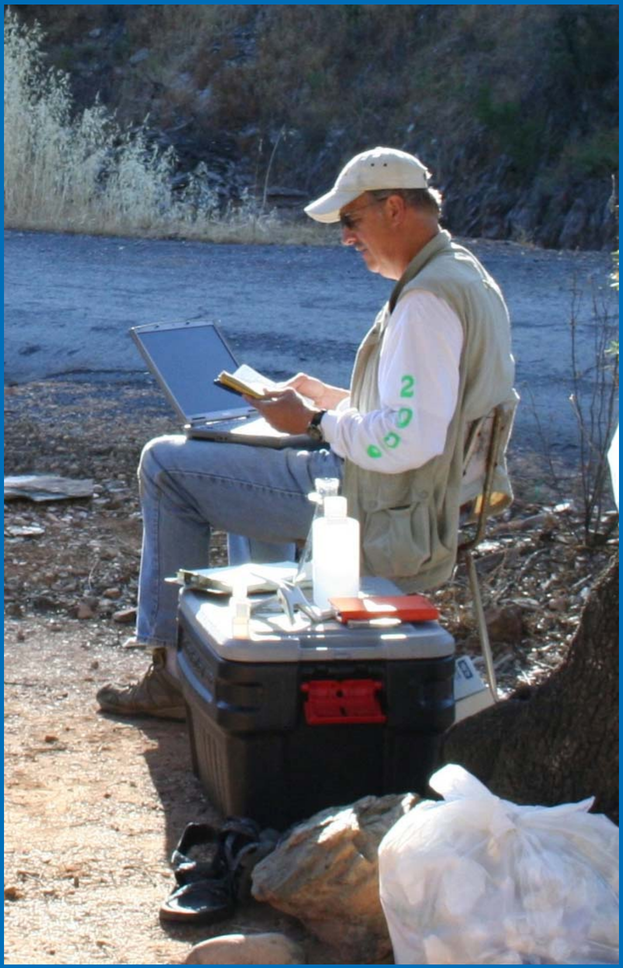
The Rest of Our Research Team

Chris Gammons

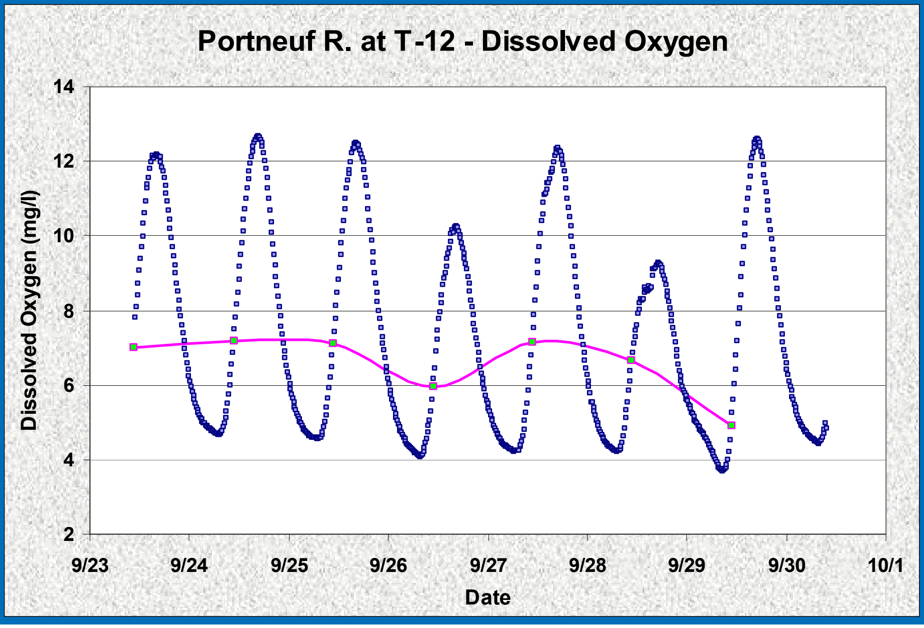


Steve Parker

Montana Tech, Butte, Montana

Variability in Water Quality

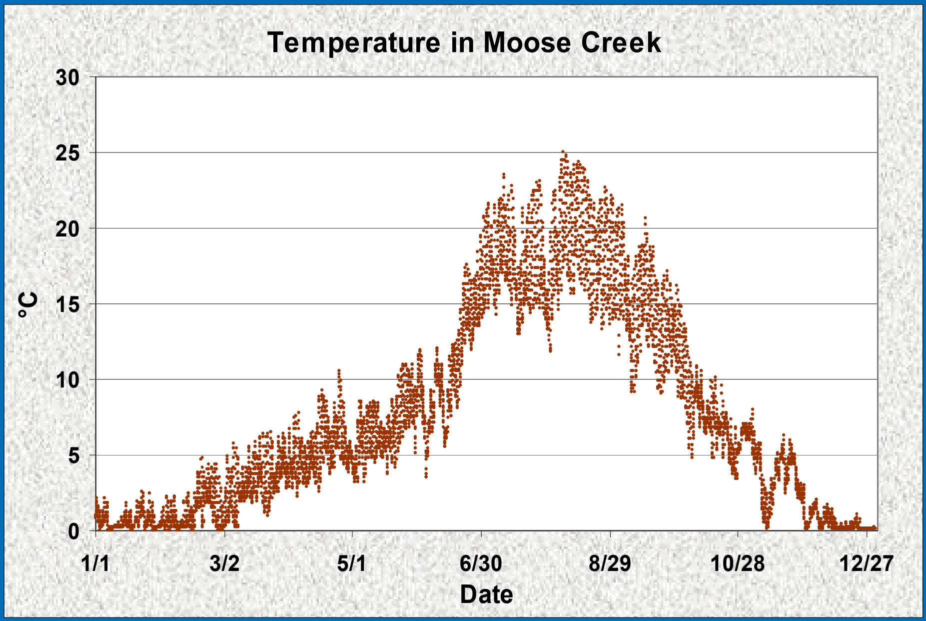
- Changing conditions (weather, seasonal, annual)
- Episodic events (rainfall runoff, spills)
- Anthropogenic activity (WWTP effluent, reservoir release for power generation, irrigation withdrawal)
- Diel biogeochemical cycling

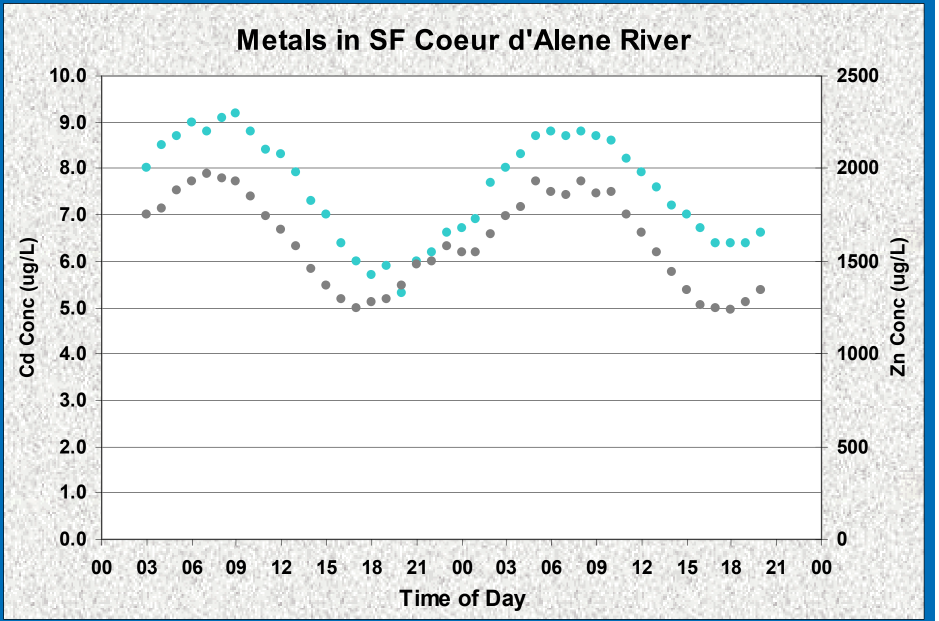


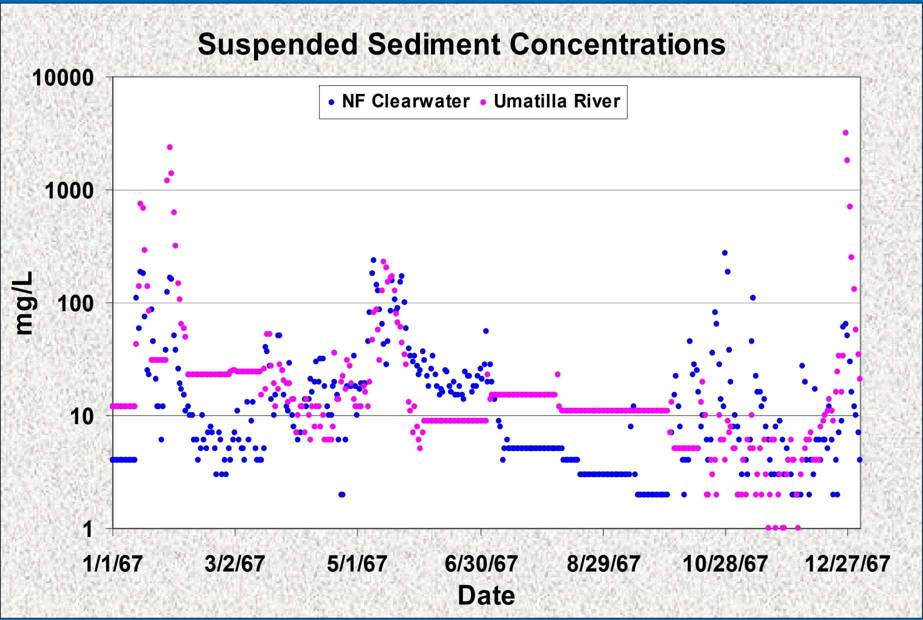
"Intensity of monitoring likely controls your perception of variability"

(Don Essig, Idaho DEQ)

Variability in Water Quality



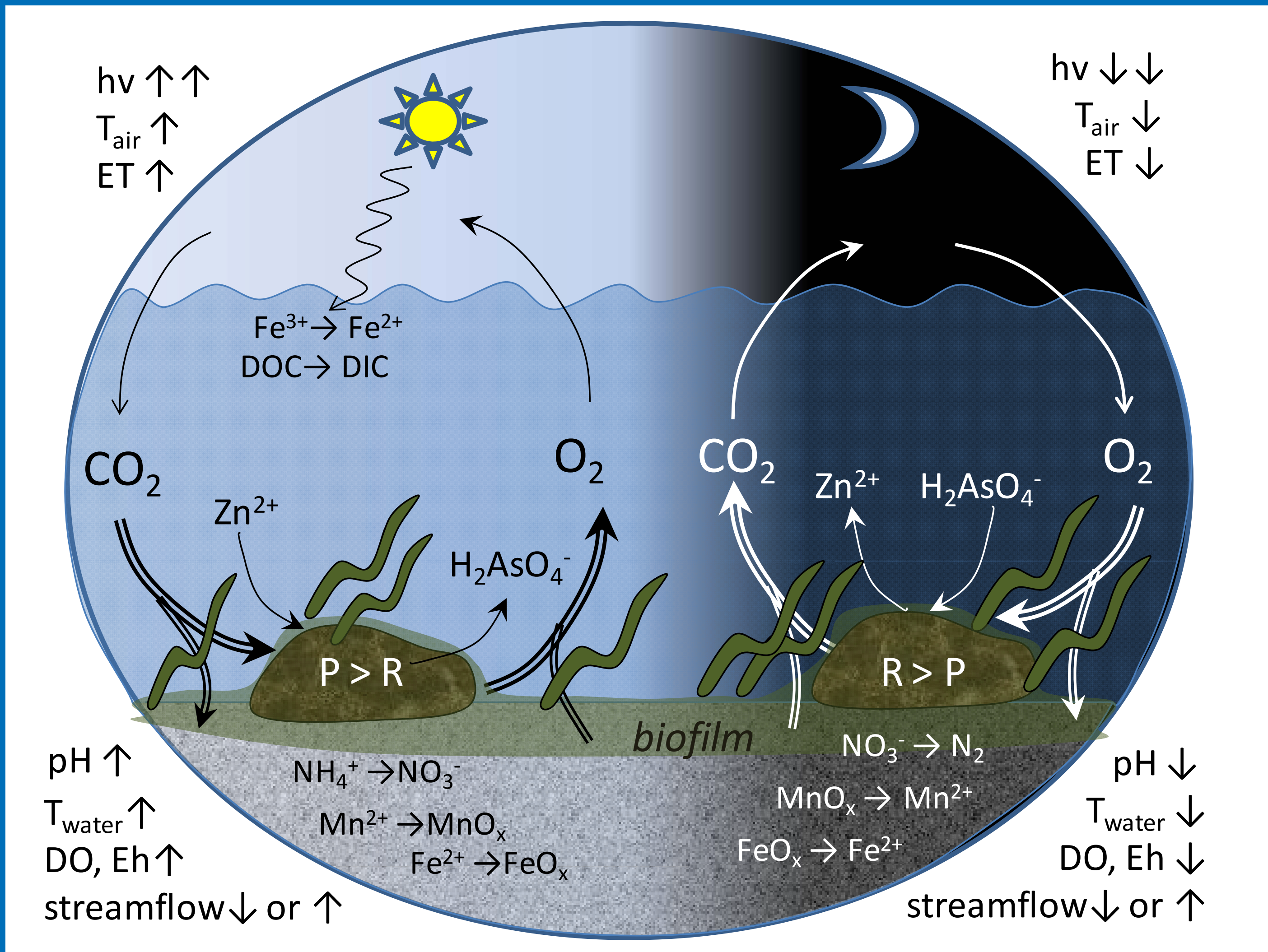




"Water quality is more variable than we know, and the more we look, the more we find."

(Don Essig, Idaho DEQ)

Diel Biogeochemical Cycling



USGS

(Nimick et al., 2011)

Diel Cycles: Mechanisms

Physical Processes

- Water temperature
- Streamflow
- Particle settling
- Nocturnal aquatic activity

Biogeochemical Processes

- Photosynthesis/respiration
- Photochemical reactions
- Reductive dissolution
- Adsorption/desorption
- Mineral and gas solubility
- Biological assimilation

White = primary process driven directly by sunlight

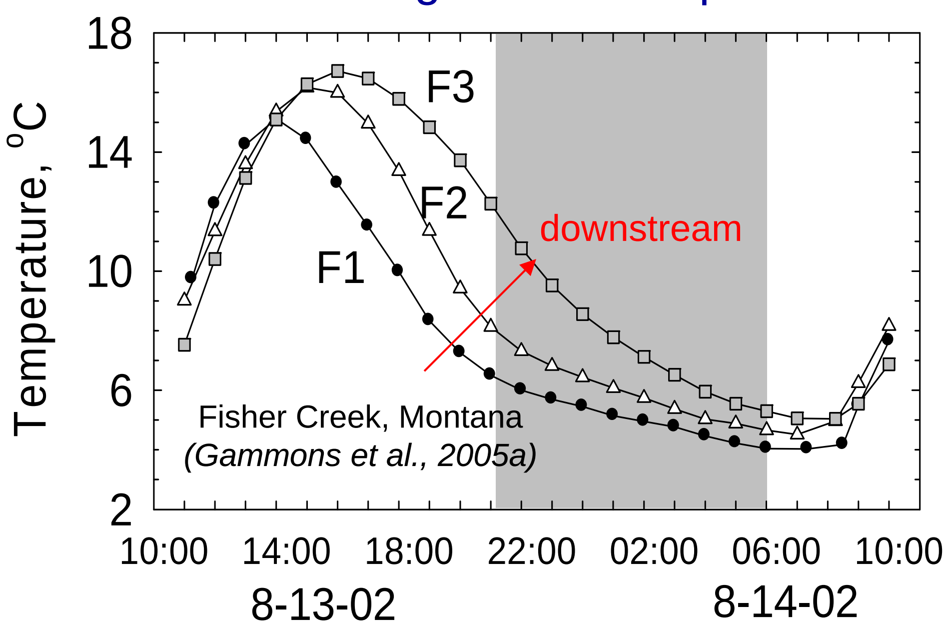
Pink = secondary process reacting to a primary process

Diel Temperature Cycles

Causes

- Solar heating
- Radiative cooling
- Groundwater inflow

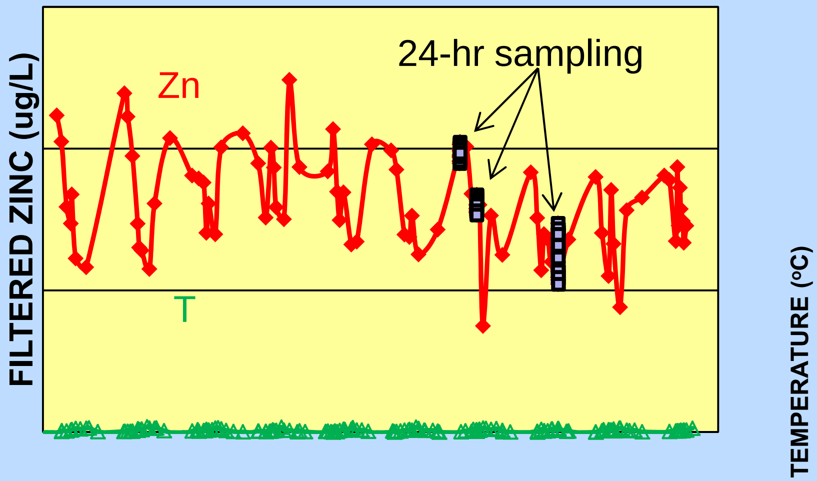
Downstream change in diel temperatures

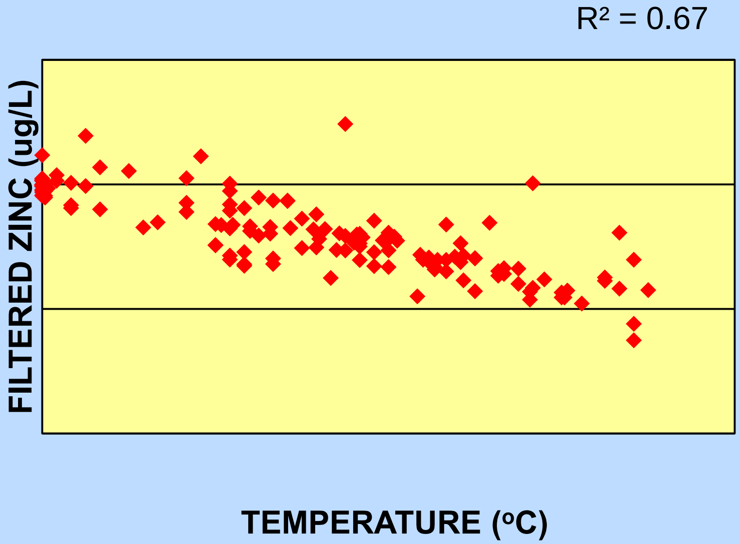


Importance

- Ecological stress
- Influences *kinetics* and *equilibrium* of aqueous reactions
  - Microbial reactions
  - Mineral and gas solubility
  - Adsorption
- Water viscosity
  - Streambed hydraulic conductivity
  - Particle settling

