NORTHWEST ENVIRONMENTAL ADVOCATES

December 16, 2021

Susan Braley
Washington Department of Ecology
P.O. Box 47696
Olympia, WA 98504-7696 *submitted via on-line portal*

Re: **Proposed Freshwater Dissolved Oxygen and Fine Sediment Criteria (WAC 173-201A)**

Dear Ms. Braley:

These comments pertain to Washington Department of Ecology's proposed changes including: (1) adding definitions to WAC 173-201A-020; (2) amending WAC 173-201A-200(1)(d), aquatic life dissolved oxygen criteria for fresh water; and (3) adding subsection WAC 173-201A-200(1)(h), aquatic life fine sediment narrative criterion. These comments also pertain to the following court-ordered requirement: "If the proposed rule is a narrative criterion, Washington will concurrently issue draft guidance regarding how it will interpret and apply its fine sediment criterion, including, but not limited to, its use in establishing Washington's CWA section 303(d) list[.]" Stipulated Order of Dismissal in *NWEA v. EPA*, No. C14-196 RSM, (October 18, 2018), ¶ 2.b.

I. LEGAL BACKGROUND

In 1972, Congress adopted amendments to the Clean Water Act ("CWA") in an effort "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." 33 U.S.C. § 1251(a). The CWA establishes an "interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife." *Id*. at § 1251(a)(2). To those ends, the CWA requires states to develop water quality standards that establish, and then protect, the desired conditions of each waterway within the state's regulatory jurisdiction. *Id*. at § 1313(a). Water quality standards "serve both as a description of the desired water quality for particular waterbodies and as a means of ensuring that such quality is attained and maintained." 64 Fed. Reg. 37,073, 37,074 (July 9, 1999); 40 C.F.R § 131.2. They are the benchmarks by which the quality of waterbodies is measured: waterbodies that do not meet these benchmarks are deemed "water quality limited" and placed on the CWA § 303(d) list. States must develop total maximum daily loads ("TMDLs") for all such 303(d)-listed waters to establish the scientific basis to clean the waters and bring them back into compliance. 33 U.S.C. § 1313(d)(1)(C).

Water quality standards must include three elements: (1) one or more designated "uses" of a waterway, such as swimming or fish propagation; (2) numeric and narrative "criteria" specifying the water quality conditions necessary to protect the designated uses; and (3) an antidegradation policy ensuring continued protection of any uses that have existed since 1975 and maintenance and protection of high quality waters, along with methods to implement the antidegradation policy. *Id.* at §§ 1313(c)(2), 1313(d)(4)(B); 40 C.F.R. Part 131, Subpart B. Implementation methods must be identified as part of the policy's adoption. 40 C.F.R. § 131.12.

States must review and revise their water quality standards at least every three years, in a process called "Triennial Review," thereafter submitting all new and revised standards to EPA for review and action. 33 U.S.C. § 1313(c)(1), (3); 40 C.F.R. § 131.20(c). EPA must review the submitted standards and determine if they meet CWA requirements. 33 U.S.C. § 1313(c)(3); 40 C.F.R. §§ 131.5, 131.13, 131.21(b). A state-developed water quality standard, including any policies affecting those standards, does not become effective until it receives EPA approval. 40 C.F.R. § 131.21(c). When EPA's approval of state water quality standards could have an adverse effect on threatened or endangered species, EPA must consult with U.S. Fish & Wildlife Service ("FWS") and National Marine Fisheries Service ("NMFS") (together "Services"), pursuant to the Endangered Species Act ("ESA"). 16 U.S.C. § 1536(a)(2), 50 C.F.R. § 402.14.

Once approved by EPA, water quality standards serve as the regulatory basis for the establishment of water quality-based controls. For point sources of pollution, EPA retains direct control—which states, such as Washington, may be authorized to carry out—to enforce effluent limitations through the National Pollutant Discharge Elimination System ("NPDES") permitting program. 33 U.S.C. §§ 1311(a), 1342. Congress did not establish an analogous federal permitting scheme for nonpoint source pollution, such as pollution from timber harvesting and agriculture. Instead, Congress assigned states the task of implementing water quality standards for nonpoint sources, with oversight, guidance, and funding from EPA. *See, e.g.*, 33 U.S.C. §§ 1288, 1313, 1329. "[S]tates are required to set water quality standards for all waters within their boundaries regardless of the sources of the pollution entering waters." *Pronsolino v. Nastri*, 291 F.3d 1123, 1127 (9th Cir. 2002) (emphasis in original).

Numeric water quality criteria are central to ensuring protection of designated uses. 33 U.S.C. § 1313(c)(2)(A); 40 C.F.R. § 131.3(b). Criteria "must be based on sound scientific rationale and must . . . protect the designated use." 40 C.F.R. § 131.11(a). Importantly, criteria "shall support the most sensitive use" of the waterbody. *Id.*

II. ECOLOGY'S PROPOSED FINE SEDIMENT NARRATIVE CRITERION

Ecology has proposed a fine sediment narrative criterion that reads as follows:

> (h)(i) Aquatic life fine sediment criterion. The following narrative criterion

applies to all existing and designated uses for fresh water:

(ii) Water bodies shall not contain fine sediment (<2 mm) from anthropogenic sources at levels that cause adverse effects on aquatic life, their reproduction, or habitat. When reference sites are used, sediment conditions shall be compared to sites that represent least disturbed conditions of a neighboring or similar water body.

Proposed WAC 173-201A-200(1)(h). One primary problem with this proposed criterion is in the last sentence in which it refers to the use of "least disturbed conditions" of reference sites. The concept of "least disturbed" implies, correctly, that reference sites are generally somewhat disturbed by anthropogenic activity. The mandatory ("shall be compared") use of least disturbed conditions as a method of applying a criterion that prohibits ("shall not contain") anthropogenic sources of fine sediment is both illogical and inconsistent. The phrase "shall be compared" is just a process but it does not establish an explicit benchmark or rule; likely Ecology means that the information from the reference sites will inform its decision about whether anthropogenic sources have contribute fine sediment. For these reasons, we propose the following language as a replacement:

> <u>When reference sites are used, benchmark natural sediment conditions shall be determined by reference to measured conditions at sites and within watersheds that represent least disturbed conditions, selected within neighboring or comparable water bodies, and screened to assure that the reference site conditions do not reflect temporary fine sediment increases associated with infrequent natural events, or sustained elevation of fine sediment from past human disturbances.</u>

For an explanation of the need for this additional language regarding the need for *screening* reference site conditions, please see the comments attached by Dr. Christopher Frissell.

Additionally, for the reasons explained in Dr. Frissell's comments, the following language should also be added: "<u>All methods of evaluating the impacts of fine sediment shall be demonstrated to be reliable indicators of salmonid-egg-to-fry survival</u>." The ambiguity inherent in the narrative criterion combined with the "weight of evidence" approach Ecology includes in its draft and incomplete implementation guidance for this narrative criterion demonstrates Ecology's clear intention to allow the use of fine sediment measurements that are *not* reliable for the purpose of assessing the effect of fine sediment on the egg-to-fry life cycle stages of salmonids, and to allow those measurements to override others that do provide an indication of unacceptable fine sediment. The example of the weight of evidence approach offered by Ecology fails to weigh evidence according to its scientific veracity and the reliability of inference that can be drawn from it. Thus, it implies that several categories of poor data or indicators that only weakly relate to salmonid-egg-to-fry survival could be used to override one or more categories of far more inherently scientifically reliable data or indicators of well-established consequence to salmonid survival. For this reason, it is essential that the narrative criterion explicitly prohibit the use of fine sediment measures that are not sensitive to the impacts on the very existing and designated uses the narrative criterion seeks to protect.

The narrative criterion refers to prohibiting "levels that cause adverse effects." Ecology is hiding the ball here. What are these levels that cause adverse effects? If there is one level, why are there multiple levels? Assuming that Ecology does not know what these levels are, or it would propose numeric criteria, the narrative criterion must link the prohibition on such adverse effects to the methods set out in the guidance. For this reason, we propose the addition of a new subsection (iv) to read "All methods of evaluating the adverse effects of fine sediment shall be demonstrated to be reliable indicators of salmonid-egg-to-fry survival." In order to make the rule more clear, we suggest moving the discussion of reference sites to its own subsection.

Finally, we propose that Ecology explicitly address the greater need for protection of the most sensitive designated uses, namely those threatened and endangered species whose population numbers are on a downward trend. We propose that the following language be added to modify the protection of the beneficial uses: "<u>taking into account the population status of threatened and endangered species</u>." The currently precipitously small population sizes for these species amplifies the harmful effects that fine sediment has on the species' remaining populations and critical habitat. As Ecology is well aware, Puget Sound Chinook salmon continue to be in significant decline and are today at greater risk of extinction than when the species was first listed. The total Puget Sound Chinook run size in 2021, including both hatchery and wild fish but not including spring Chinook, is down 11 percent from the 2020 forecast of 233,000 fish and two percent below the recent 10-year average of that run. The most recent 10-year average for wild Puget Sound Chinook is 28 percent below the 10-year average for this species in 1999, when it was first listed as threatened pursuant to the federal Endangered Species Act. *See, e.g.,* Washington Department of Fish and Wildlife, *Chinook Historical Run Size—Puget Sound*, *available at* https://wdfw.wa.gov/fishing/management/puget-sound-management-plan#status (last visited December 15, 2021).

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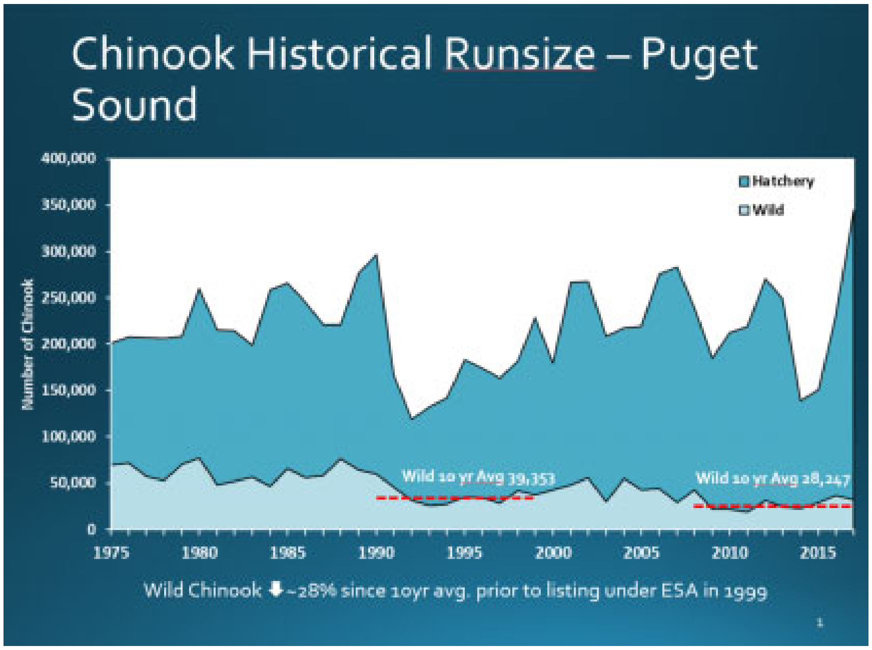

Puget Sound Chinook are in crisis with a future status predicted by the Washington Department of Fish and Wildlife to be less than 25 percent of the recovery goal. Washington Department of Fish and Wildlife, *Status and Trends Analysis of Adult Abundance Data, Prepared in Support of Governor's Salmon Recovery Office 2020 State of Salmon in Watersheds Report* (January 31, 2021), *available at* https://data.wa.gov/Natural-Resources-Environment/FINAL-WDFW-Status-and-Trends-Analysis-Report-Packa/7ir3-4v4j (last visited December 15, 2021) at 16. Similarly, Lake Ozette sockeye, Snake River spring/summer Chinook, Puget Sound steelhead, and Upper Columbia spring Chinook, are in crisis, with populations projected to reach less than 25 percent of the recovery goal in the near future. *Id.* Other populations are deemed to "not keeping pace" include Lower Columbia River coho, Lower Columbia River Chinook, Upper Columbia River steelhead, and Middle Columbia River steelhead. *Id*. Bull trout populations are also decreasing. U.S. Fish and Wildlife Service, *Bull Trout (Salvenlinus confluentus) 5- Year Review: Summary and Evaluation* (2008) at 44 (identifying a "decreasing trend" in bull trout abundance).