Importance of Temperature



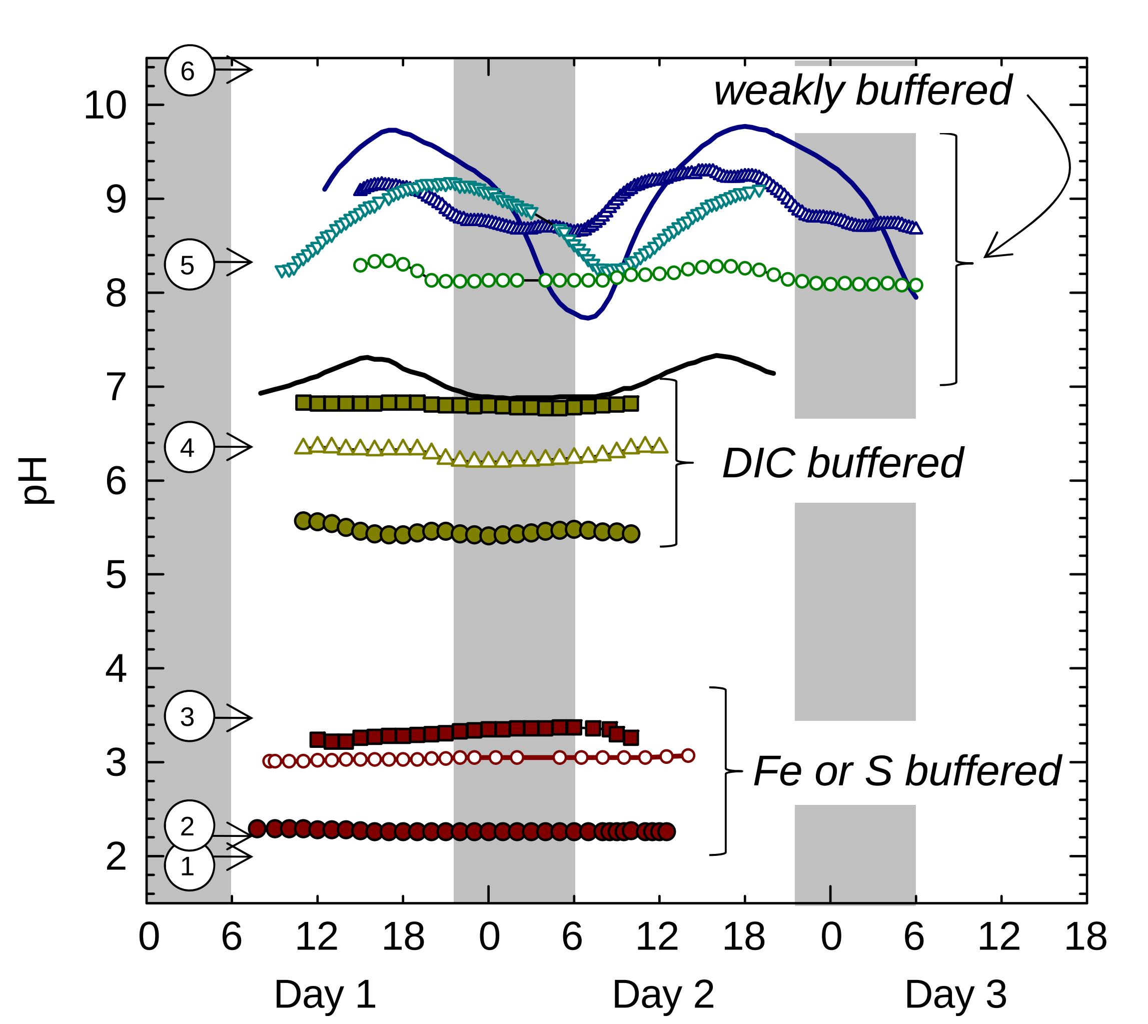


Silver Bow Creek, Montana

(USGS long-term monitoring data for 2002-2011)

Diel pH Cycles

- Diel pH changes are greatest for high-productivity, neutral-to-alkaline streams
- Diel pH changes in acidic streams are usually small



(Nimick et al., 2011)

USGS

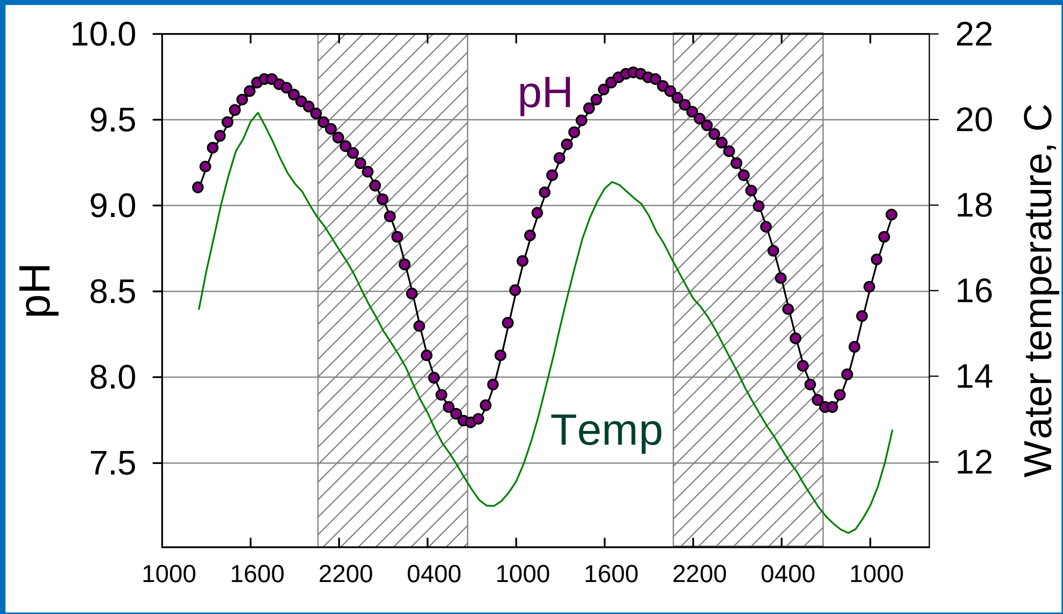
Diel pH Cycles

- Causes
  - Photosynthesis/respiration

$$CO_2 + H_2O \underset{\text{Night}}{\overset{\text{Day}}{\rightleftharpoons}} CH_2O + O_2$$

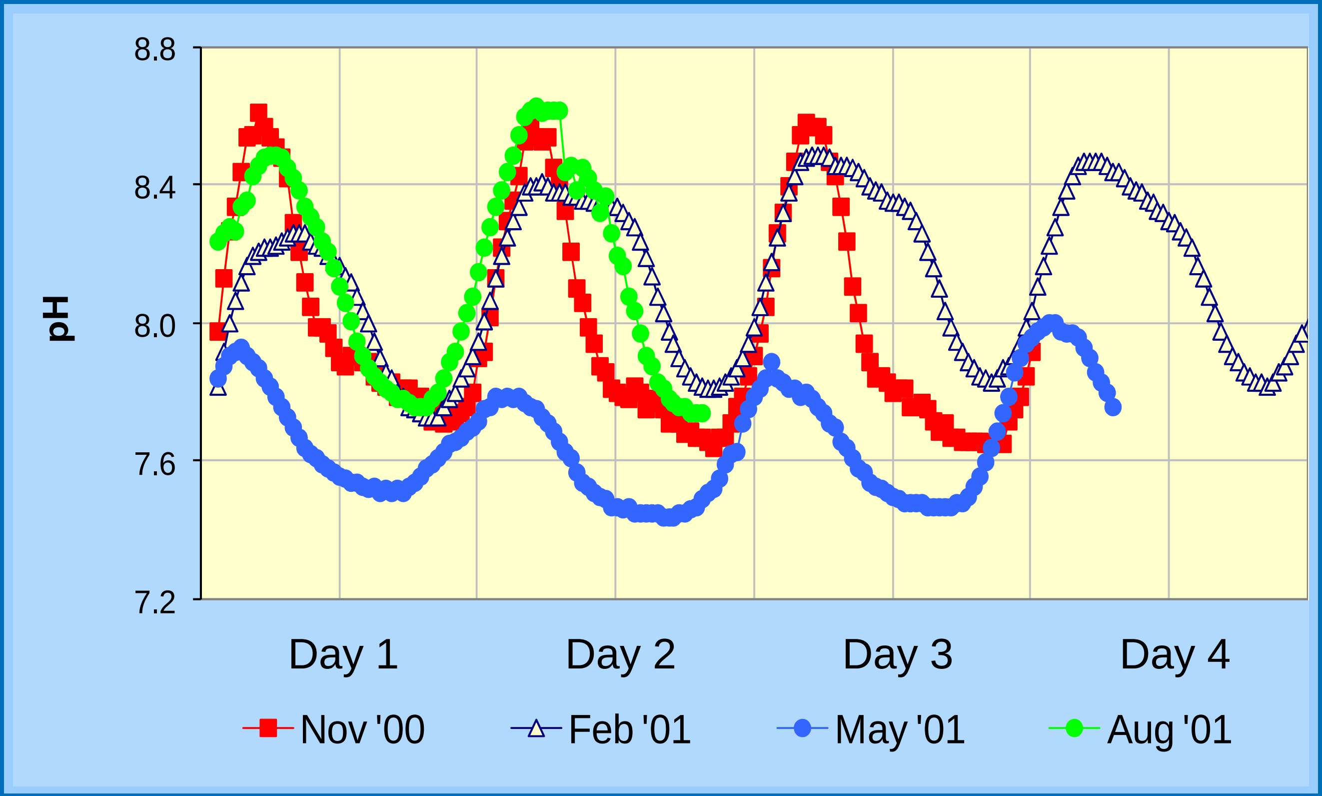
  - Changes in temperature
  - Changes in ground-water inflow
  - Fe chemistry

- Importance
  - Many reactions are pH-dependent:
    - Mineral solubility
    - Gas solubility
    - Adsorption



USGS

Seasonal Changes in Diel Cycles



- As long as the sun shines and the water is open, there are diel cycles!

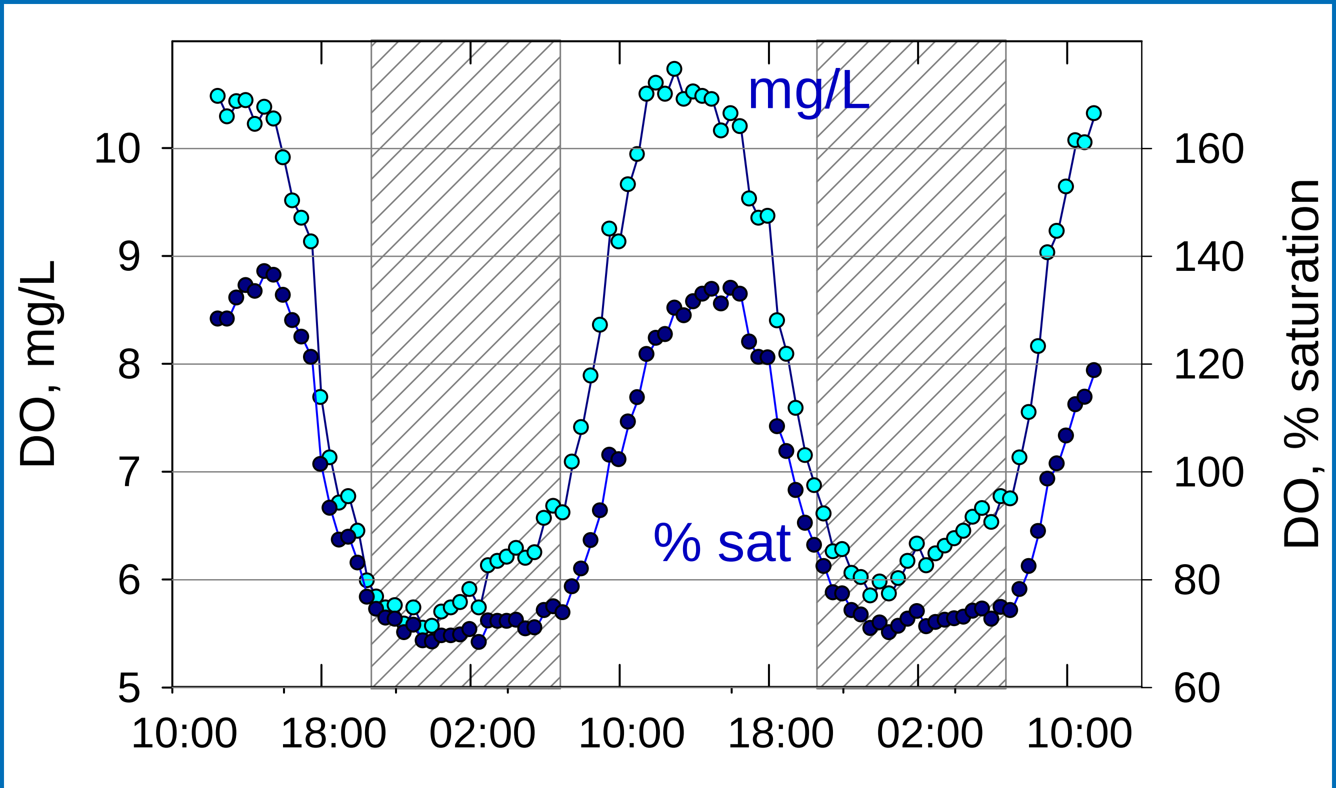
(Chris Gammons, Montana Tech)

Big Hole River in winter

USGS

Diel Cycles in Dissolved Oxygen

Photosynthesis/respiration: $CO_2 + H_2O \underset{\text{Night}}{\overset{\text{Day}}{\rightleftharpoons}} CH_2O + O_2$

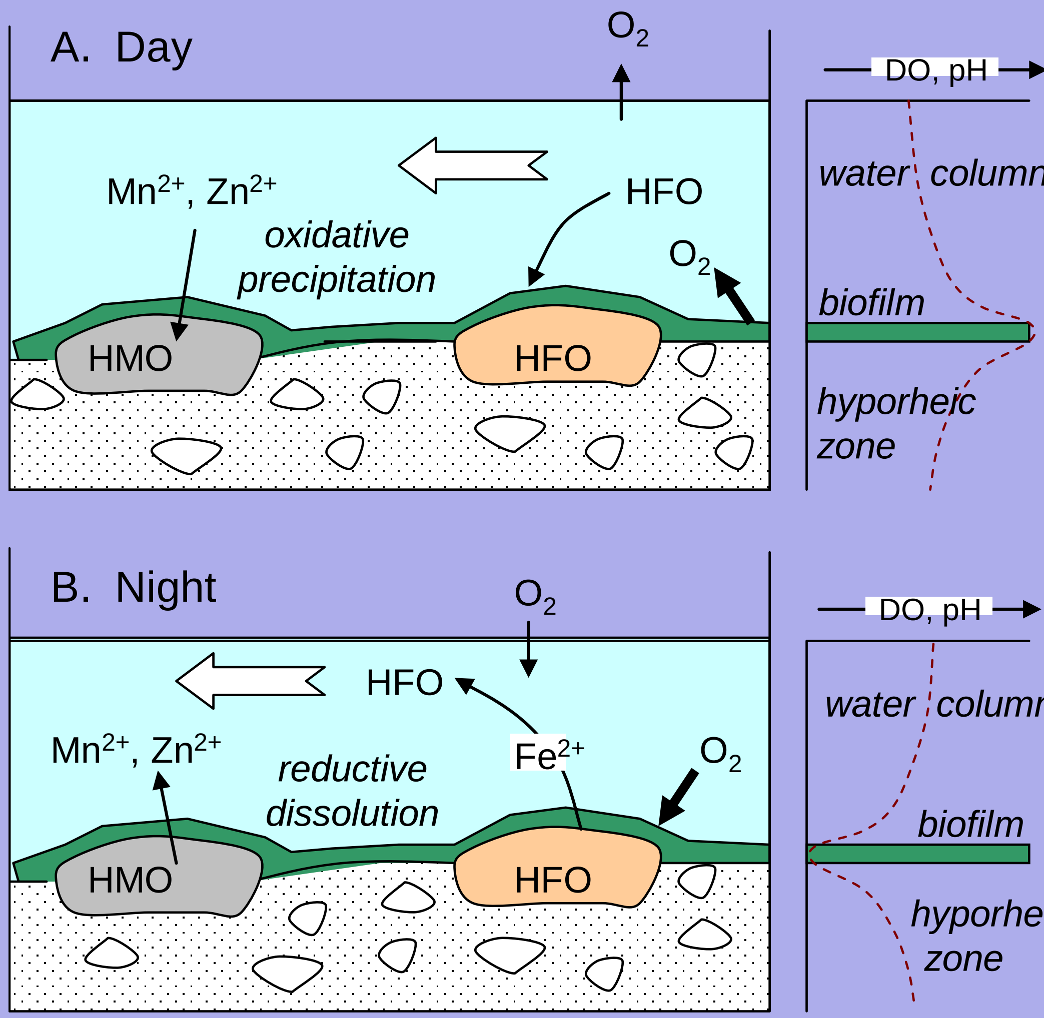


Big Hole River, MT

- DO changes are largest in slow-moving, high-productivity streams
- DO usually peaks at noon (sun is directly overhead)

Diel Cycling in Biofilms vs. Bulk Water

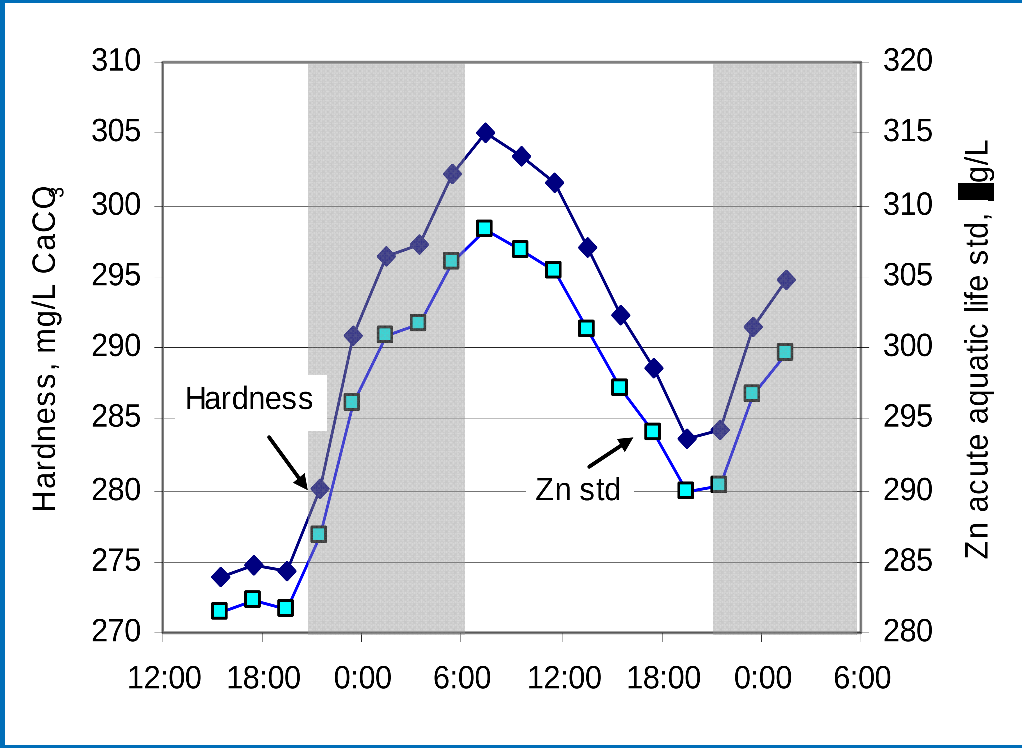
- **Changes in pH, DO, and redox are magnified in biofilms relative to the bulk water!**



(Parker et al., 2007)

Diel Cycles in Hardness

- **Hardness is proportional to Ca & Mg concentration**
- **Diel hardness cycles caused by diel changes in**
  - **Streamflow**
  - **Calcite ($CaCO_3$) precipitation and dissolution**
- **Importance: Aquatic life standards for many toxic metals are hardness-dependent**