Southern Resident killer whales continue to be significantly affected by pollution problems including those that affect their essential prey, Chinook salmon. Today there are only 73 Southern Resident killer whales, down from 78 individuals in 2016 when NMFS completed its last five-year review. NOAA, Southern Resident Killer Whale (*Orcinus orca*) *available at* https://www.fisheries.noaa.gov/west-coast/endangered-species-conservation/southern-resident-killer-whale-orcinus-orca (last visited December 15, 2021) (identifying Southern Resident killer whale abundance for 1996); NMFS, *Southern Resident Killer Whales (Orcinus orca) 5-Year*

Review: Summary and Evaluation (December 2016) at 16 (identifying abundance for fall 2016); Orca Network, Births and Deaths, *available at* https://www.orcanetwork.org/Main/index.php?categories_file=Births%20and%20Deaths (last visited December 15, 2021) (reporting Southern Resident killer whale abundance as 73 individuals as of September 20, 2021). As the primary food source for Southern Resident killer whales, the continued decline in Chinook populations directly affect the whale's continued decline. *See* Lacy, R.C., Williams, R., Ashe, E. *et al. Evaluating Anthropogenic Threats to Endangered Killer Whales to Inform Effective Recovery Plans*, 7 *Sci Rep* 14119 (2017), *available at* https://doi.org/10.1038/s41598-017-14471-0.

These species' continued declines make them more vulnerable to the effects of fine sediment. Small populations have disproportionately higher chances of going extinct because environmental and biological forces function differently in these smaller populations and may result in positive feedback loops driving them towards extinction. The forces acting on small populations in "extinction vortices" include increased vulnerability to stochastic impacts, Allee effects on population dynamics, genetic deterioration from inbreeding and genetic drift, increased vulnerability to environmental stressors, such as pollution, and synergistic impacts. *See* Michael *Gilpin and Michael E. Soulé, Minimum Viable Populations: Processes of Species Extinction in CONSERVATION BIOLOGY: THE SCIENCE OF SCARCITY AND DIVERSITY 13-34 (M. E. Soulé ed., 1986*; Barry. W. Brook, Navjot S. Sodhi, and Corey J.A. Bradshaw, *Synergies among extinction drivers under global change*, 23 Trends in Ecology and Evolutionary Biology 453, 455 (2008); Anna-Marie Winter, Andries Richter, and Anne Marie Eikeset, *Implications of Allee effects for fisheries management in a changing climate: evidence from Atlantic cod*, 30 Ecological Applications (2020); Priyanga Amarasekare, *Allee Effects in Metapopulation Dynamics*, 152 The American Naturalist 299 (1998); Marty Kartos, *et. al*, *The crucial role of genome-wide genetic variation in conservation*, 118 Proc. Nat. Acad. Sci. (2021). The continued declines and increasingly low abundances of these species put them at a disproportionately greater risk of extinction than they would be if their populations were abundant, making protection of these designated uses more sensitive than they would be otherwise. Criteria to protect these species must be adjusted accordingly.

Taken as a whole, we propose the following changes in the proposed narrative criterion:

(h)(i) Aquatic life fine sediment criterion. The following narrative criterion applies to all existing and designated uses for fresh water:

(ii) Water bodies shall not contain fine sediment (<2 mm) from anthropogenic sources at levels that cause adverse effects on aquatic life, their reproduction, or habitat, taking into account the population status of threatened and endangered species. ~~When reference sites are used, sediment conditions shall be compared to sites that represent least disturbed conditions of a neighboring or similar water body~~.

(iii) When reference sites are used, benchmark natural sediment conditions shall be determined by reference to measured conditions at sites and within watersheds that represent least disturbed conditions, selected within neighboring or comparable water bodies, and screened to assure that the reference site conditions do not reflect temporary fine sediment increases associated with infrequent natural events, or sustained elevation of fine sediment from past human disturbances.

(iv) All methods of evaluating the adverse effects of fine sediment shall be demonstrated to be reliable indicators of salmonid-egg-to-fry survival.

III. ECOLOGY'S FINE SEDIMENT IMPLEMENTATION GUIDANCE

The draft implementation guidance is, in fact, no guidance at all on how it "will interpret and apply its fine sediment criterion, including, but not limited to, its use in establishing Washington's CWA section 303(d) list[.]" Most obviously, this draft guidance fails to explain how Ecology will use the new narrative criterion for the purposes of CWA section 303(d) assessments as is evidenced by its own statements:

> The addition of a narrative fine sediment criterion *will require the development of a methodology to evaluate when the fine sediment standard is being exceeded*. Ecology will provide guidance on the parameters used to characterize fine sediment in a waterbody. Subsequently, the listing methodology to determine a fine sediment-based impairment will be developed by the water quality program through a public process. Appendix A provides sampling recommendations and approaches for making a determination of an exceedance of fine sediment criteria. The final methodology for assessing fine sediment will be in a revision to Water Quality Program Policy 1-11.

Ecology, *Preliminary Rule Implementation Plan Chapter 173-201A WAC, Water Quality Standards for Surface Waters of the State of Washington Salmon Spawning Habitat Protection Rule* (October 2021) ("Draft Guidance") at 7 (emphasis added); *see also id*. at 31 ("The methods to determine a fine sediment impairment for purposes of the Clean Water Act Section 303(d) *will be finalized* in Ecology's Water Quality Policy 1-11, Chapter 1. However, the following recommendations may be useful in *developing* an approach to determining a fine sediment exceedance.") (emphasis added). Ecology's use of the *future* tense in its draft guidance demonstrates that this is, in fact, no guidance at all on how the narrative criterion will be used for the purpose of 303(d) listings or other regulatory actions.

In addition to the lack of a methodology for using the narrative criterion in 303(d) listing, Ecology's draft implementation guidance also fails to address the other regulatory contexts in which it will need to interpret and apply the criterion, namely the following: (1) in the development of total maximum daily loads under CWA section 303(d); in the establishment of best management practices for nonpoint sources under CWA section 319 and the Coastal Zone Act Reauthorization Amendments ("CZARA"), 16 U.S.C. § 1455b(b)(3) (each state shall provide for the "implementation and revision of management measures" for nonpoint sources "that are necessary to achieve and maintain applicable water quality standards under [CWA] section 1313 of Title 33 and protect designated uses"); and in the issuance of National Pollutant Discharge Elimination System ("NPDES") permits pursuant to CWA section 402. The purported draft guidance does none of these things.