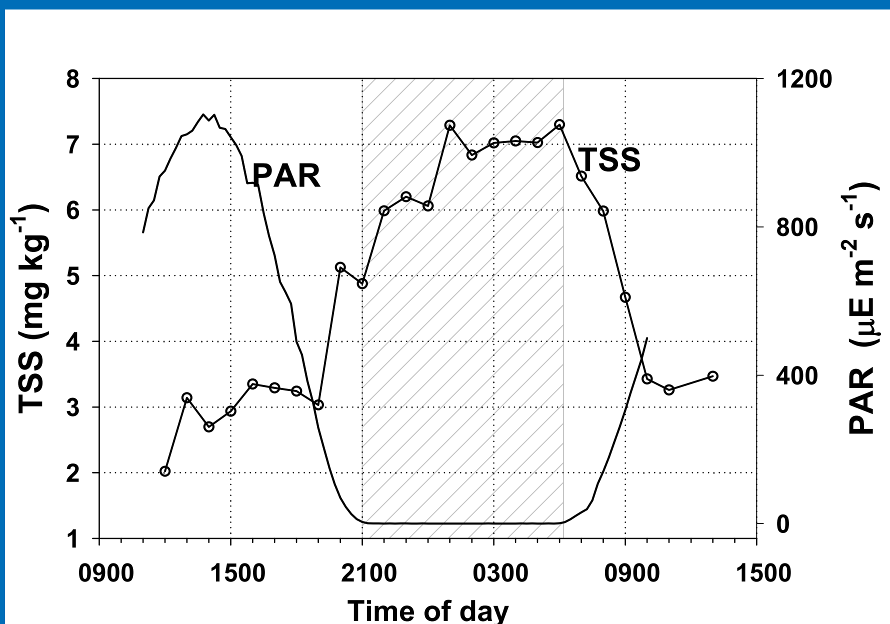


Mill-Willow Bypass, Montana, August 2005
(Gammons et al., 2007)

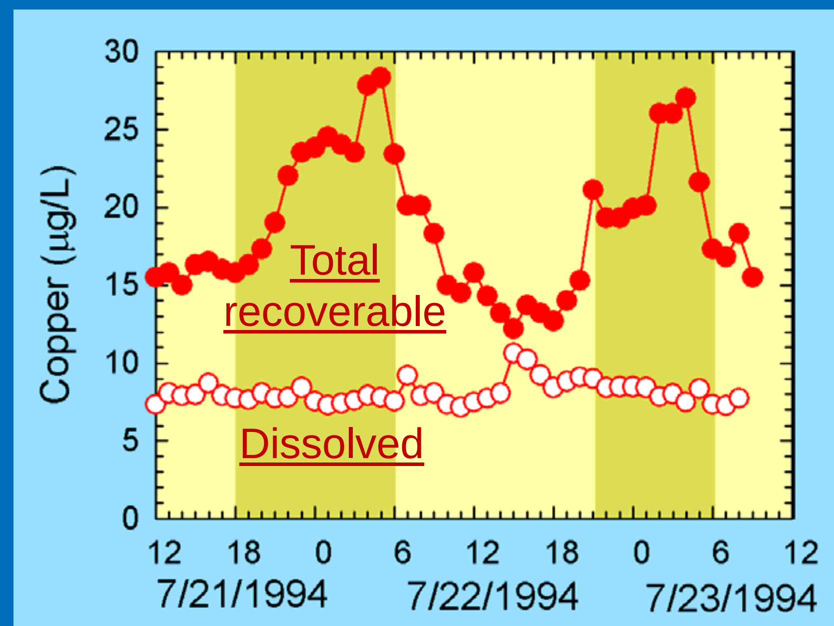
USGS

Diel Cycles in Suspended Solids

- **Particulate concentrations increase at night:**
  - Foraging of benthic macroinvertebrates
  - Oxides form as Fe is released by reductive dissolution in biofilms
  - Particle settling rate decreases as temperature decreases



Clark Fork River, Montana
(Parker et al., 2007)

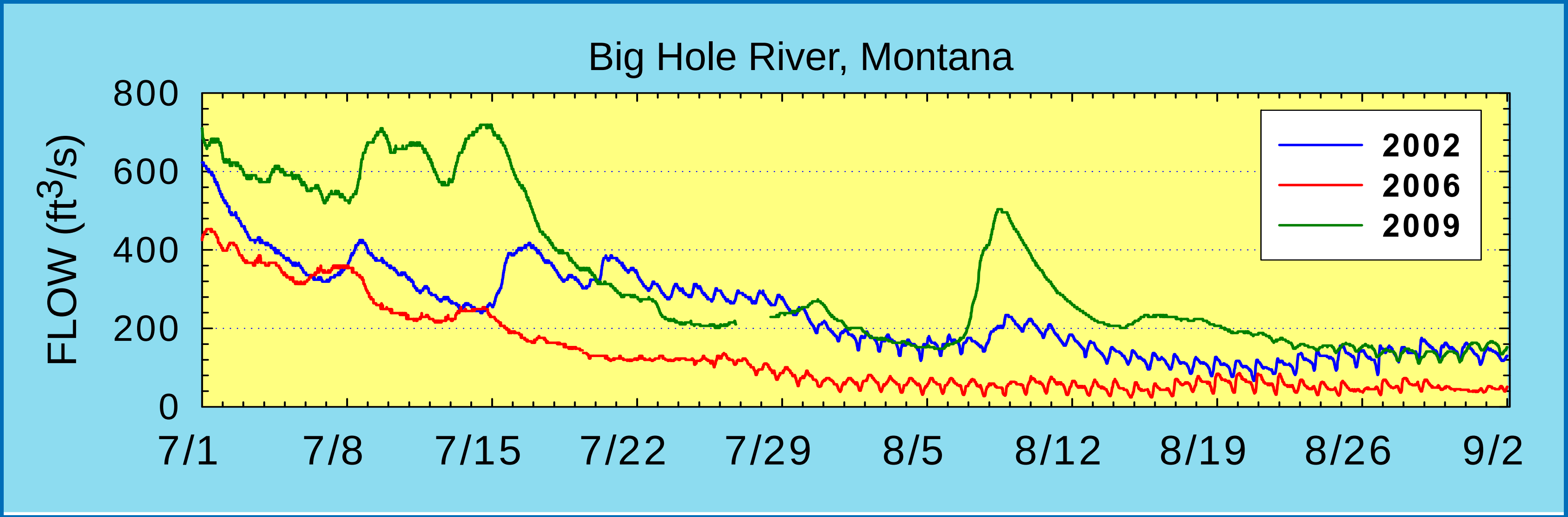


Clark Fork River, Montana
(Brick and Moore, 1996)

USGS

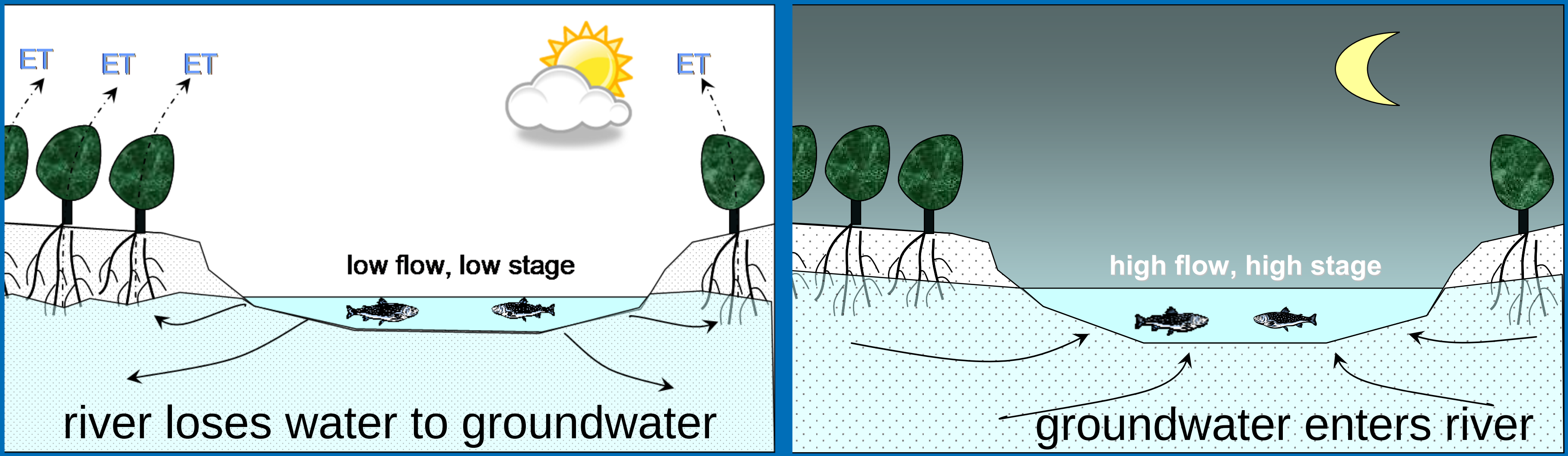
Diel Streamflow Cycles

- Freeze/thaw
  - Ice formation
  - Snow melt
- Evapotranspiration
- Temperature-dependent streamflow loss
- Anthropogenic
  - Wastewater or reservoir discharge
  - Irrigation withdrawals
- Macrophyte dams



Diel Streamflow Cycles

Evapotranspiration (ET) typically changes flow by <20%



- Diel streamflow cycles affect:
  - Solute <u>concentration</u> (dilution)
  - Solute <u>load</u> (load = concentration x flow)

USGS

Diel Cycling of Nutrients

NITROGEN

- Nitrate (NO_3^-)
- Nitrite (NO_2^-)
- Nitrous oxide (N_2O)
- Nitrogen (N_2)
- Ammonia (NH_4^+)
- Organic-N
- Suspended solids

PHOSPHORUS

- Orthophosphate (HPO_4^{-2})
- Organic-P
- Suspended solids

Big Hole River, Montana

USGS

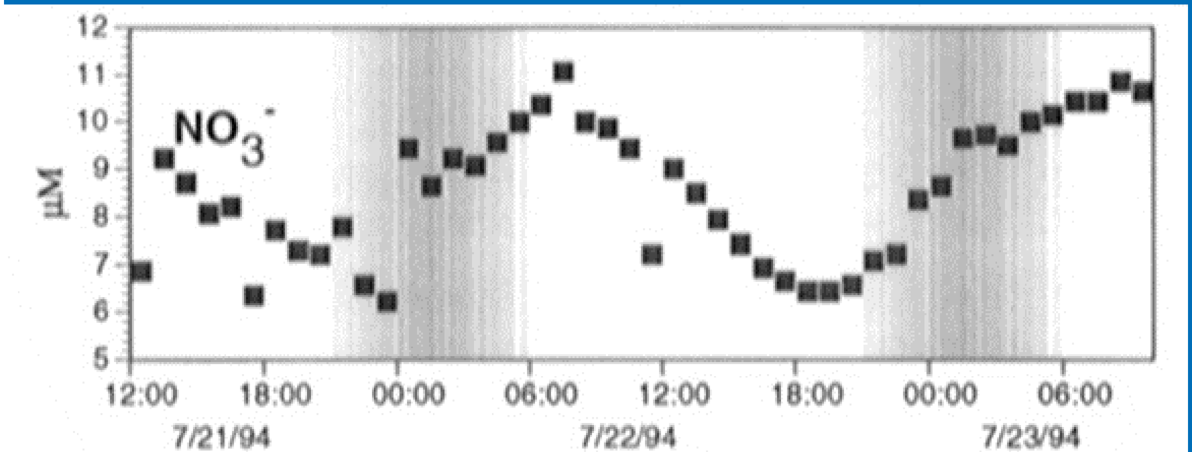
Diel Cycling of Nutrients

- Diel redox cycles
  - Nitrification (ammonia + $O_2 \rightarrow$ nitrate)
  - Denitrification (nitrate + organic C $\rightarrow N_2$)
  - Anammox (ammonia + nitrate $\rightarrow N_2$)
- Diel changes in rate of uptake by biota
- Diel changes in delivery rate from hyporheic or benthic zones
- Sorption/desorption of P

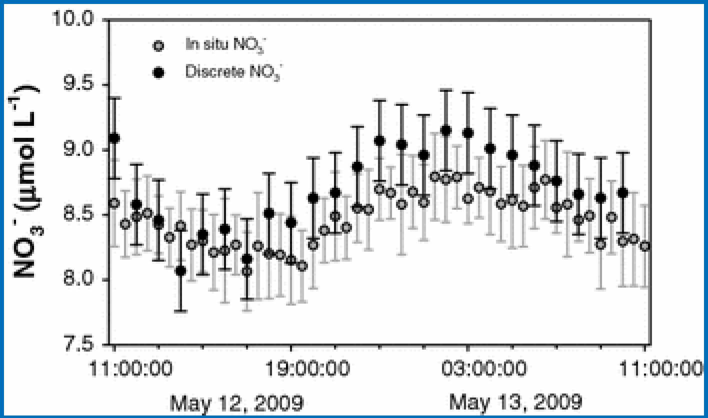
Silver Bow Creek, Montana

USGS

Diel Cycling of Nitrate



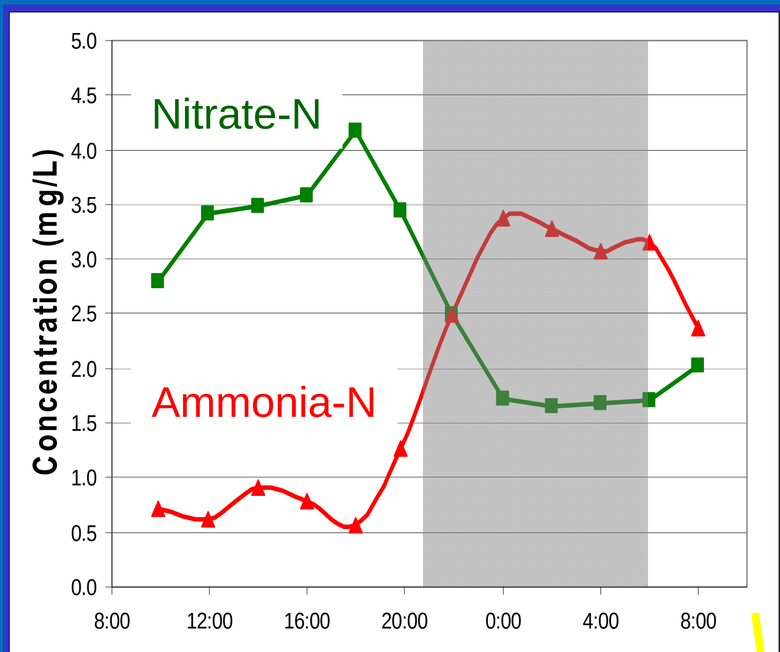
Clark Fork River, Montana (Brick and Moore, 1996)

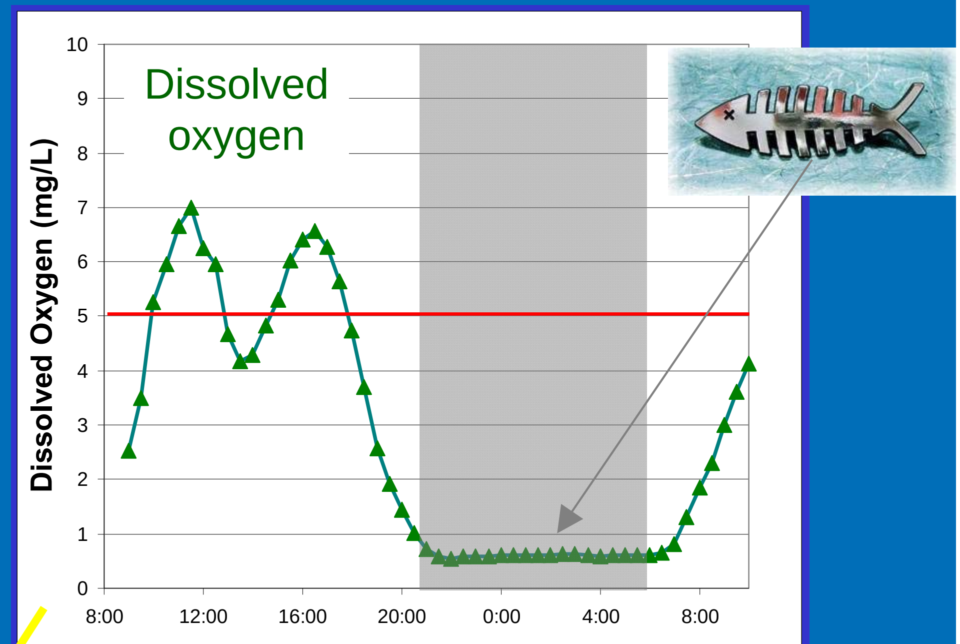


USGS

Sleepers River, Vermont (Pellerin et al., 2012)

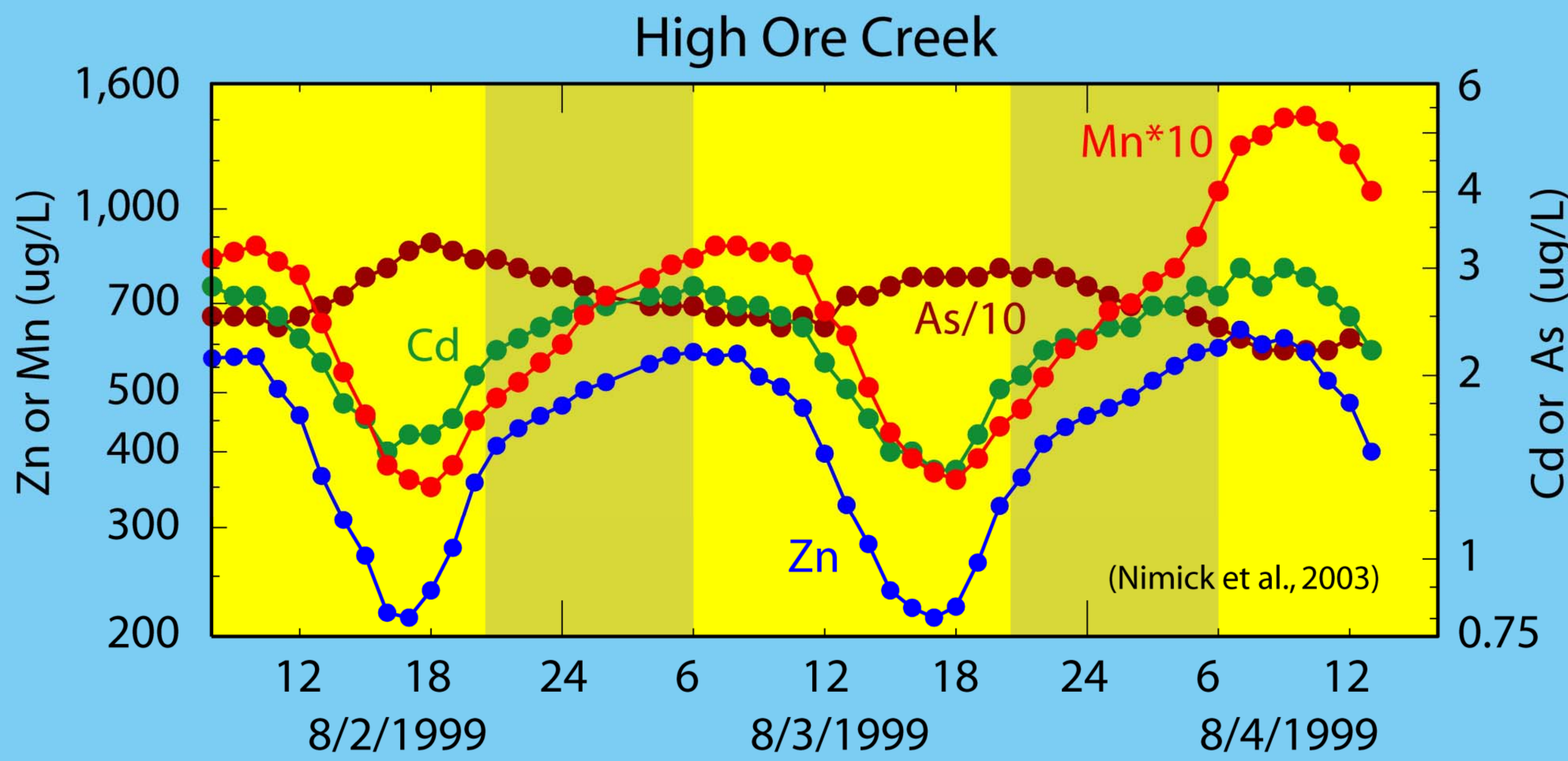
Diel Nutrient Cycling in Silver Bow Creek





(Gammons et al., 2011)

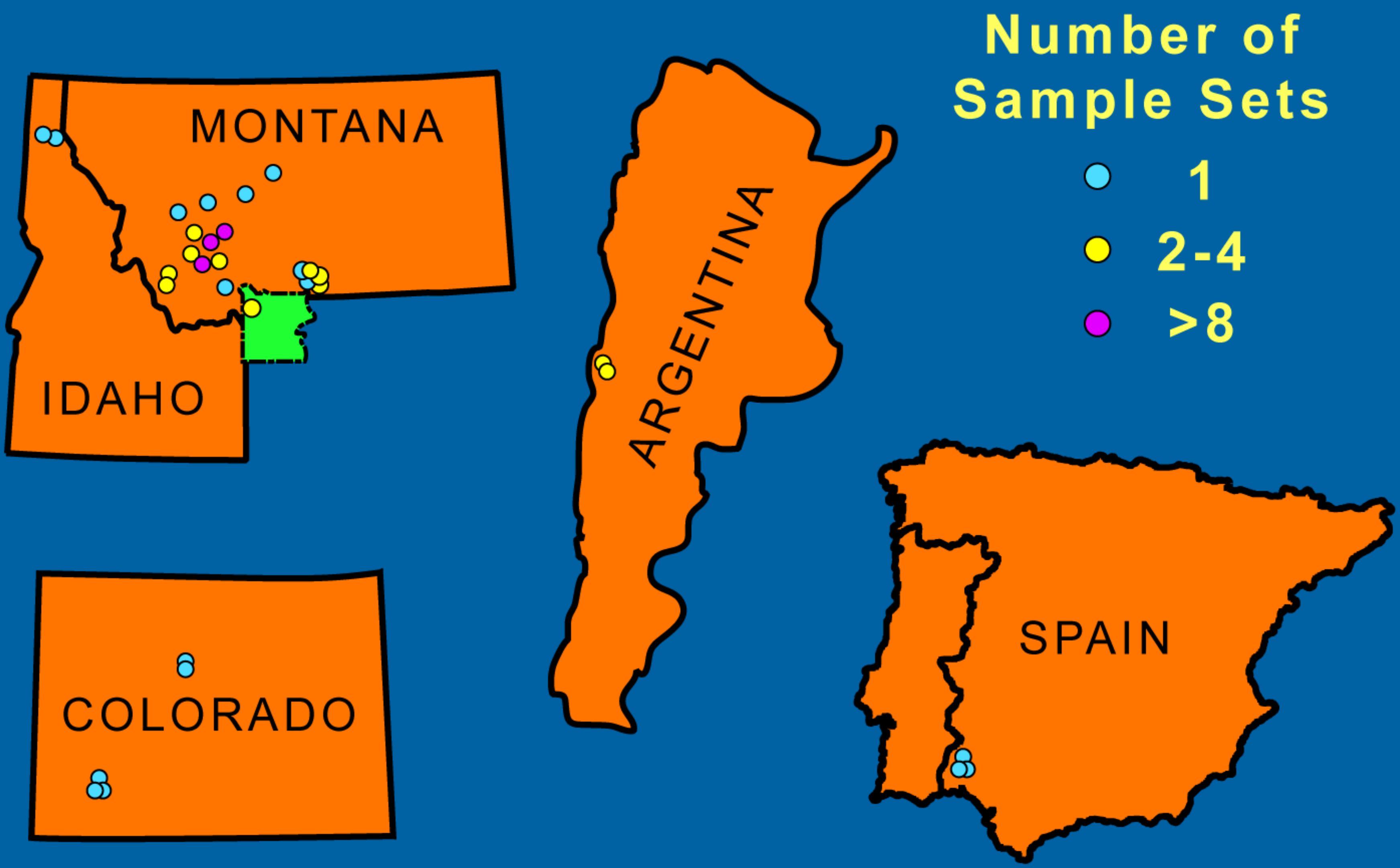
Diel Trace-Element Cycles in Neutral and Alkaline Streams



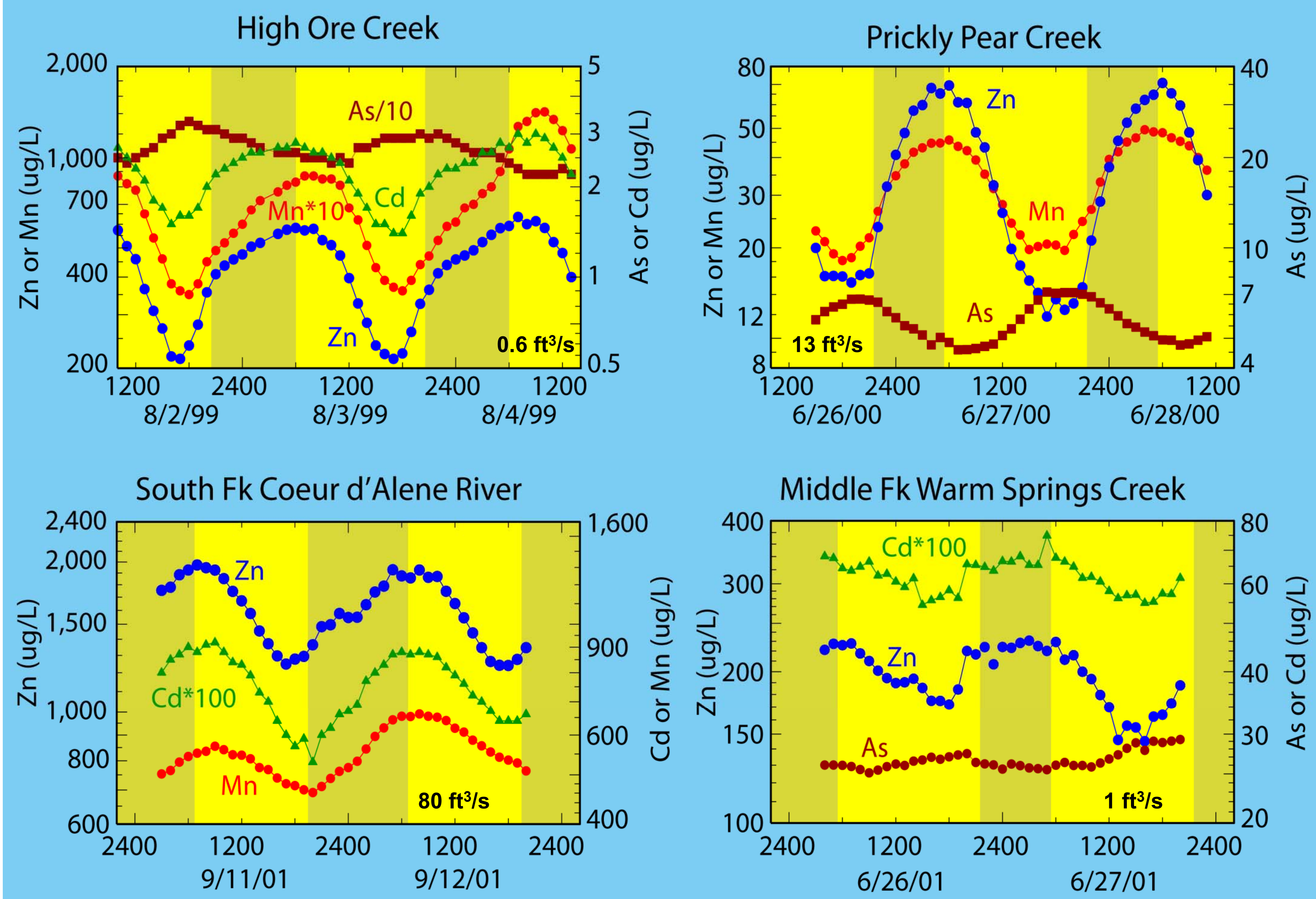
Arsenic	Cadmium	Manganese	Zinc
22-33 μg/L	1.4-3.0 μg/L	35-142 μg/L	214-634 μg/L
50%	110%	306%	196%

USGS

Diel Trace-Element Cycles



Diel sampling sites – 1990-2011



(Nimick et al., 2003)

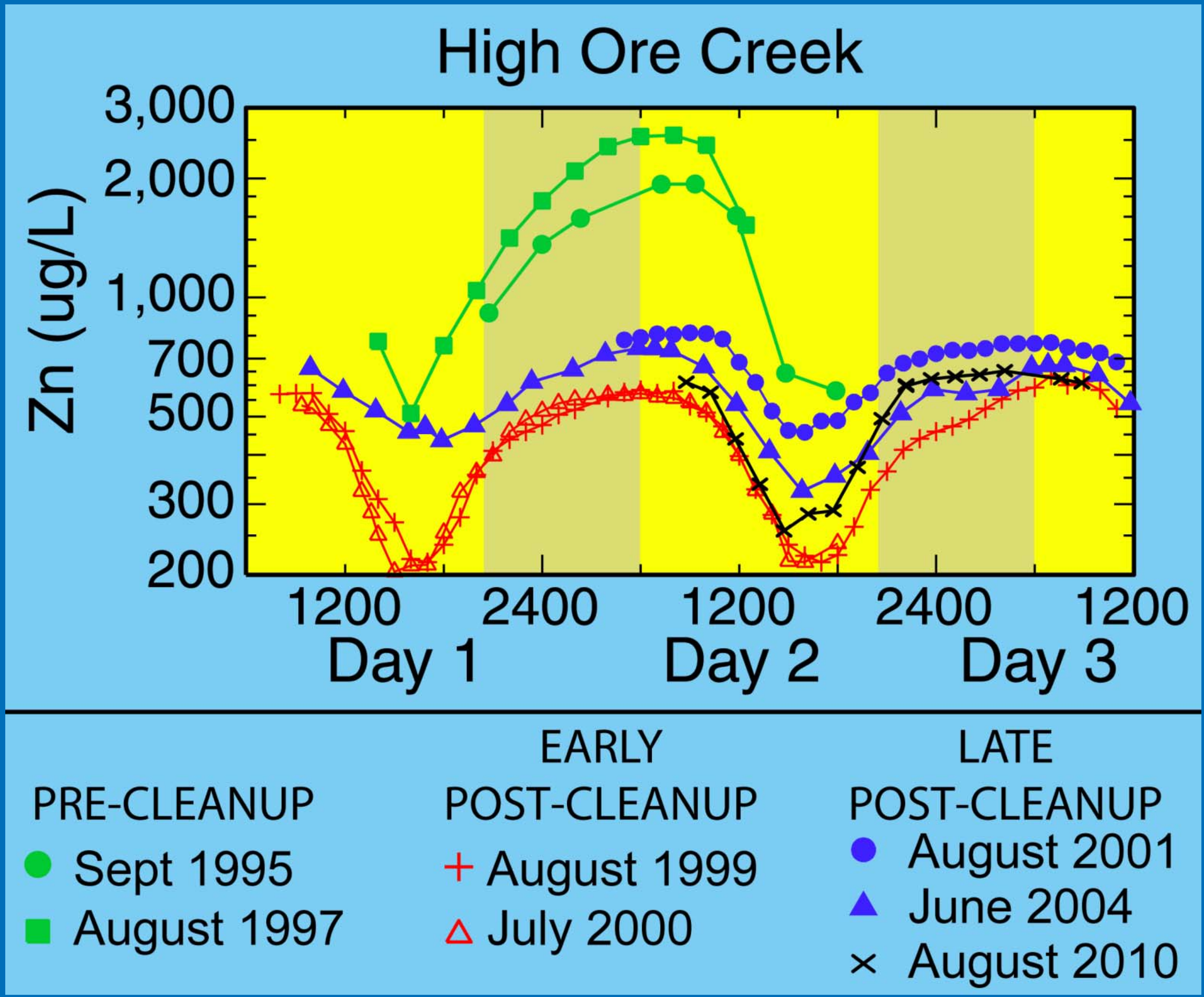
Magnitude of Diel Cycles for Dissolved Trace Elements

Trace Element[1]	Maximum Daily Increase (%) [2]	Number of Diel Samplings[2]
Zn	990	>35
Rare earth elements	830	2
Cd	330	12
Mn	306	20
Ni	167	1
U	125	2
Methyl Hg	93	2
As	54	>25
Cu (pH = 6.8 – 7)	140	3
Cu (pH > 7)	<10	12
Se	<10	1

1. Near-neutral to alkaline streams unless otherwise noted
2. See Nimick et al. (2011) and Balistrieri et al. (2012) for references

USGS

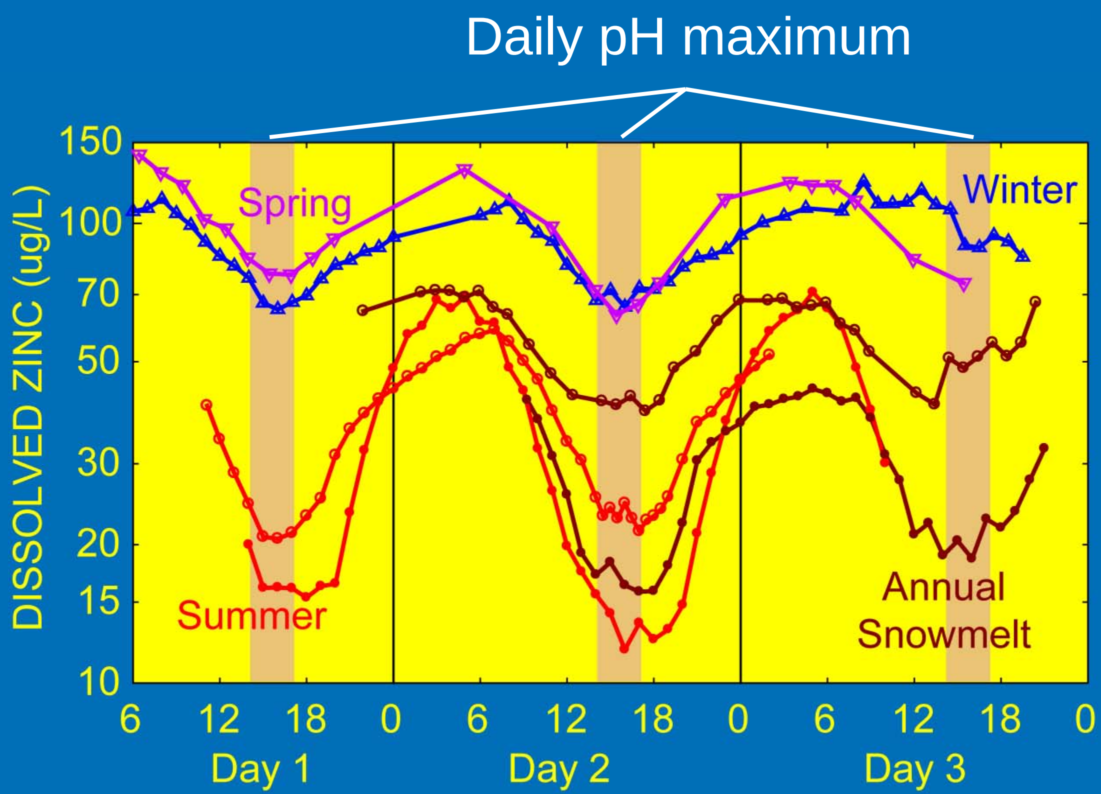
Year-to-Year Variation



USGS

Seasonal Variation

Prickly Pear Creek, Montana

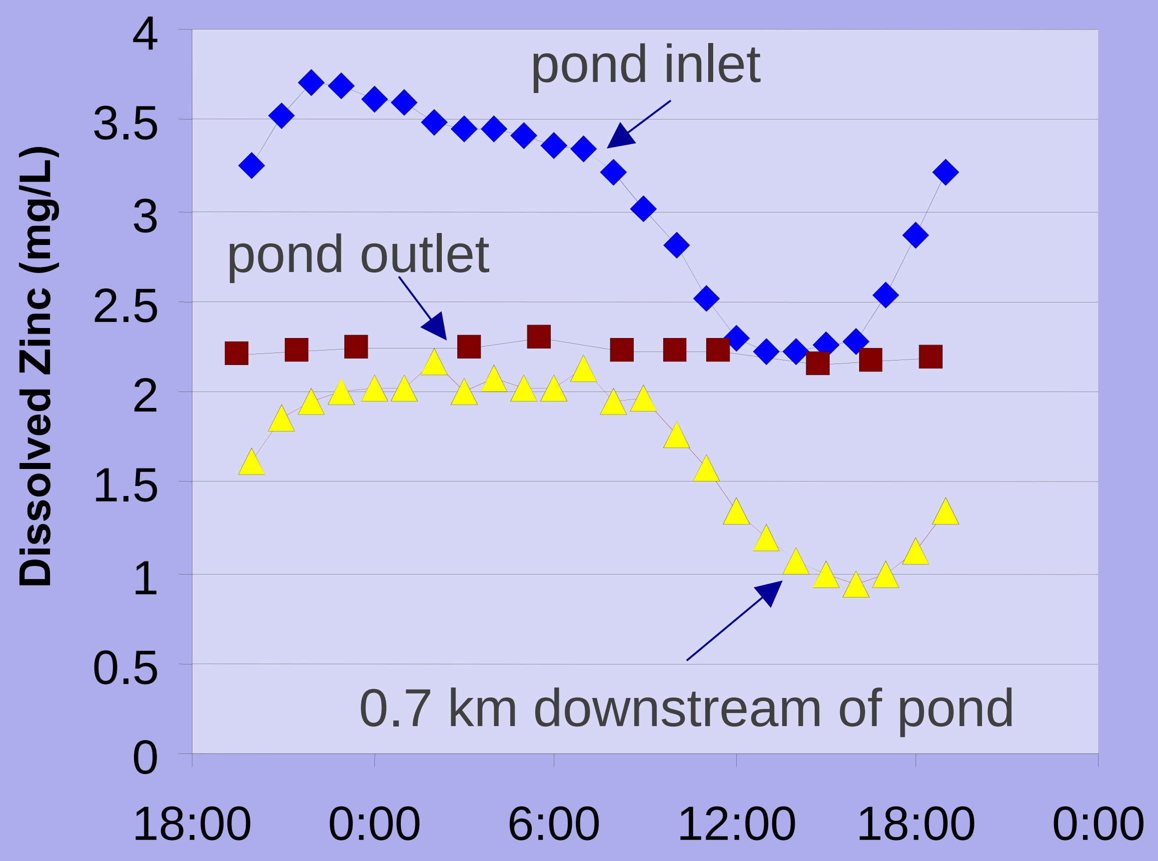


(Nimick et al., 2005)

USGS

Lakes versus Rivers

- **Lakes and ponds tend to "even out" diel cycles found in streams**



High Ore Creek, MT
(*Gammons et al., 2007b*)

USGS

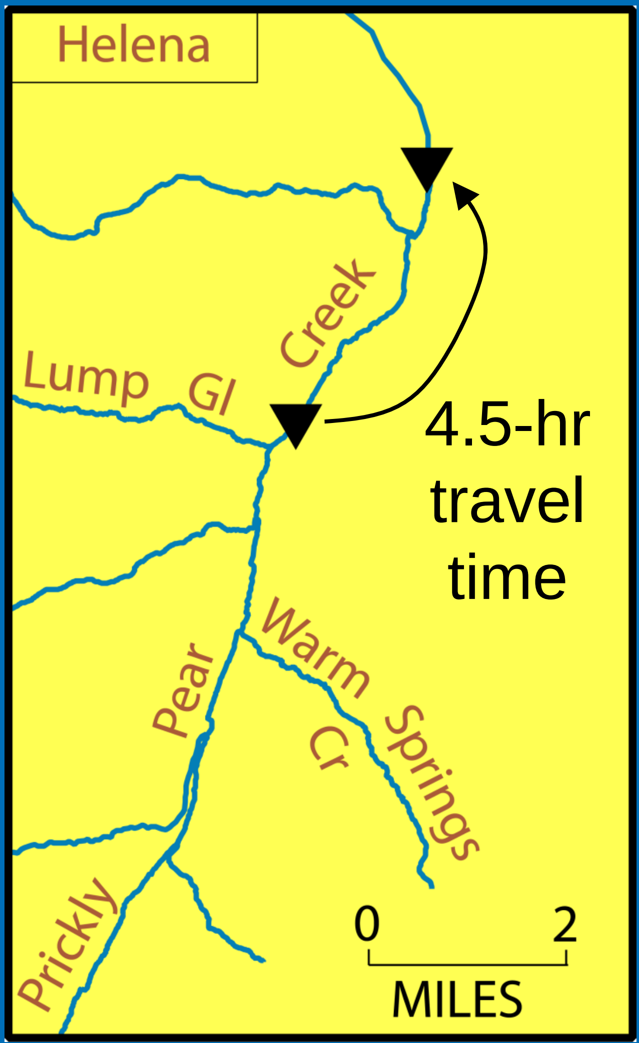
Possible Causes – Dissolved Metal Cycles