CONCLUSION

Just days ago, Washington Governor Jay Inslee published his *Governor's salmon strategy update; Securing a future for people and salmon in Washington* (December 14, 2021). Under the heading "How we will improve wastewater management to achieve clean water for salmon and people" his strategy update includes "Revise and implement water quality standards to respond to aquatic ecosystem needs." *Id.* at 7. Numerous other provisions in this strategy update reference the need to meet water quality standards, for example that agriculture needs to meet "water quality standards for salmon rearing and spawning." *Id*. at 6. In order for Ecology's water quality standard for fine sediment to fully support the Governor's strategy, the agency must significantly strengthen the narrative criterion as well as fully set out implementation guidance for that narrative.

Sincerely,

Nina Bell

Nina Bell
Executive Director

Attachment: Letter from Dr. Christopher Frissell, Frissell & Raven, to Susan Braley, Ecology, Re: *Comments on Proposed Freshwater Dissolved Oxygen and Fine Sediment Criteria (WAC 173-201A)* (December 16, 2021).

FRISSELL & RAVEN

Hydrobiological & Landscape Sciences

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16 December 2021

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Subject:
Comments on Proposed Freshwater Dissolved Oxygen and Fine Sediment Criteria (WAC 173-201A)

1. Introduction

During 2020-2021 I served on the Science Panel established by Washington Department of Ecology (hereafter "Ecology") to help inform development of the fine sediment and dissolved oxygen criteria. Here I offer comments focusing on three topics; 1) how and whether input of the Science Panel is reflected in the draft narrative criteria and guidance, in particular the Fine Sediment Criteria; 2) certain content of the draft implementation guidance document that ostensibly supports attainment of the narrative criteria for fine sediment, and 3) the scope of science considered during Science Panel review process.

I have some reservations about the rationale employed by Ecology to support the proposed dissolved oxygen standard, but in the context of implementation in the field, I believe those concerns will have relatively minor consequence. Overall, in my opinion the Science panel process for dissolved oxygen was more rigorously and completely conducted that that for fine sediment, hence the outcome for dissolved oxygen is more defensible. Accordingly, I focus my comments herein on the fine sediment criteria.

2. Maintaining or Restoring Beneficial Use of Salmonid Spawning

Ecology defines the beneficial use as providing habitat conditions sufficient to fully support the recovery of threatened, endangered, and declining salmonid populations. Conversely, to provide for clarity and precision in developing protective criteria, a clear and concise definition of impairment is also needed. I suggest that appropriate language would be as follows: *Impairment is any human-caused change in habitat conditions that reduces the capability of threatened endangered, and declining populations to recover to stable, self-sustaining and status sufficiently productive to support commercial and sport fisheries.* The lack of a clear definition of what condition, action, effect, our outcome is to be governed by the standard and guidance lead to troublesome early confusion in the Science Panel proceedings. I believe my wording accurately captures the consensus that developed among the participating scientists as the discussions progressed, although I cannot speak with certainty about all non-Ecology participants. And I would add that it was not at all clear from their verbal input at the Science panel meetings that Ecology-employed participants would agree with my wording.

I would further offer that in my participation in the Science Panel process, I assumed that these criteria for protection and impairment applied broadly to water quality-related actions by Ecology and the state of Washington, including: point source permitting; designation of impaired water bodies; establishing effective targets and implementation plans for TMDLs; and evaluation of effectiveness or "bestness" of so-called best management practices for non-point source pollution control.

With regard to the biological underpinning that determines this specific beneficial use, it is difficult to conceive how density-independent mortality of fish at the egg-to-fry stage, where fine sediment conditions conditions most acutely affect survival, does not impair recovery and productivity of salmonid populations whose status is known to be declining or greatly reduced in abundance. By contrast, density-dependent mortality processes that prevail at other life stages can often be "absorbed" and biologically self- compensated at the population level in several ways (e.g., reduced density of juveniles may increase individual growth and thus per capita survival rates). In support of my point, pleas see Karieva et al. (2000) and Honea et al. (2009), who modeled life-stage specific survival of spring-run Chinook salmon to evaluate the magnitude of net effect of habitat change on whole-life-cycle survival and population trend. Both studies concluded that survival at the egg-to-fry stage generally is the most consequential stage, or is among the two most consequential stages, at which improvement in habitat conditions could increase population productivity and adult population size. Conversely, therefore, degrading spawning habitat conditions can have or has had the largest magnitude of negative life cycle impact.

Although Ecology has not facially disputed the conclusions in the preceding paragraph, at the same time Ecology's draft guidance is premised on assumptions that plainly contradict those conclusions. In particular, Ecology provides no biological or physical rationale to support the existence of a threshold in terms of acceptable or sustainable egg-to-fry survival or mortality. Nevertheless, in its draft guidance document Ecology advances the implicit assumption that deterioration and impairment of existing streambed sediment conditions in spawning habitat when below 20% fines (<2mm diameter) would not equal impairment. I see no logical basis in available science to assume that fine sediment conditions that currently exist below any given threshold metric are not impaired by any increase in sediment. In other words, there is no reliable evidence that a "safe level" of fine sediment exists (at 20% or any other concentration, and certainly not a concentration greater than 10%), nor that a given addition of human-caused fine sediment to any system, regardless of present state, would be free of harm and not cause impairment. See the next section of my comments for more detailed discussion of this concern.

To establish a threshold of safe and acceptable fine sediment conditions in a scientifically defensible way, Ecology would need to determine the prevailing fine sediment conditions in spawning habitat within streams where previously declining or depleted salmonid populations have been shown to have recovered, or at least to have demonstrated a sustained long-term recovery trend (e.g., survival to adult return increasing over at least three fish generations, or at least *ca*. 12-15 years). To my knowledge, and judging by what I have seen in the record of writing and presentations by Ecology, no such analysis has been conducted.

3. Thresholds in the Fine Sediment Relationship to Survival?

A key question pertaining to establishment of a fine sediment standard concerns the oft-assumed existence of some threshold concentration fine sediments, below which egg-to-fry survival is not measurably impaired. In regulatory terms, this equates to the assumption that a “safe level” of fine sediments exists; fine sediment increases are presumed to have no effect on survival until this threshold is breached. For example, this assumption is embraced in Ecology’s Draft Guidance, e.g. on pp. 31-32. The threshold effect assumption is convenient because if true, it provides some rationale for establishing a fine sediment standard that confers regulatory flexibility to allow increases in fine sediment pollution in streams where spawning gravel conditions are currently excellent. That is, where fine sediment concentrations are well below a presumed ”safe” threshold, fine sediment increases could be tolerated, permitted or allocated. However, that level of detail about consequences is seldom voiced to support a threshold-based criterion. Most often, a presumed threshold is simply considered by agencies as a convenient way to dismiss the probability of adverse impacts or injury over a broad sweep of conditions, with the intent of easing or simplifying a regulatory burden.

However, in the present context, as a scientific matter the assumption of a “safe” threshold for fine sediment is wholly untenable. During Science Panel meetings, supported by submitted published material, I contended that data from most available studies do not in fact support the existence of such a threshold, instead indicating a linear or possibly somewhat inflected curvilinear reduction in survival as fine sediments increased, beginning at fine sediment percentages of 10% or less.

Experimental work in Montana on bull trout and westslope cutthroat trout survival to emergence (Weaver and Fraley 1991, and see Reiman and McIntyre 1993 for further interpretation) included enough data points at fine sediment percentages of less than 20% to show a clear linear decrease in survival between 0 and 20 percent fine sediment concentration (Fig.1). In their field experiment employing a wide range of sediment mixtures, Weaver and Fraley (1991) reported that survival to emergence of both species in a 0% fines mixture was significantly greater than survival in all other mixtures with greater fines except 10%, and survival at 10% was significantly greater than survival at mixtures above 30 percent fines. Higher variation in survival in the range of 20-30 percent fines within the study reach was likely a result of groundwater upwelling mitigating the effect of fine sediment colmation in certain sites; however, a handful of higher- survival cases should not be invoked to obscure trend of a clear decline in mean survival rate through the range of increasing fine sediment percentage (Rieman and McIntyre 1993). Note that this study used a larger diameter to classify fine sediments than does Ecology's proposed narrative criteria, but based on Jensen et al. (2009) and other sources, this should have minimal impact on the basic shape of the response curves.

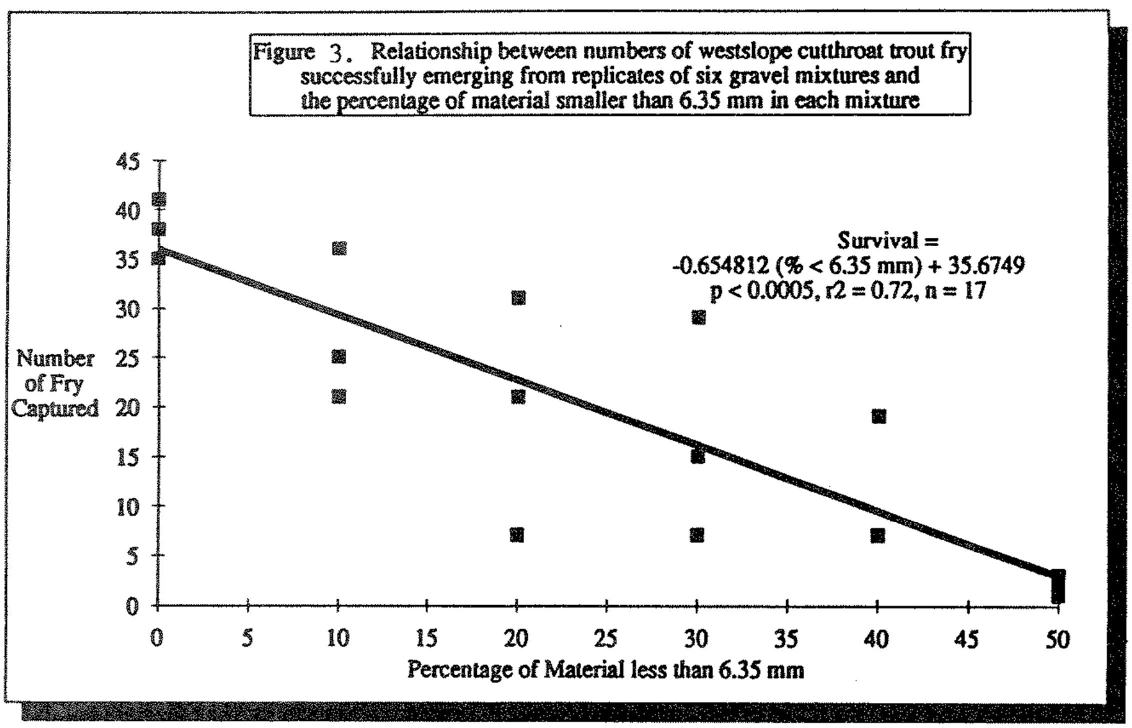

Fig. 1. Weaver and Fraley (1991) Fig. 3 (p. 20), survival to emergence of westslope cutthroat trout eyed eggs in relation to percent fines (<6.35 mm diameter). Note that in salmonids generally eyed eggs are less sensitive than green eggs to fine sediment effects (in this case green eggs were too sensitive to survive the experimental implantation procedure).