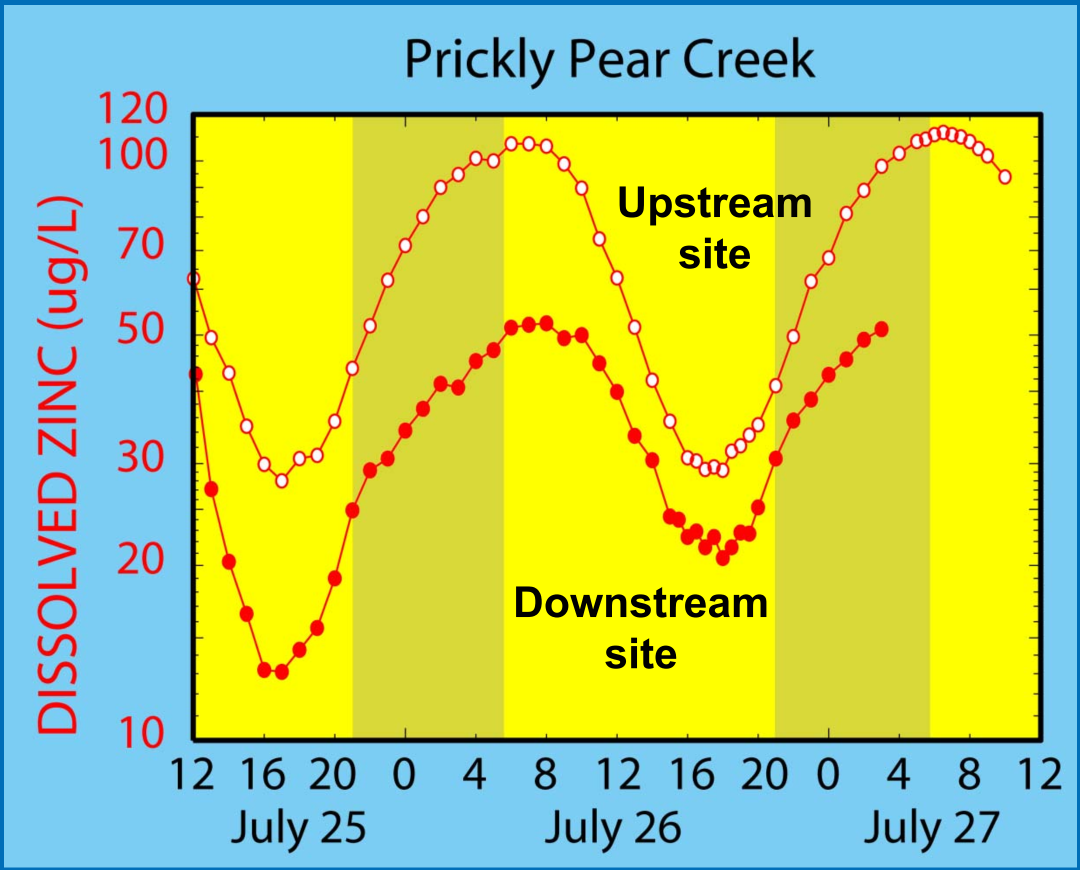
- Diel variation in metal input
- Biological uptake
- Precipitation-dissolution reaction
- Sorption-desorption reaction

South Fork Coeur d'Alene River

USGS

Cause: Diel Source Input or Instream Process?





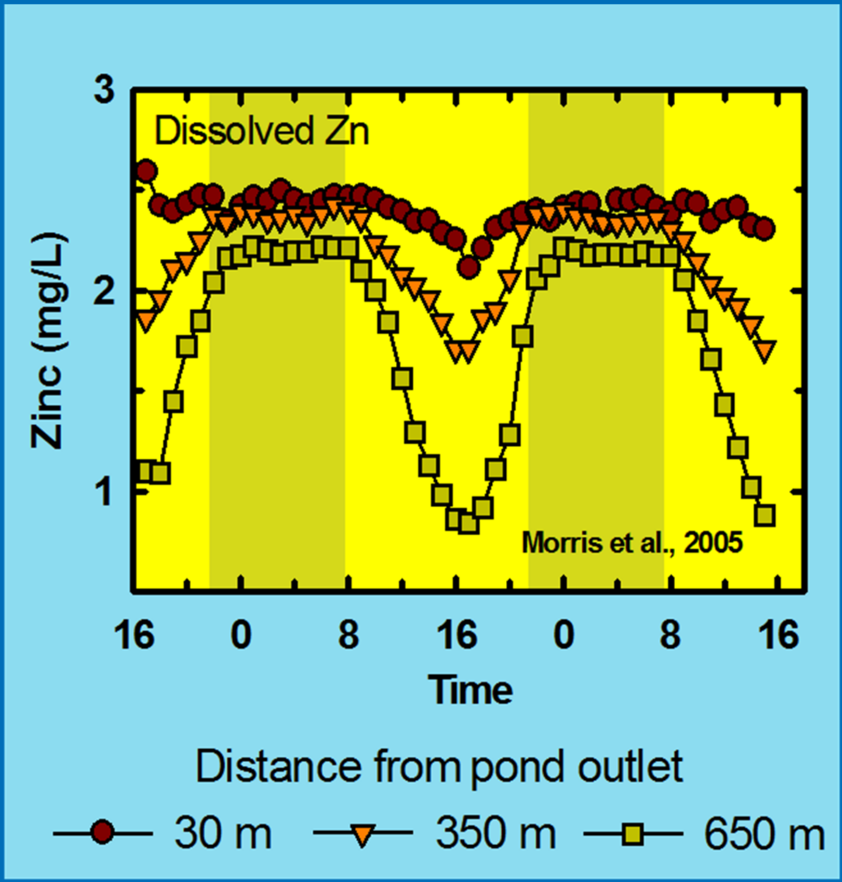
No lag time means cycles caused by instream process

USGS

Cause: Biological Uptake

- **Uptake by biofilm and periphyton is plausible reason for Zn cycles but not As cycles, which have opposite timing**

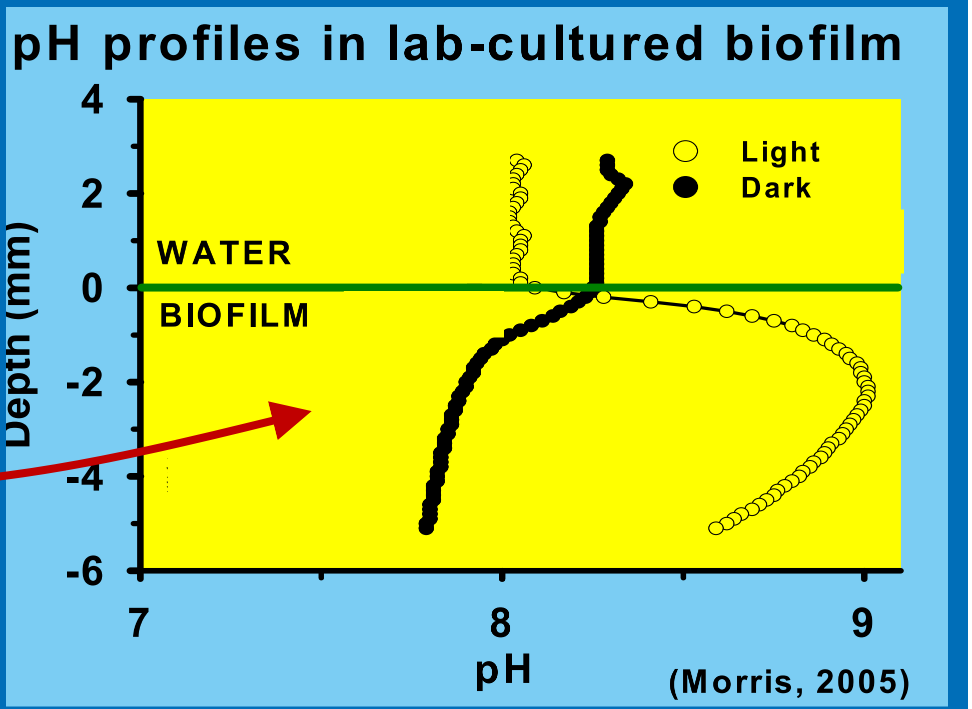
High Ore Creek



USGS

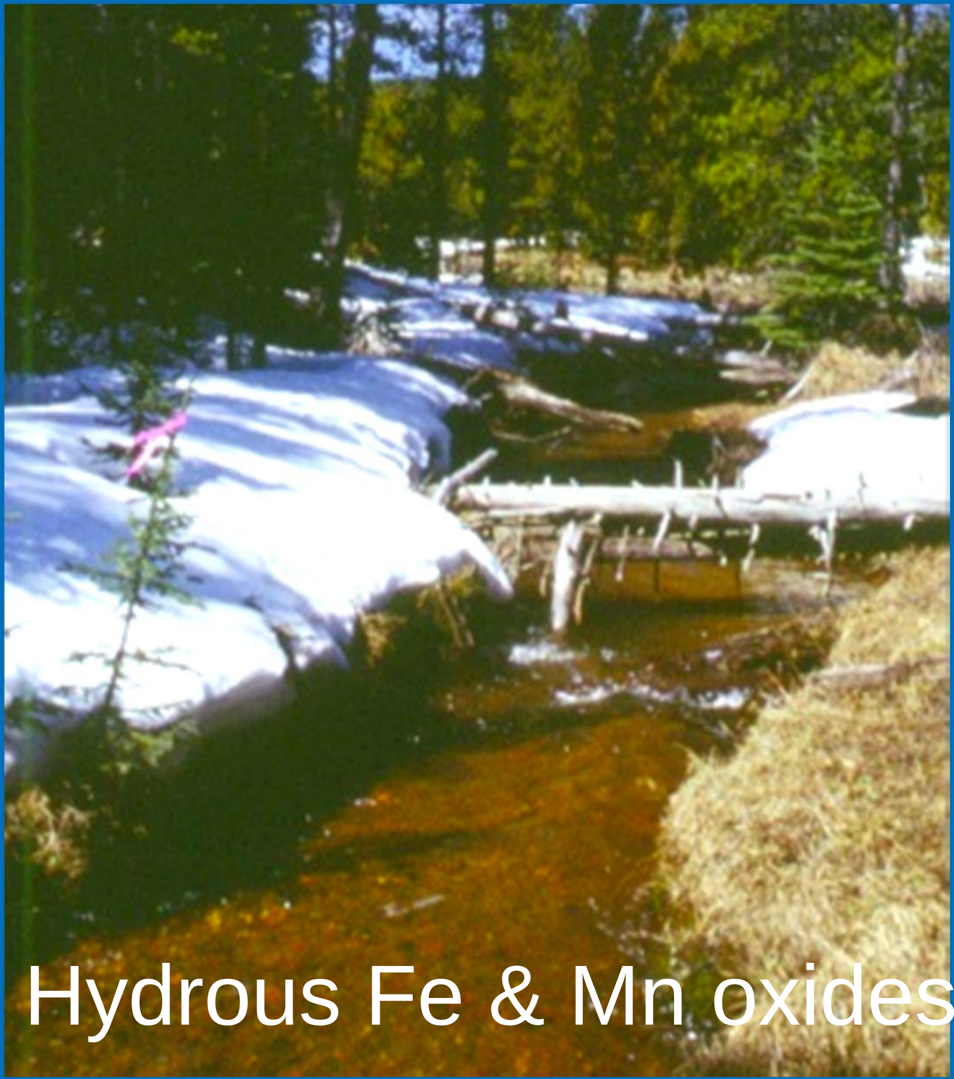
Cause: Precipitation-Dissolution

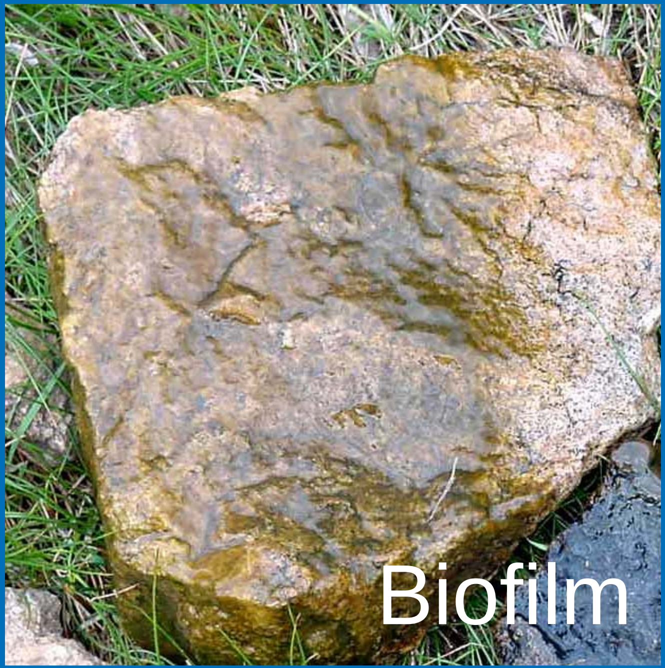
- Daytime increases in pH and water temperature increase mineral saturation and precipitation
  - $Zn^{+2} + CO_3^{-2} = ZnCO_{3(s)}$ (smithsonite)
  - $Ca^{+2} + CO_3^{-2} = CaCO_{3(s)}$ (calcite)
- Reversible reaction
- pH changes much greater within biological surface
- Does not explain arsenic



USGS

Cause: Sorption-Desorption



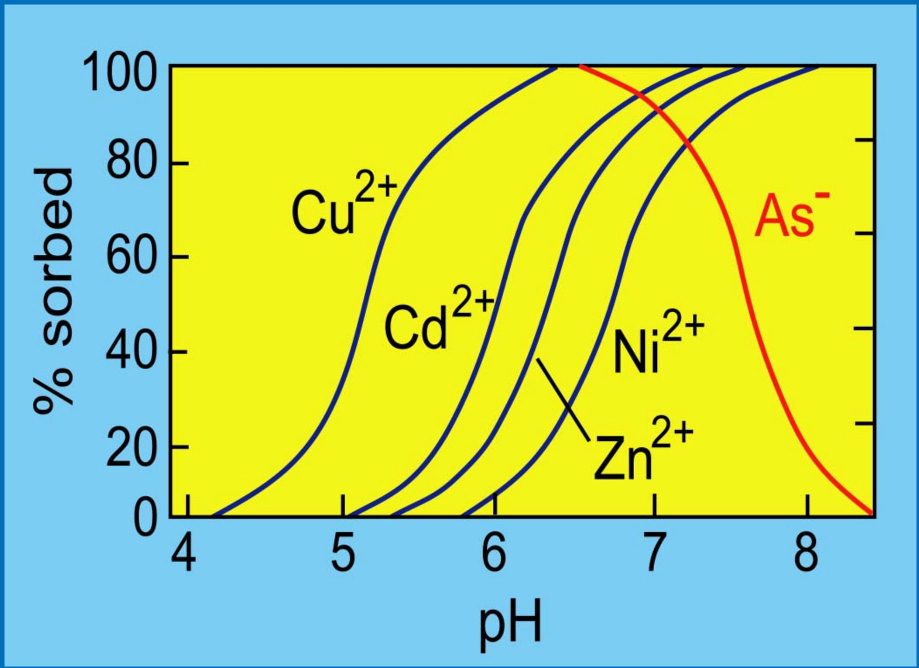


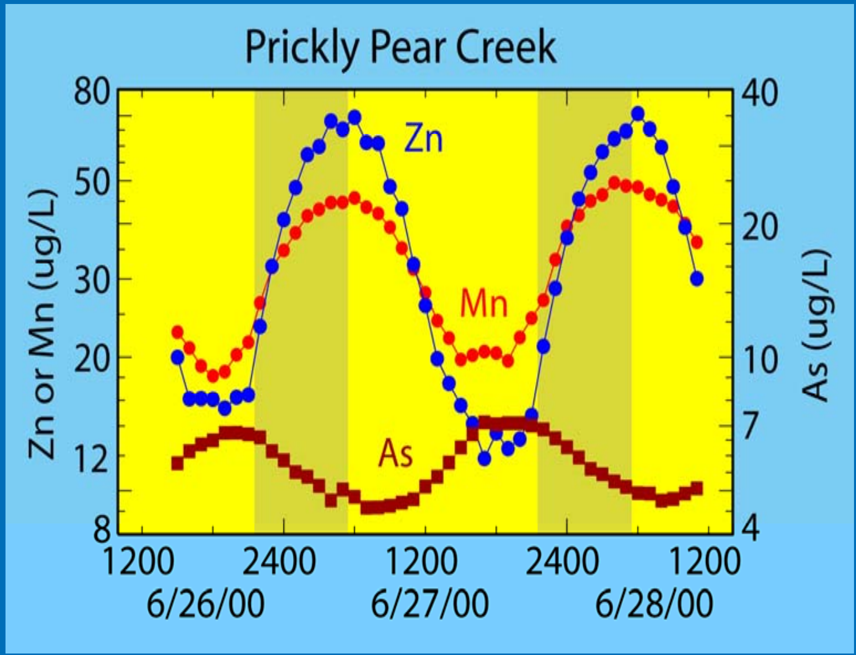
Possible inorganic and organic sorption substrates

USGS

Cause: Sorption-Desorption

- Cation sorption increases and anion sorption decreases with either:
  - increased pH, or
  - increased temperature





Not All Streams Exhibit Diel Cycling

Big cycles

- Shallow, clear
- High productivity
- Large pH and T changes

Small or nonexistent cycles

- Deep, turbid, shaded
- Low productivity
- Small pH and T changes

USGS

Diel Processes in Acidic Streams

(Gammons et al., 2010)

(Gammons et al., 2005a,b)

(Gammons et al., 2008)

(Parker et al., 2008)

Fe(III) Photoreduction

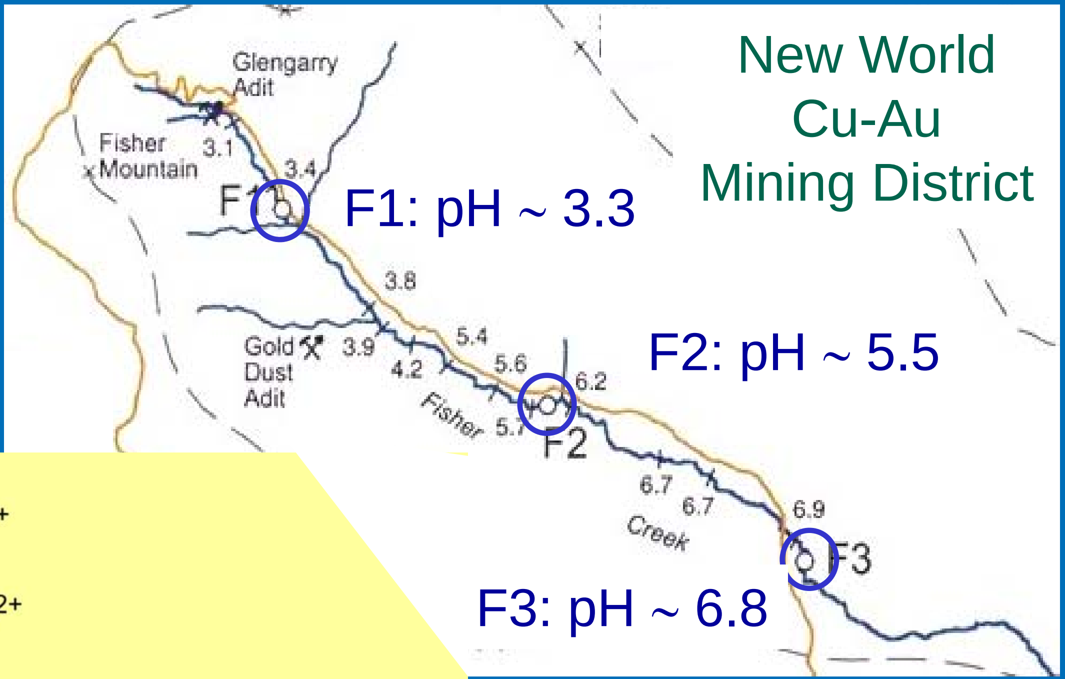
$$Fe^{3+} + H_2O + h\upsilon \rightarrow Fe^{2+} + H^+ + OH^\bullet$$