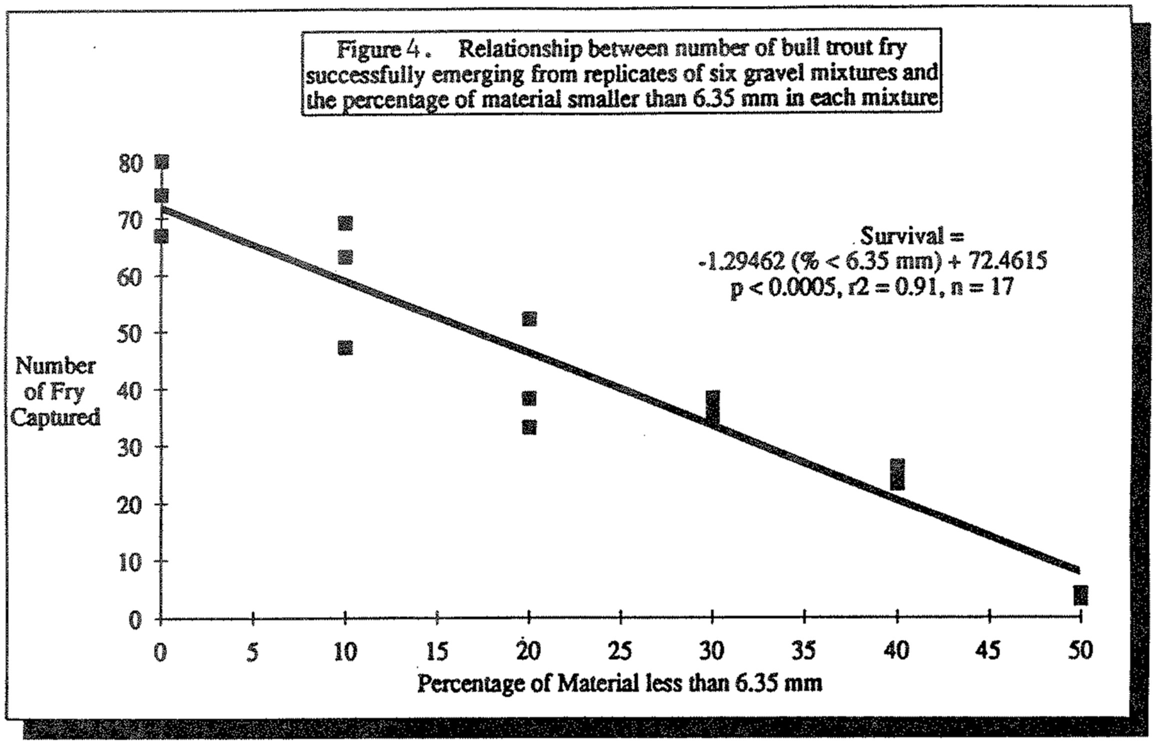

Fig. 2. Weaver and Fraley (1991) Fig. 3 (p. 25), survival to emergence of green eggs of bull trout in relation to percent fines (<6.35 mm diameter).

Newcomb and Jensen's (1991) meta-analysis and synthesis examining fine sediment impacts across all fishes also identified no generalized "safe level" of fines, rather concluded the general pattern is for cumulative increases in harm with each increment of increase in suspended or deposited fine sediment. Regardless of this science, Ecology's draft guidance document embraces the presumption of a "no effect" threshold of fine sediment for salmonids of 20% fines (<2mm diameter), without citation, and with no response to the input I provided and the sources supporting it.

Perhaps the state-of-art publication on the nature of the relationship between survival of Pacific salmon eggs and fry and fine sediments in stream gravels is the meta-analysis by Jensen et al. (2009). This study, which is of obvious central importance to inform fine sediment criteria for salmon and steelhead, was available to Ecology and was discussed during the Science Panel process, so it is puzzling that Ecology's draft guidance should be in conflict with it. Jensen et al. (2009) compiled data from available sources with quantitative data on egg-to-fry survival of Pacific salmon and steelhead in relation to fine sediment concentrations, and among other analyses, fitted general percentage survival curves, as well as estimating per capita survival probability, to the massed data set. I include as Fig. 3 here Figure 1 from Jensen et al. as a straightforward summary of the results plotted from multiple studies, for chinook salmon, coho salmon, steelhead, and chum salmon. The first point to note is that green eggs are far more sensitive to fine sediments at relatively low concentrations than more mature eyed eggs. Accordingly, sediment standards must be established to protect the most sensitive and vulnerable stage, that is, less mature "green" eggs.

Of the four panels in Jensen et al. Fig. 1, only panel d depicting the survival relationship for eyed eggs shows a fitted curve that implies a threshold below which fine sediment increases have no measurable effect on survival. In the other three panels, including panel c plotting the relationship for green eggs, the curves fitted to the massed data show virtually no evidence of a low-sediment-concentration survival threshold. As I argued in Science Panel meetings, the best-fit curves do not support the notion that a "safe level" of fine sediments exists; instead they suggest decreasing survival with increasing fine sediment percentage across essentially the entire range of the data. In fact, the only clear threshold evident is that *above a value of roughly 30 percent fine sediments, coho and chinook survival declines to effectively zero.*