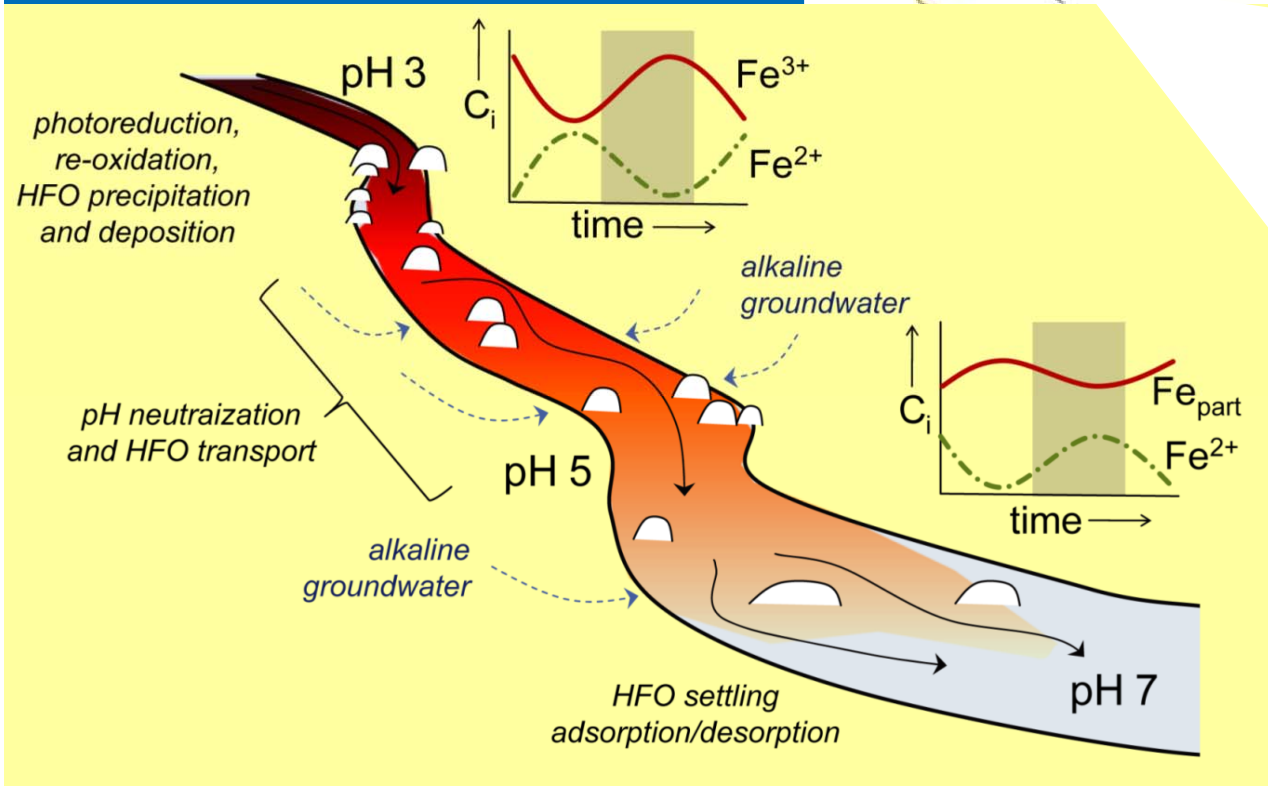
Rio Tinto, Spain

- Light can reduce Fe(III) in both dissolved and solid forms
- Less important at pH > 6
- ($h\upsilon$ = photons)

USGS

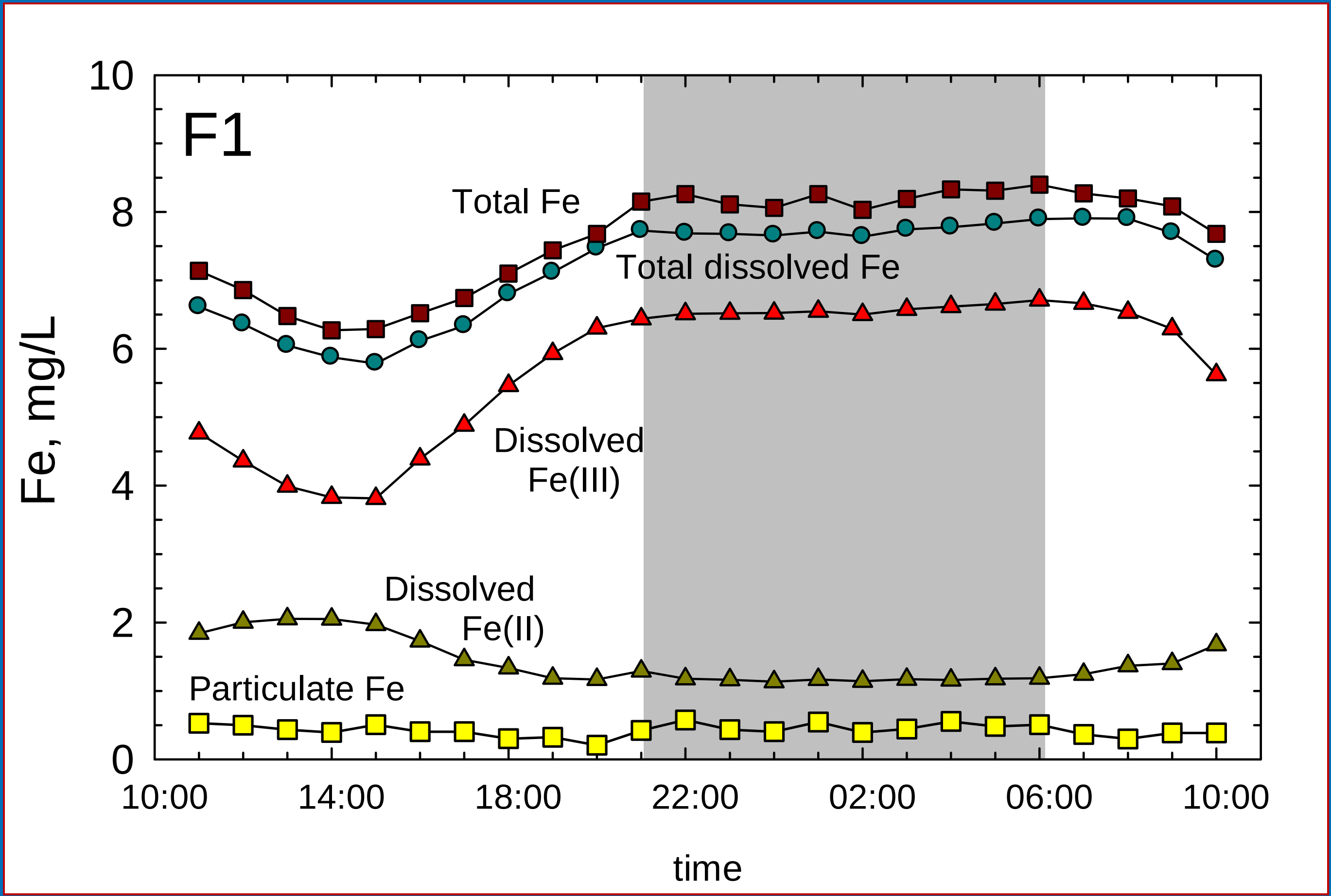
Fe Chemistry along a pH Gradient

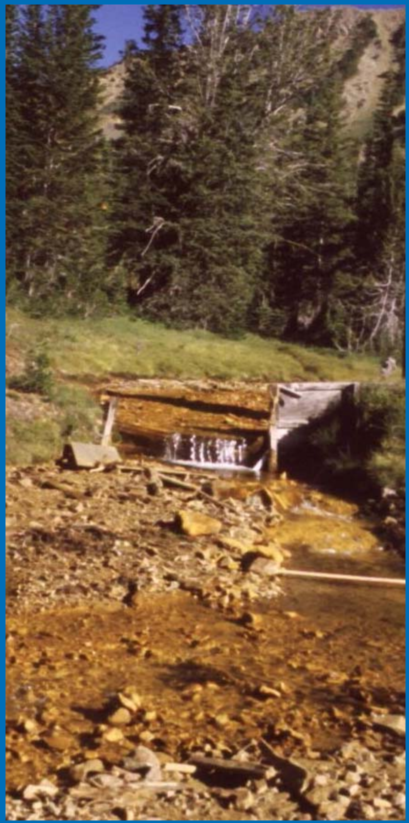




Fisher Creek, Montana
(Gammons et al., 2005a)

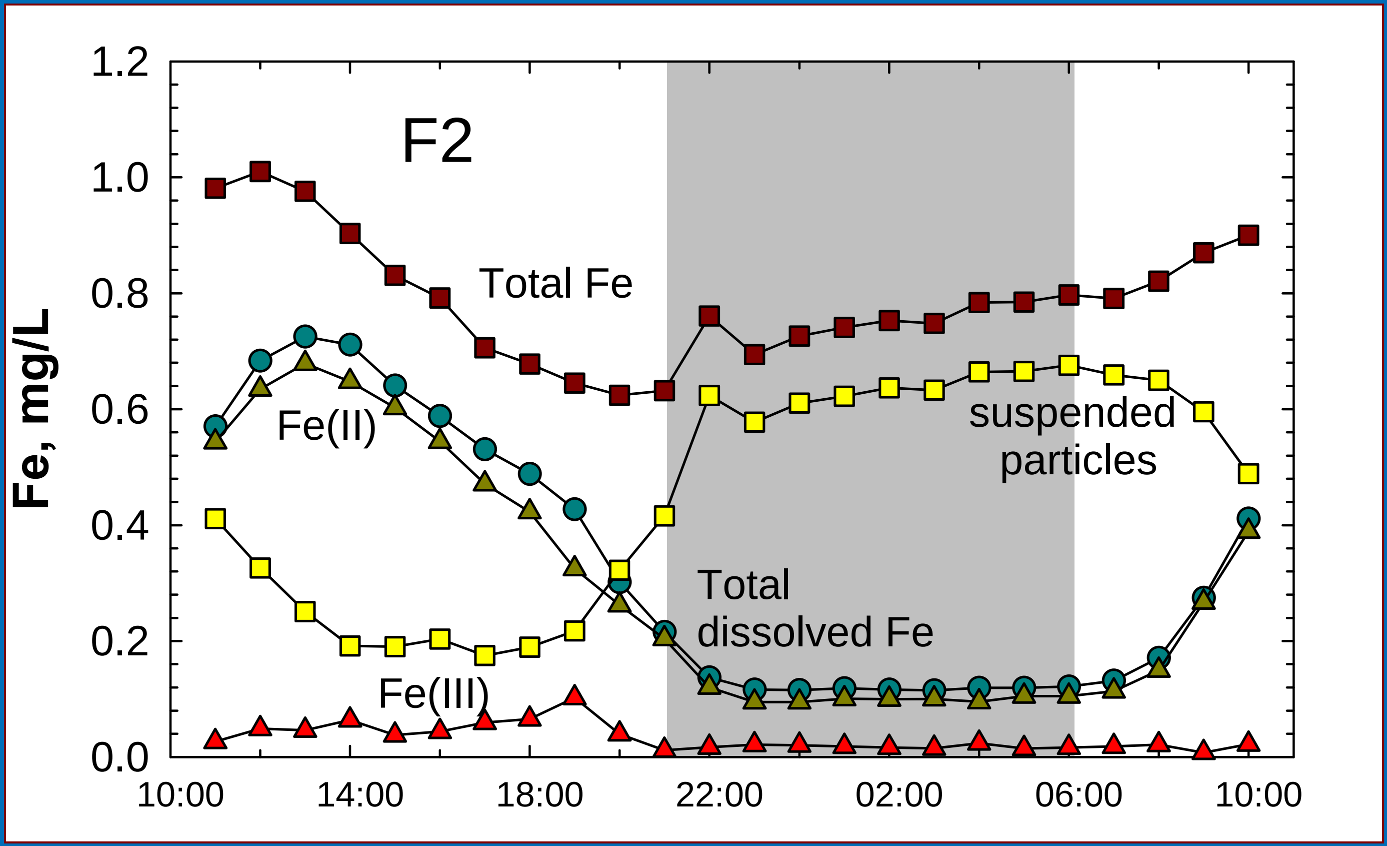
Fisher Creek F1 site: pH ~ 3.3





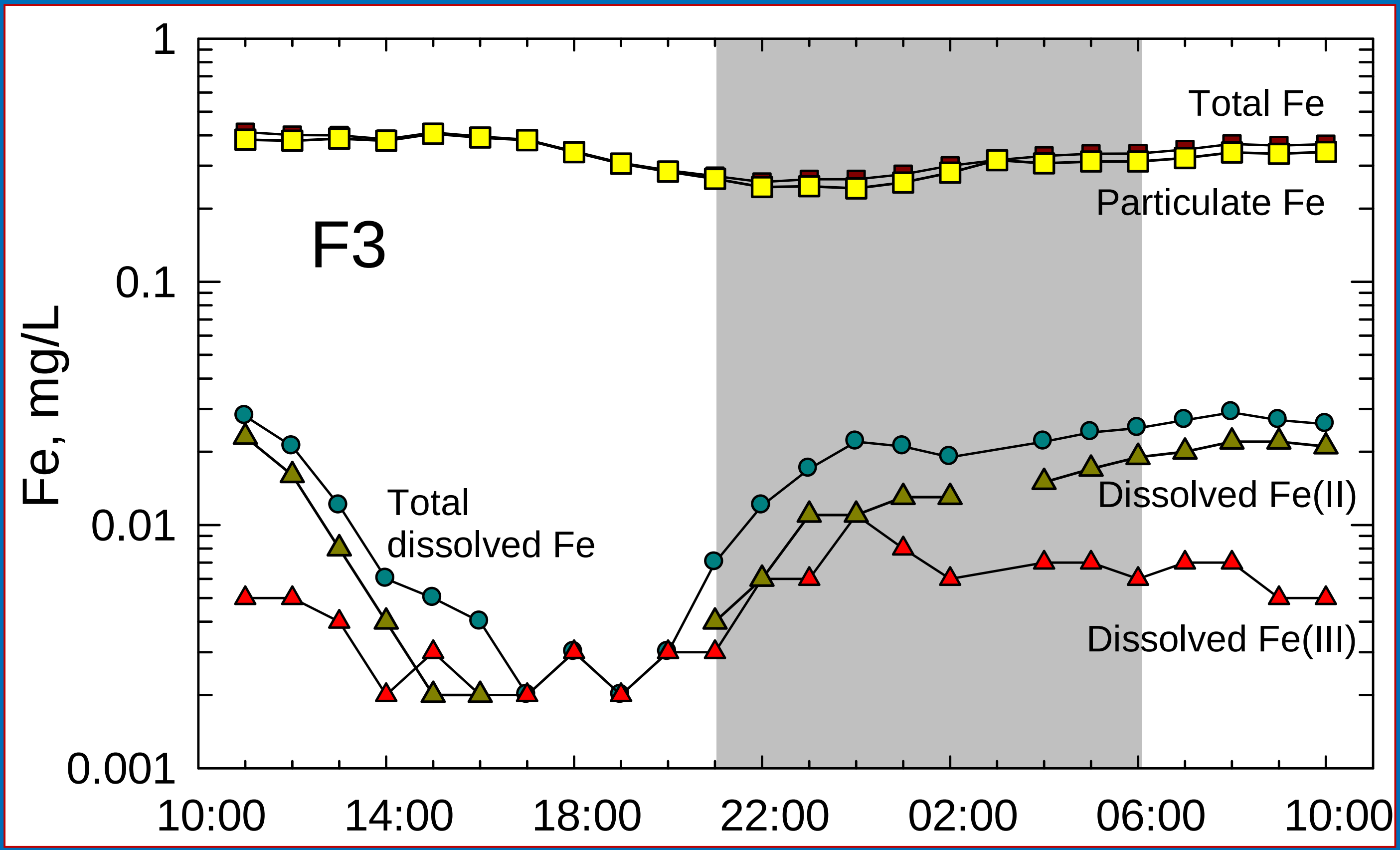
- Daytime decrease in total Fe (solubility of Fe ↓ as T ↑)
- Daytime photoreduction of Fe(III) to Fe(II)

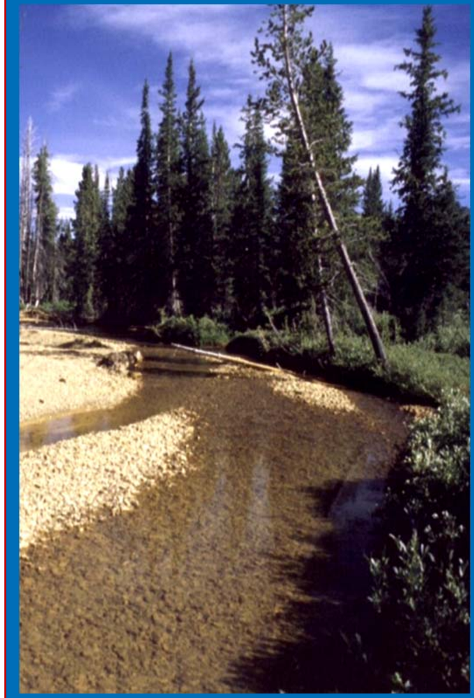
Fisher Creek F2 site: pH ~ 5.5



- Photoreduction of HFO causes daytime increase in Fe(II) and total dissolved Fe concentrations
- Fe mainly dissolved during day, particulate at night

Fisher Creek F3 site: pH ~ 6.8



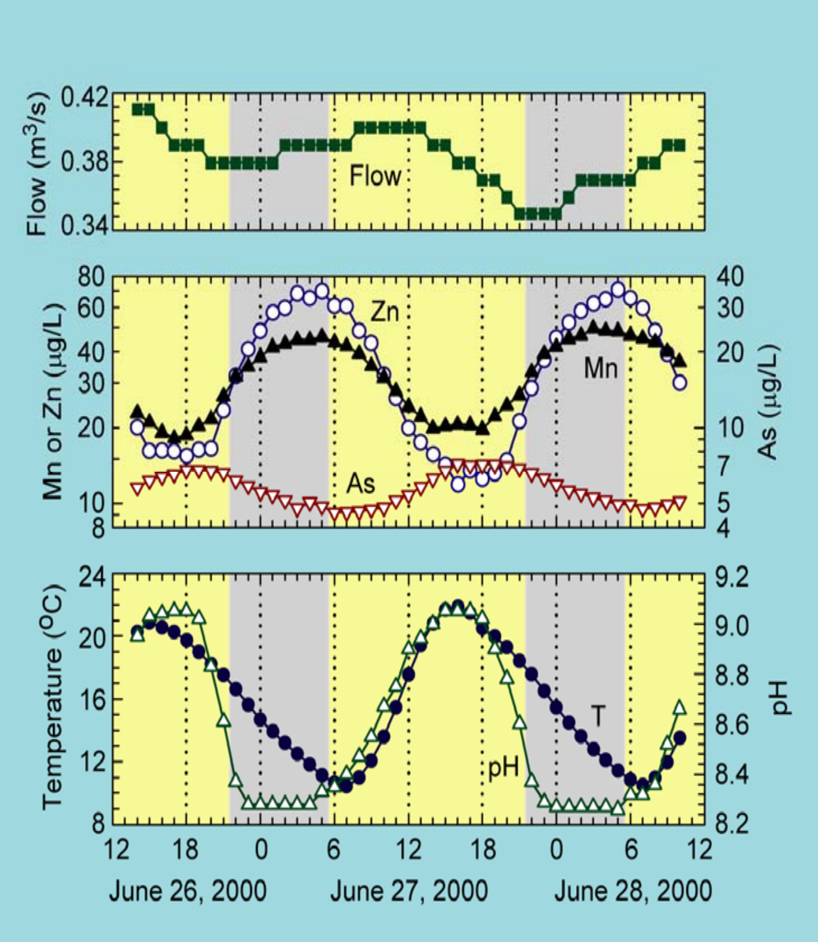


- No evidence of Fe(III) photoreduction
- Night-time increase in Fe(II) and total dissolved Fe mainly due to temperature-dependent sorption