REVIEW OF EGG-TO-FRY SURVIVAL OF PACIFIC SALMON 353

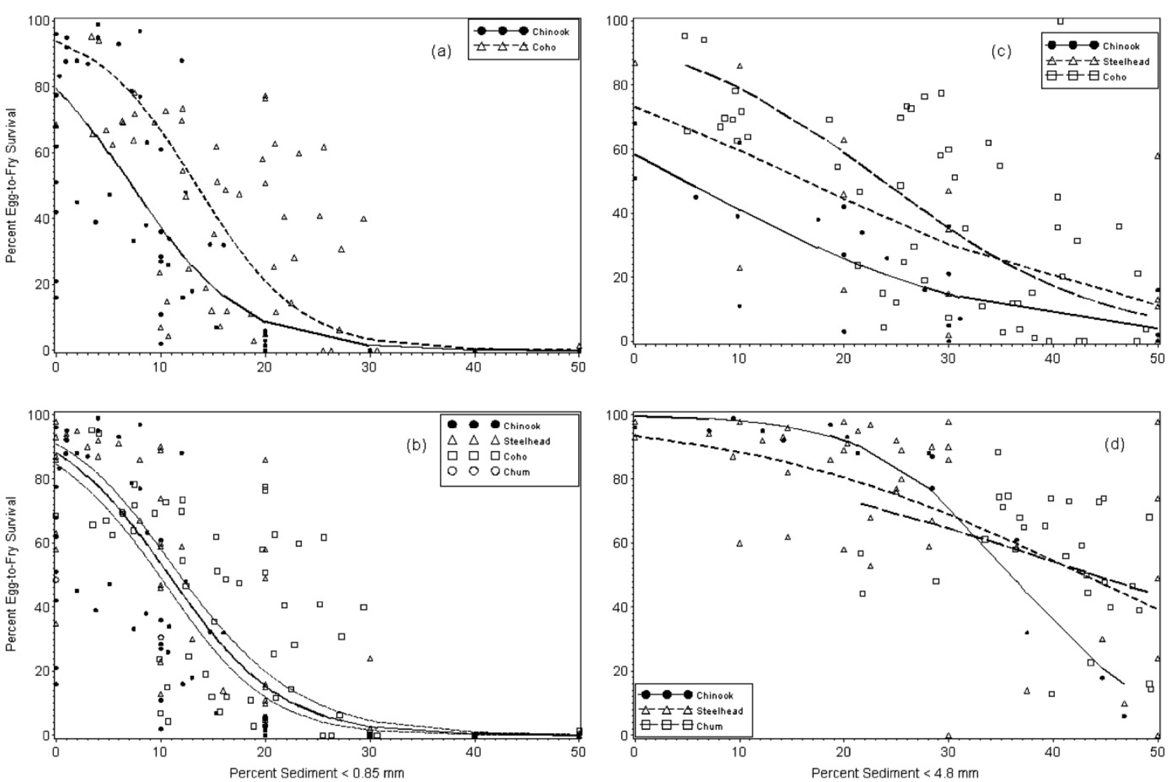

Figure 1 Relationship between egg-to-fry survival and percent sediment, showing data from the literature used in analysis. (a) Data and modeled egg-to-fry survival of Chinook and coho salmon vs. percent sediment < 0.85 mm.. (b) Egg-to-fry survival for Chinook, coho, and chum salmon vs. percent sediment < 0.85 mm. Line is the modeled mean survival based on data from all species and includes the 95% confidence interval. (c) Data and modeled green egg-to-fry survival for Chinook salmon, steelhead, and coho salmon vs. percent fines < 4.8 mm. (d) Data and modeled eyed egg-to-fry survival for Chinook salmon, steelhead, and chum salmon vs. percent fines < 4.8 mm.

Fig. 3. Fig. 1 Excerpted from Jensen et al. (2009).

Of all studies considered in Jensen et al.'s meta-analysis, only one, Tappel and Bjornn (1983), suggests a threshold of no measured impact below about 15-20 percent fines (see Jensen et al., Fig. 2). It appears all others do not suggest such a threshold is present. It is curious that Tappel and Bjornn (1983) remains the most commonly cited paper on survival of chinook eggs and fry in relation to fine sediments, considering it is an extreme high outlier (across all levels of fine sediment) among comparable studies.

What does this all mean for a fine sediment standard? First, in the absence of locally or regionally specific data and a rigorous finding to the contrary, *no protective standard should assume a "safe threshold" of fine sediment exists*. Second, and consequently*, a narrative or quantitative standard should protect against any increase in fine sediment over existing conditions.* Third where and when existing fine sediments appear to be in excess of natural background (e.g., relative to fine sediment conditions measured at least-impacted reference sites in minimally altered watersheds), *a protective standard must mandate a trend of decreased fine sediment concentrations over time in sediment-impaired waters.* There is nothing unusual or impractical about establishing trend-based standards for fine sediment concentrations, especially in impaired waters; for a review and numerous examples, see the state of Idaho's *Guide for Selection of Sediment Targets for Use in Idaho's TMDLs* (Rowe et al. 2003).

Ecology's draft guidance offers no such clarity or direction to inform implementation to inform and enforce the proposed narrative criterion that "Water bodies shall not contain fine sediment (<2 mm) from anthropogenic sources at levels that cause adverse effects on aquatic life, their reproduction, or habitat…" In fact the so-called guidance defers not only the specific methods of implementation, it seems to defer on a vast portion of the general and specific scientific content that the Science Panel provided input on over multiple meetings. More specifically, the inclusion of a threshold value for percent fines in the "weight of evidence" example described in the draft guidance would essentially give all streams in Washington with a fine sediment concentration (fines <2mm diameter) a "free pass." This is so regardless of the fact that a stream with natural low mean concentration of fine sediment of 10 percent, for example, has or is expected to have the concentration of fines doubled to 19 or 20 percent. An examination of panel c in Figure 1 of Jensen et al. will clearly show that an increase from 10 percent fines to 20 percent fines is expected decrease the mean egg-to-fry-survival rate from 50% to roughly 17%. So for this simple example, a 66 percent loss of egg to fry survival would be permitted by Ecology under its suggested guidance. In my opinion, that does not remotely qualify as a protective standard. In fact it's an effective recipe to allocate future man-made sediment inputs and habitat degradation to the streams that are presently those least impacted and most productive. This should help illustrate why it seems clear to me that Ecology has not adequately or accurately accounted for the available most relevant science in preparing the guidance document.

4. Reference Site Applications

Employing reference sites to establish an estimated or assumed natural condition as a baseline is a valid approach to measuring and assuring stream resource protection for many physical and biological factors, including fine sediment. This approach is implemented in other states, including Idaho and California, to tailor quantitative criteria regionally and locally to ensure that narrative criteria are appropriate to potential natural conditions (Rowe et al. 2003). On a positive note, the draft guidance provided does acknowledge limitations of Ecology's reference site data for this purpose, and seems to make clear that the limitations need to be explicitly accounted for implanting this approach. That said, no road map or direction is provided to suggest what the limitations of the data might be, and how they might be accommodated and accounted for in an effective analysis. By contrast, in Idaho's guidance (Rowe et al. 2003) includes or refers to numerous examples of *a priori* analysis to systematically stratify streams by ecoregion and empirically validate potential benchmark fine sediment conditions by stream geographic grouping.

It has been extremely disappointing to me that during the process of development of the narrative standard and proposed draft guidance, Ecology never produced a shred of data from reference sites, let alone a simple example of how such a comparative analysis might be conducted, to offer to the science panel for review. I was extremely frustrated by this lack of an example to validate the concept and provide specifics of a methodology, because in theory it could be a key approach and a cornerstone of a protective standard. However, through other research experience I am familiar with some concerning limitations with Washington's reference site data. These limitations mean that exactly how these data are selected for relevance, screened for quality assurance, then qualified, summarized, and analyzed to establish a benchmark, are all critical to assure accurate assessment of potential conditions, hence effective resource protection, will result. Whether by intent, or through inability to muster the person-hours to follow through, Ecology kept all of these critical questions of reference site data limitations, appropriate analytic design, and other aspects of quality assurance off the table and outside the scope of Science Panel review.

5. <u>Problems with Quasi-Proposed Weight of Evidence Approach</u>

The "weight of evidence" approach rather loosely proposed by Ecology on p. 21-32 of the draft guidance document suffers from several conceptual and operational problems that in my opinion would very likely result in bad decisions that would fail to be protective, and would allow or permit impairment of salmonid spawning habitat. It almost seems as if the procedure was intended to ease the path toward putative support of decisions that would result in increased fine sediments and impairment of spawning and incubation habitat, especially in higher-quality areas (see discussion above). The problems all partially overlap and interrelate, but they can be enumerated as follows: 1) an unqualified assumption of parity among different categories and sources of information; 2) the lack of any screening process to assess what are sure to be fundamental relevance, veracity, reliability and uncertainty of data from the different categories; 3) the lack of a weighting process to give greater credence to more reliable data sources and types; and 4) built-in incentives to cherry pick what data are included in the assessment, and to include poor quality data to deliberately offset the implications of higher-quality data. That is, data with low sensitivity, reliability and relevance could be introduced to “stack the deck” and cancel out the clear implications of date with high sensitivity, reliability and relevance.