Conclusions – Diel Cycling

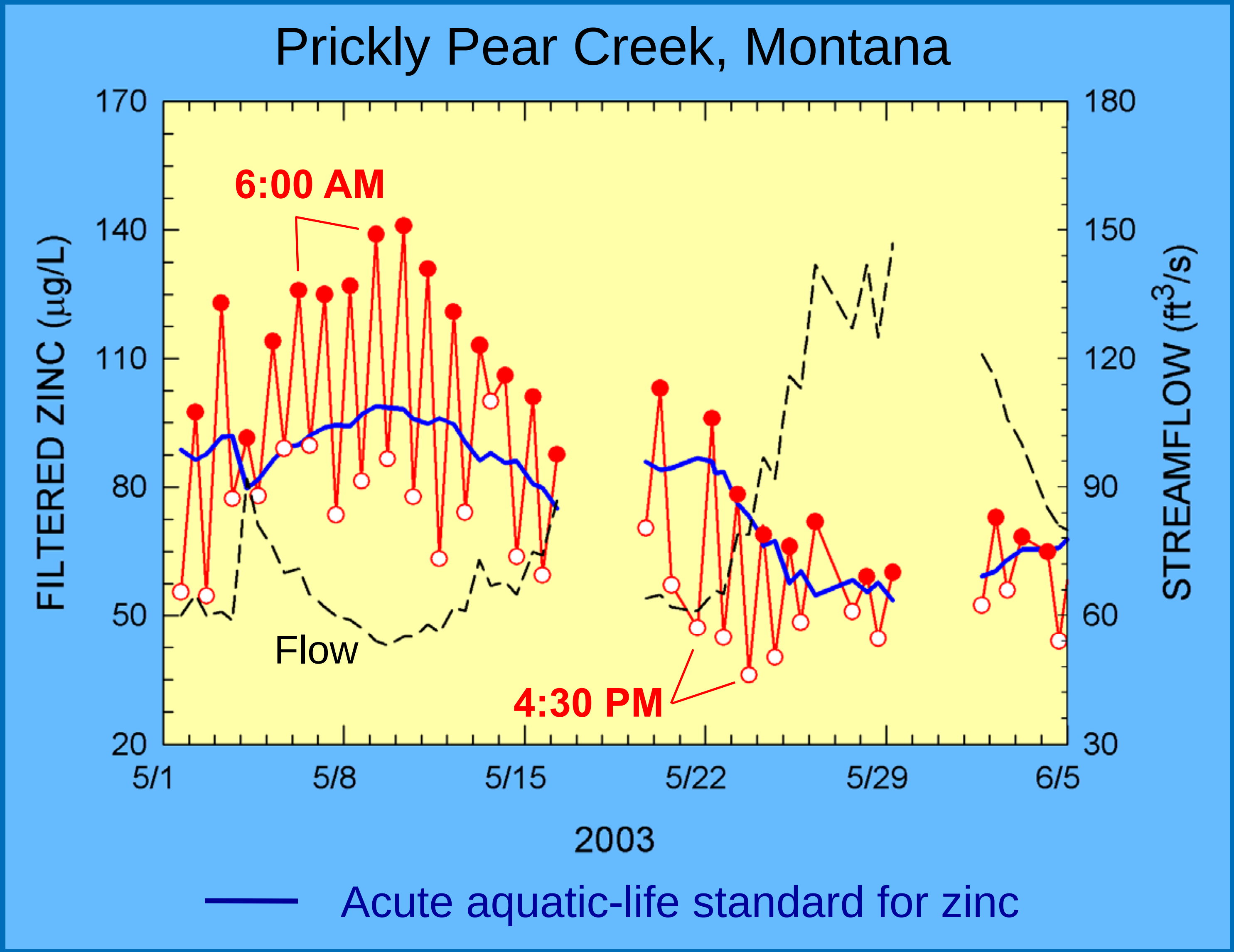
- **<u>Parameters and constituents:</u>**
  - **Streamflow**
  - **pH**
  - **Temperature**
  - **Dissolved oxygen**
  - **Trace elements**
  - **Nutrients**
  - **Hardness and alkalinity**
  - **Suspended particles**

- *Diel variations must be considered when collecting or interpreting water-quality data!*



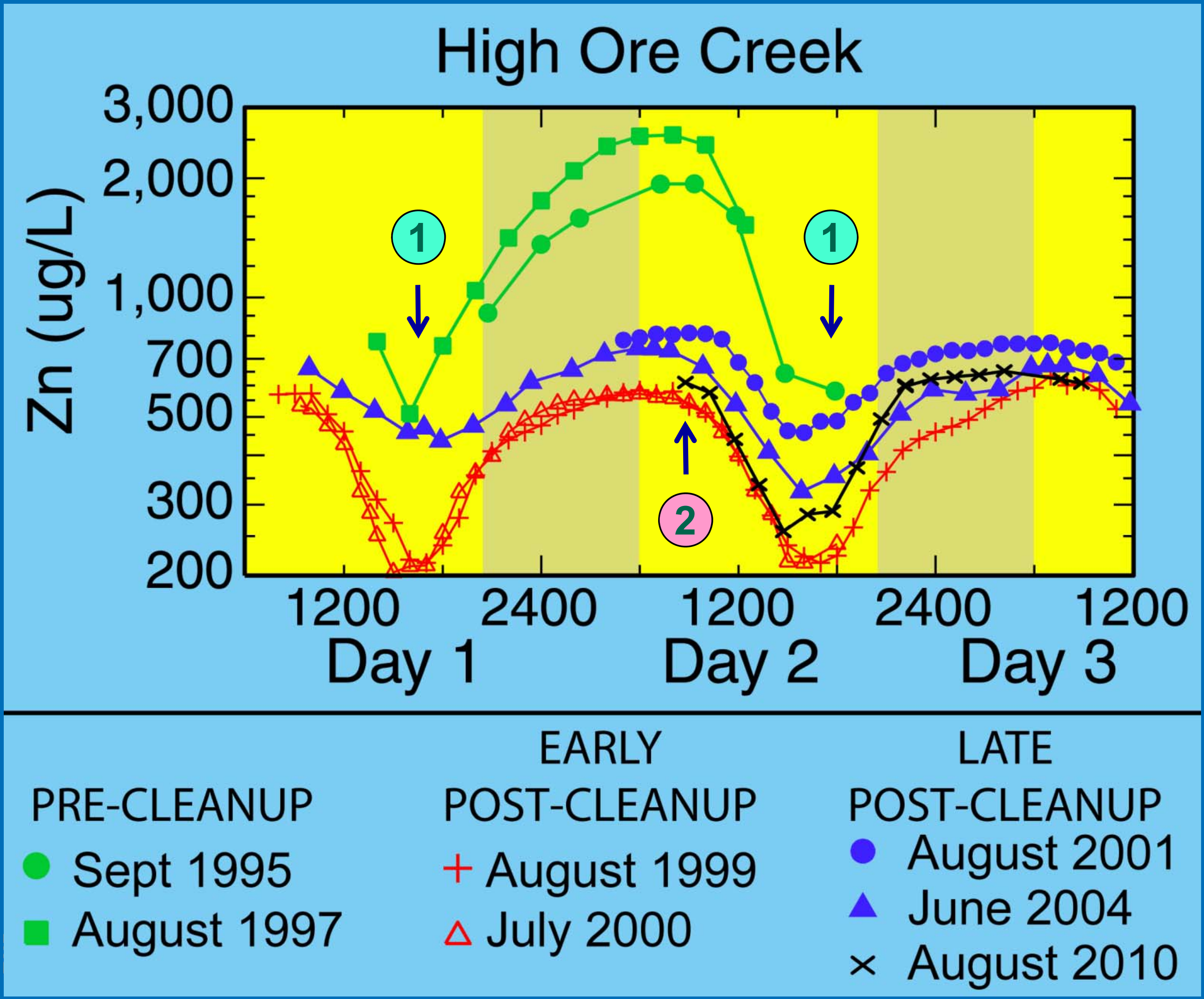
USGS

Implications: Time of Sampling Important!



USGS

Implications: Time of Sampling Important!

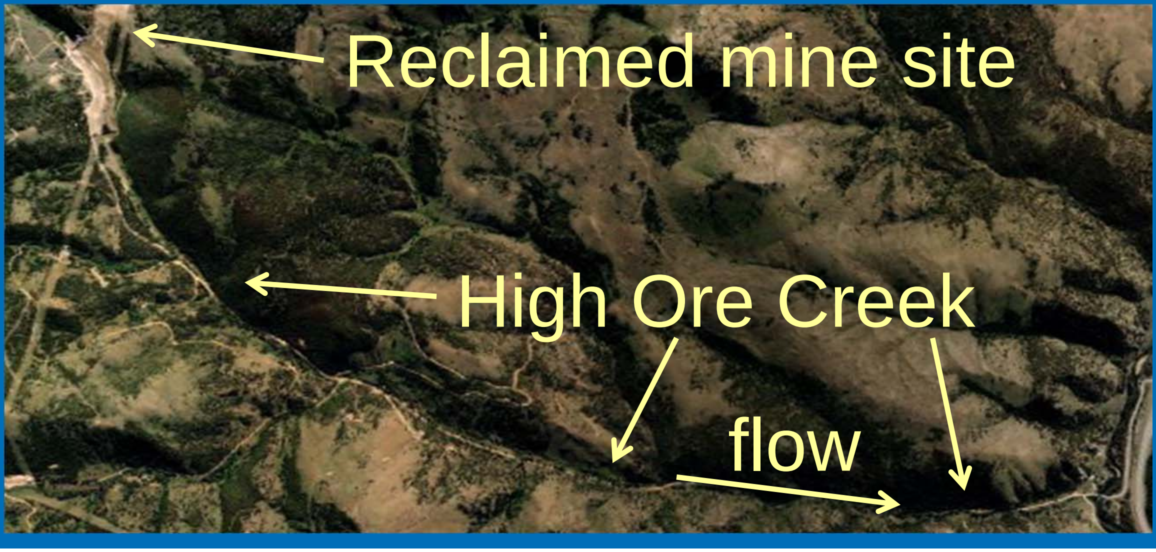


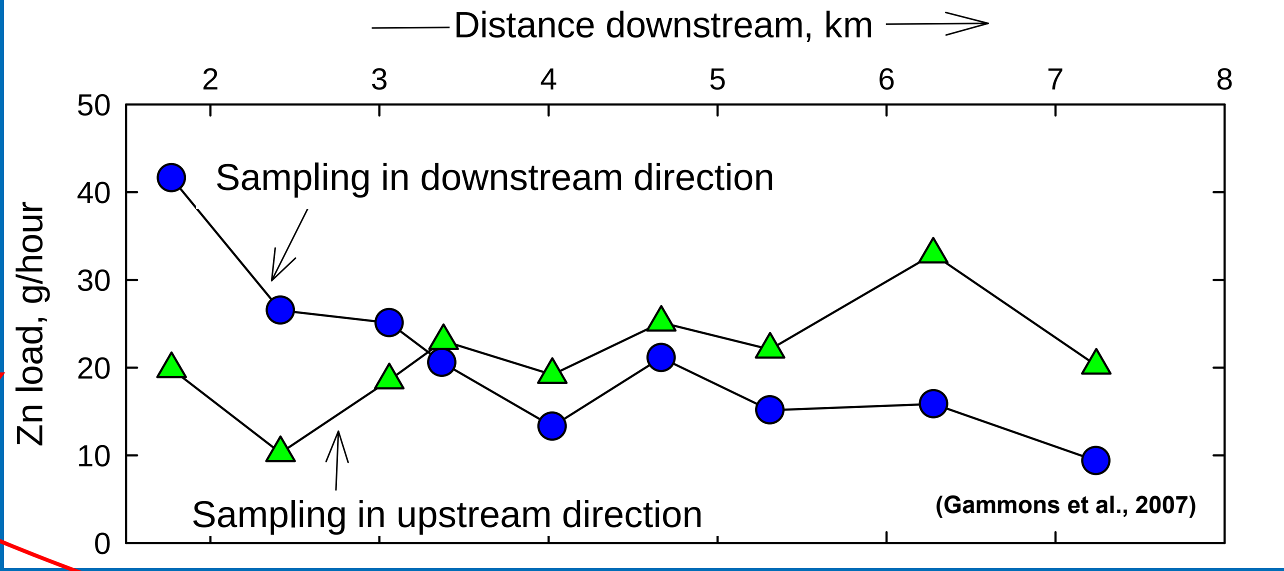
Sampling time:

1 Afternoon

2 Morning

Implications for Synoptic Sampling: Example 1

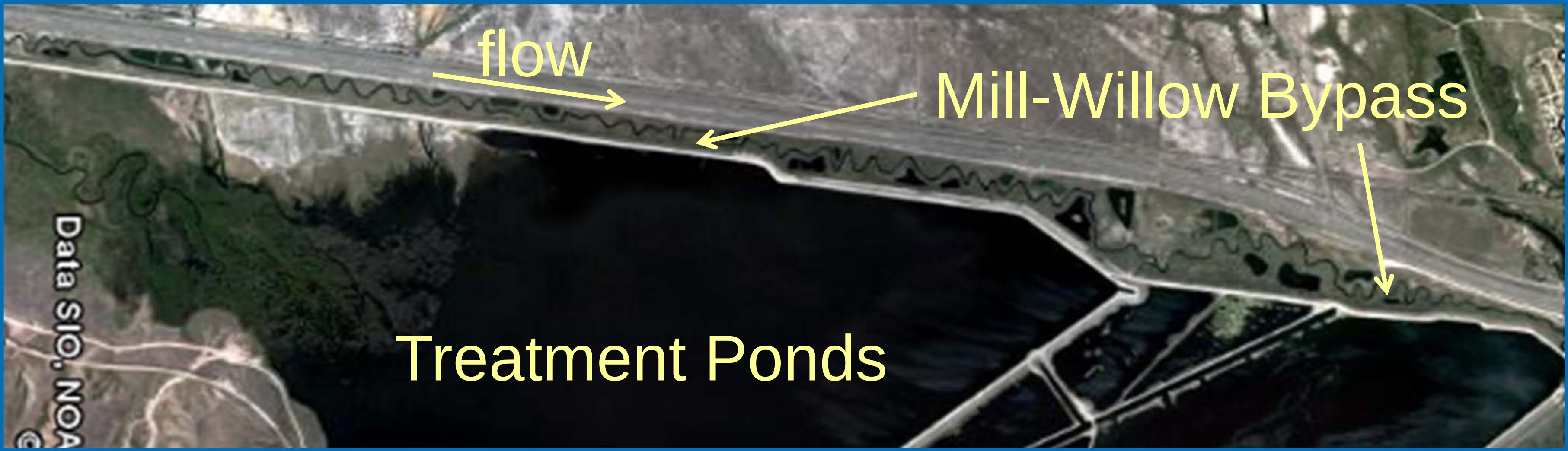


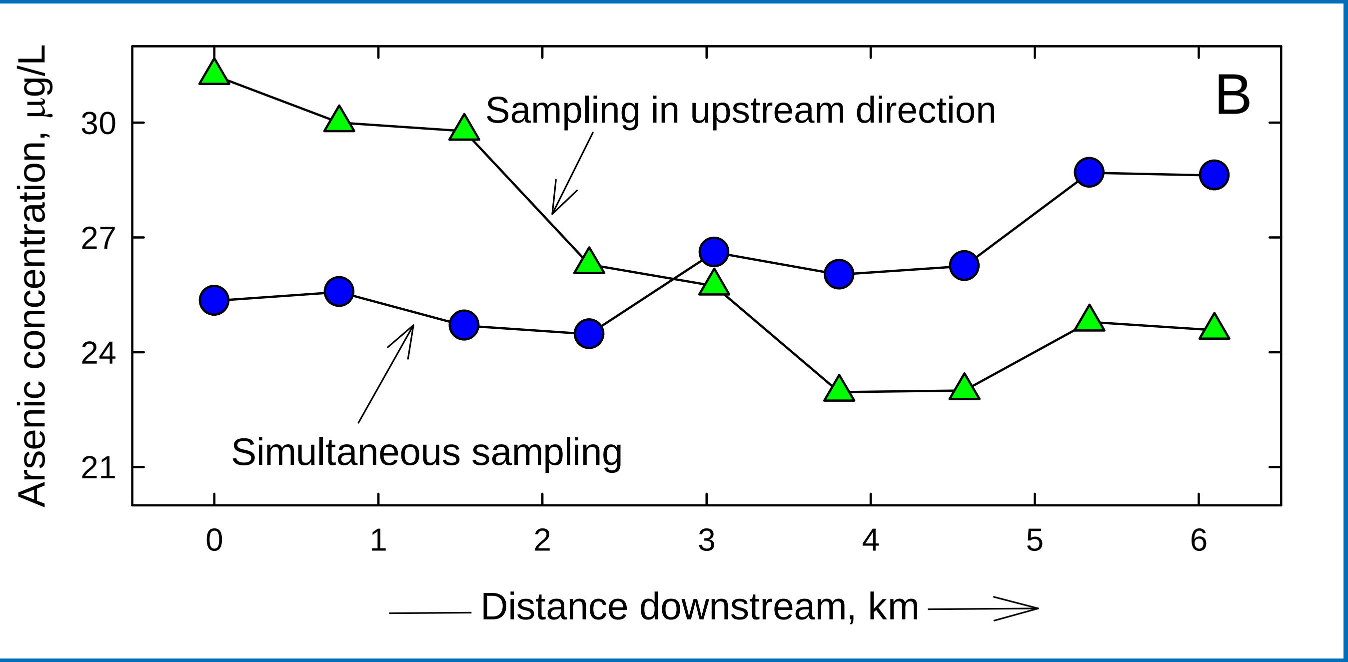


Load = Concentration x Flow

USGS

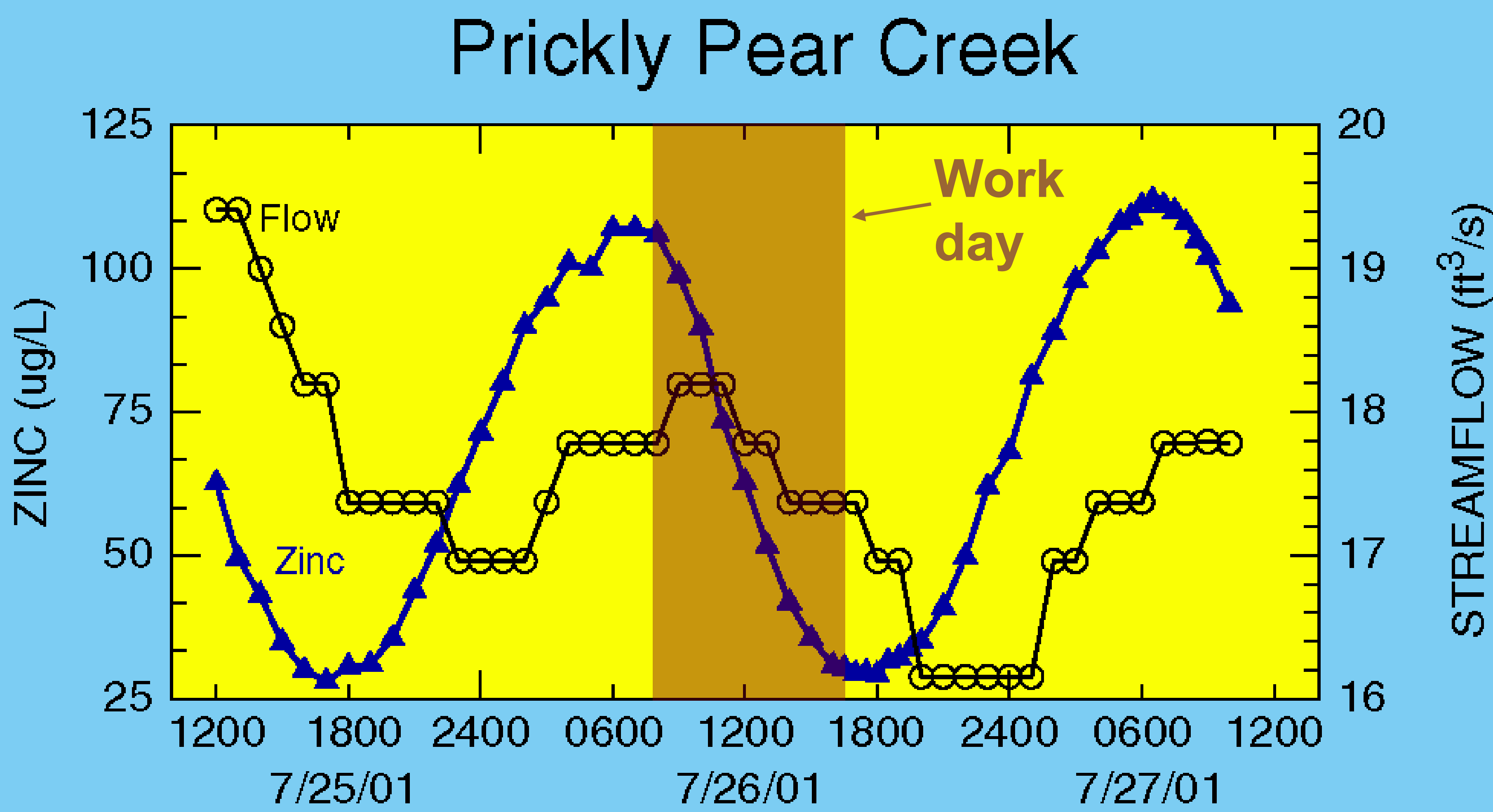
Implications for Synoptic Sampling: Example 2



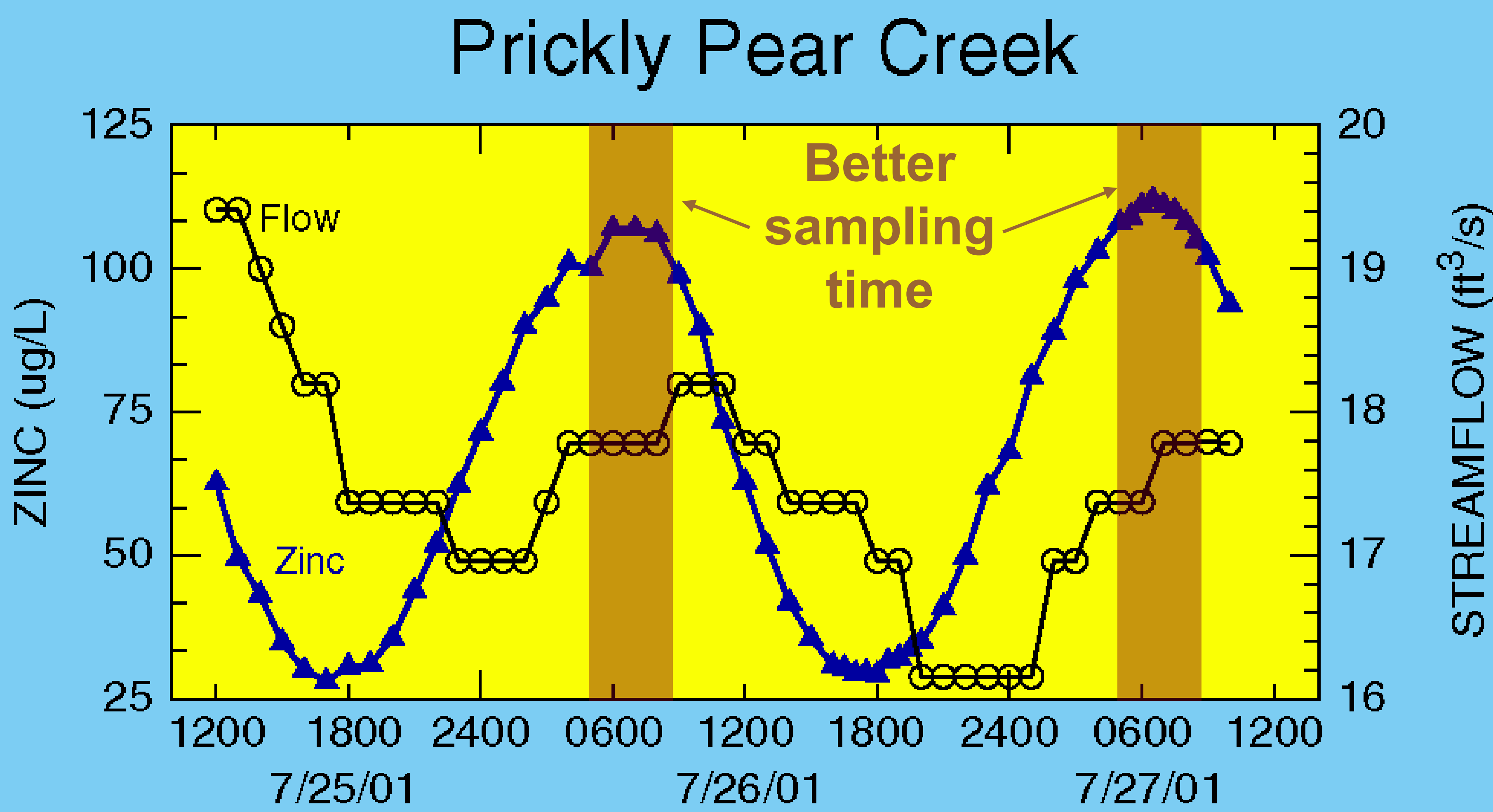


USGS

Implications for Synoptic Sampling: Example 3

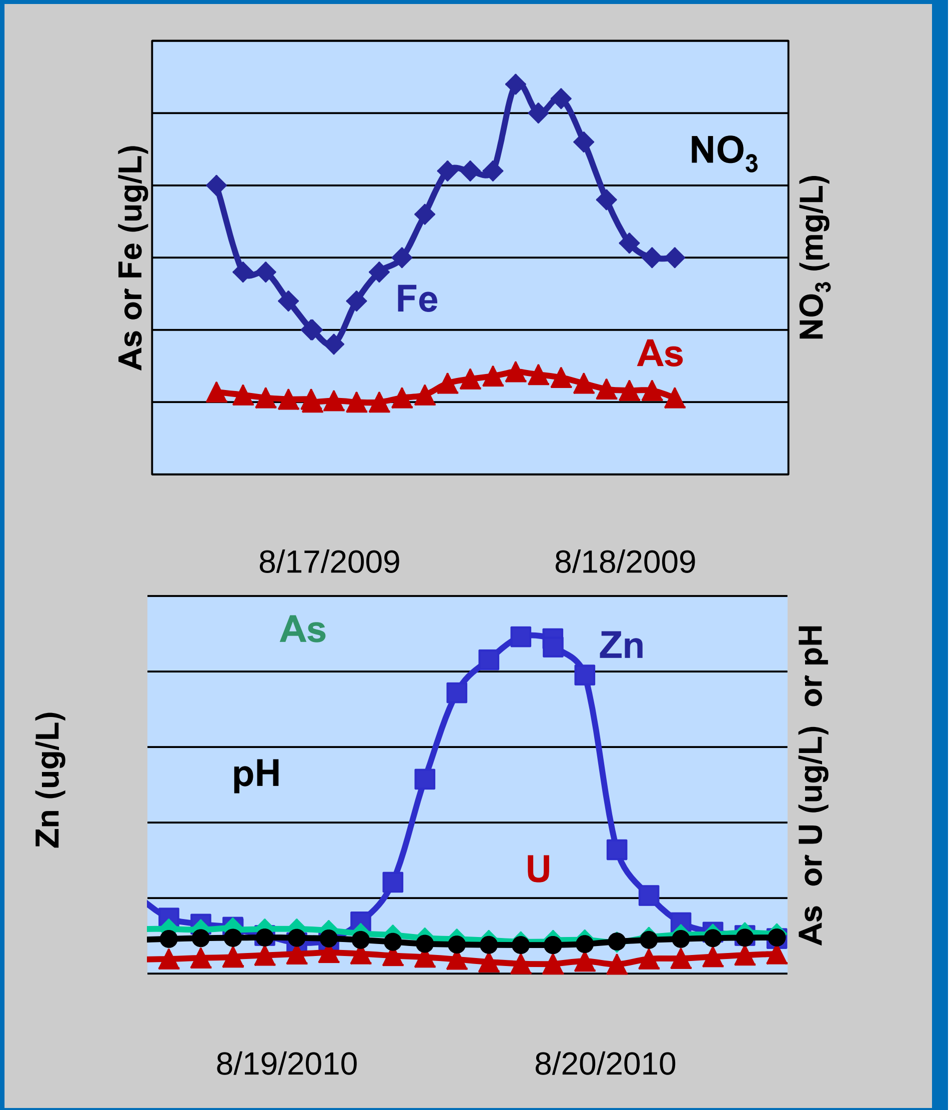


Implications for Synoptic Sampling: Example 3



Sampling Strategies

- Chronic standards
  - Sample at equal time intervals to obtain 4-day mean
- Acute standards
  - Pick sample time to coincide with daily maximum
- Temporal or spatial analysis
  - Always sample at same time or collect 24-h samples
- Comparison of loads (temporally or spatially)
  - Collect samples and measure flows over 24 hours



Silver Bow Creek, Montana

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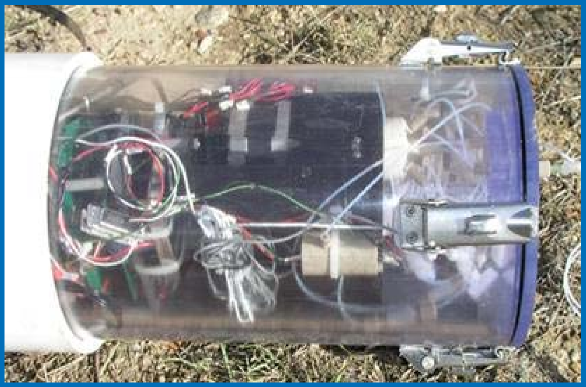
Continuous Collection Methods

- Electrometric & optical sensors (pH, DO, SC, T, turbidity, NO_3, chlorophyll, fluorescence, CDOM)
- In-situ analyzers that use bench-chemistry methods (NO_3, SiO_2, Cl, P, …)
- Lab on the streambank (GC/MS, metals, …)
- Surrogates (e.g., measure turbidity to quantify bacteria)
- Automated samplers

Multi-sensor sonde

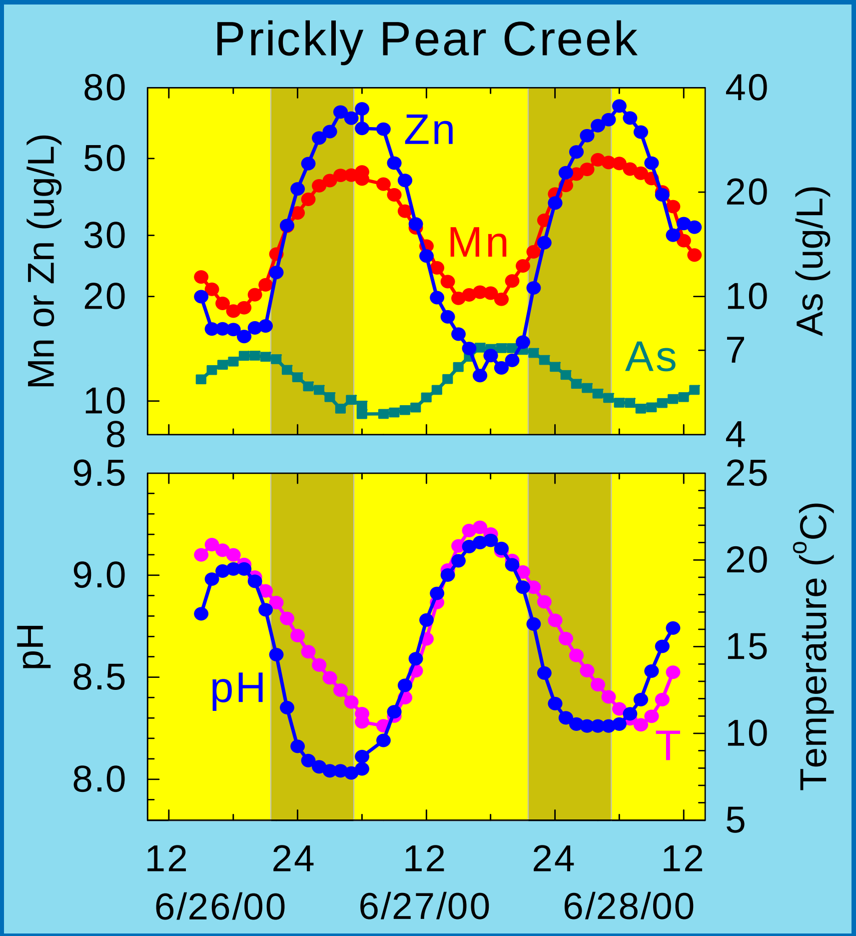


In-situ analyzer

Water-Quality Criteria and Monitoring

Environmental protection may be most effective when:

- Criteria are set with true variability and toxicity in mind
- Criteria are set with monitoring practicality in mind



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Water-Quality Criteria and Monitoring: Temperature

Criteria:

Maximum daily maximum
Maximum weekly maximum
Maximum daily average
Maximum weekly average

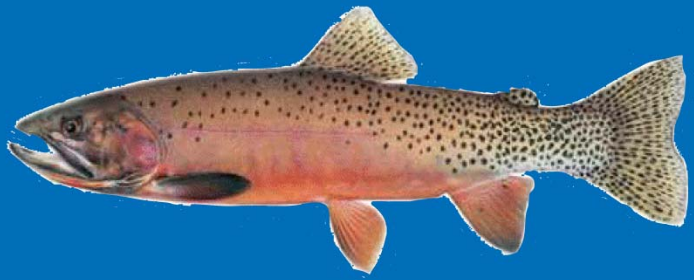
Monitoring:

Hobos, Tidbits, data sondes

Easy calibration, accurate, no drift

Conclusion:

Monitoring capability is out in front of criteria



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Water-Quality Criteria and Monitoring: Dissolved Oxygen

Criteria:

Minimum
7-day average minimum
30-day average

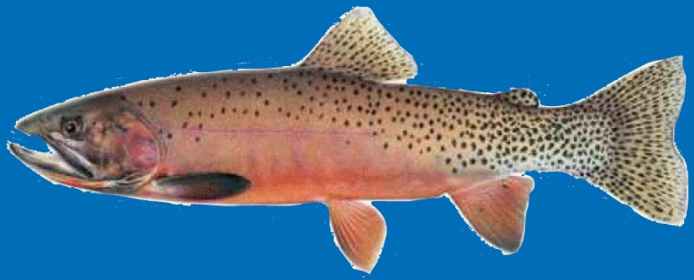
Monitoring:

Data sondes

Need periodic calibration and maintenance to offset drift and fouling

Conclusion:

Monitoring capability has caught up with criteria



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Water-Quality Criteria and Monitoring: Metals

Criteria:

Acute standard:
1-hour average concentration

Chronic standard:
4-day average concentration

…. not to be exceeded more than once in three years

Monitoring:

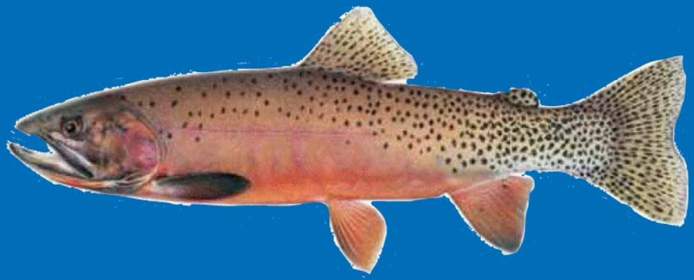
Site visits needed

Automatic samplers require attention in the field but may let you sleep

Diel variability difficult and expensive to address

Conclusion:

Criteria are out in front of monitoring. A more practical expression of criteria may be needed.



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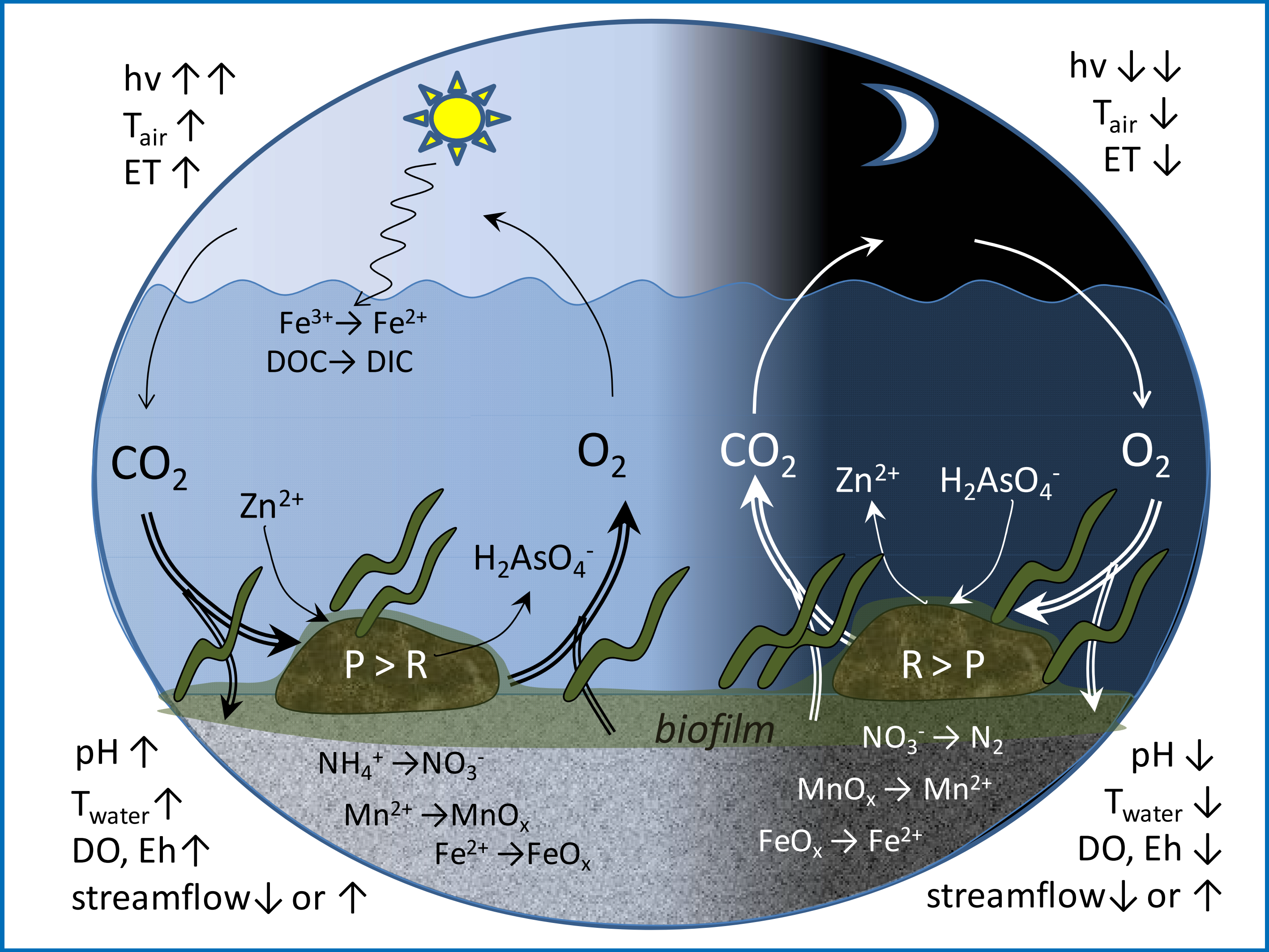
Questions?



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