The most likely outcome from applying this scheme as described by Ecology is be doubly concerning, from the standpoint of protecting salmonid spawning and incubation habitat. First, prediction of adverse effects of fine sediment increases can easily be watered down with regard to magnitude of impact and the certainty of the determination. Second, and equally important, the prediction of presumed benefits from actions intended to reduce fine sediments could be greatly inflated and exaggerated. Either of these outcomes would jeopardize the ability of Ecology to implement the narrative standard in a way that ensures protection and (where necessary) restoration of spawning habitat, fish populations, and fisheries. It is easy to anticipate that those outcomes are pretty much the same in process and outcome to the prevailing status quo, which as ESA listings of salmonids abundantly demonstrate, are systematically non-protective.

The literature on environmental assessment offers a number of general logical and practical criteria that should be applied to accurately inform a weight of evidence approach to decision-making (e.g., EFSC Scientific Committee 2017, USEPA 2016, Hull and Swanson 2006). That literature also makes clear the many ways a weight of evidence approach can fail, whether through ignorance or deliberate manipulation, if it is naively or artfully applied. Ecology offers no hint that these lessons and methods to guard against misapplication have been considered or are in place here to ensure the proposed weight of evidence approach is sound.

Besides the above-mentioned lack of procedures and criteria for assessing the relative relevance, veracity, reliability and magnitude of impact of information, Ecology’s example is tainted by the imposition of arbitrary, undefended and likely indefensible assumptions about cause and effect and biological responses to sediment conditions. The most obvious example is Ecology’s invocation in its example of a <20% fine sediment level as a “pass” criterion. As described above, this criterion is not defensible in the face of available scientific research. I am certain had provision been made for the Science Panel to review this proposal before its publication, this aspect of the proposed approach, among others, would have been roundly criticized. Nevertheless, the example serves as a highly instructive illustration of how a carelessly defined and non-peer-reviewed procedure for a weight of evidence approach can too easily produce outcomes that fail to protect the target beneficial use.

6. <u>Constraints on Science Panel Review</u>

Ecology repeatedly insisted during Science Panel meetings that Ecology was specifically not interested in entertaining independent peer input on specific means of implementing a standard or narrative criterion. Several times various Science Panel members pointed out the problem that this limitation severely constrained the ability of the panel to evaluate and offer comment on the defensibility, feasibility, and potential effectiveness of Ecology’s proposed narrative standard for fine sediment. Both the means of measurement of fine sediment and the relationship between fine sediment conditions and biological uses, including fish survival, are clearly matters of scientific endeavor, and the Science Panel members demonstrated extensive and deep expertise in these matters. Preventing the Panel from assessing and providing input on specific implementation guidance essentially equated to disallowing Science Panel members from being able to form opinions on the adequacy of a proposed standard, especially given how vague and general the proposed narrative standard is.

The proposed very general narrative criterion could in theory be protective if adequately implemented—but it could also be wholly non-protective if not adequately and rigorously implemented. The draft guidance piece belatedly document provided by Ecology does little to inspire a presumption of adequate and rigorous implementation, as outlined in my comments above. In sum, I was personally very disappointed in Ecology's management of the science panel process and in particular the deliberate limitations established to prevent the panel from reviewing implementation guidance.

In the its present form—that is, in the absence of rigorous, feasible guidance refined and supported by peer review—the proposed narrative criterion for fine sediment is little more than a tautological restatement of the agency's plain legal imperative. It's as if the Clean Water Act asks, "What will the state of Washington do to protect spawning habitat to support fisheries beneficial uses and meet ESA obligations for listed fish species?" and after one or two years of deliberation (following at least four decades of foot-dragging), the state's reply is "Yes the state of Washington will do something to protect spawning habitat." If this were how one of my students answered an exam question, I would alas be obligated to score it a zero.

Thank you for considering my comments, and for the opportunity to participate in Ecology's advisory science panel.

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Article Author: R.L. Beschta & W. L. Jackson

Article Title: The Intrusion of Fine Sediments into a Stable Gravel Bed.

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The Intrusion of Fine Sediments into a Stable Gravel Bed

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BESCHTA, R. L., AND W. L. JACKSON. 1979. The intrusion of fine sediments into a stable gravel bed. J. Fish. Res. Board Can. 36: 204–210.

A rectangular flume was used to study variables affecting the intrusion of fine sands into a stable gravel streambed. The amount of intrusion by sand (median particle diameter 0.5 mm) was determined under varied conditions of discharge, depth, velocity, flume slope, and rates of sediment transport. During all experimental tests, sand particles were trapped in voids within the upper 10 cm of an initially clean gravel bed (median particle diameter 15 mm), forming a barrier to further intrusion. An analysis of flow variables showed that flow conditions, as indexed by Froude number, significantly (90% confidence level) affected intrusion amounts, possibly by influencing the rate and depth of formation of the sand seal. Intrusion amounts, expressed as a percent of total volume, varied from 2 to 8%. Two replications used a finer grade sand (median particle diameter 0.2 mm) that intruded more and, in one case, completely filled the gravel pore space (25% by volume), further indicating that particle size, and not hydraulic variables, may have a more important influence on the total amount of intrusion.

Key words: sediment transport, intrusion, streambed, substrates, riffles, sedimentation

BESCHTA, R. L., AND W. L. JACKSON. 1979. The intrusion of fine sediments into a stable gravel bed. J. Fish. Res. Board Can. 36: 204–210.

Nous nous sommes servi d'une auge rectangulaire pour étudier les variables affectant l'intrusion de sable fin dans le lit de gravier stable d'un cours d'eau. Nous avons déterminé les quantités de sable (diamètre médian des particules 0,5 mm) introduites dans diverses conditions de débit, profondeur, vélocité, pente de l'auge et vitesse de transport des sédiments. Pendant tous les essais, les particules de sable furent emprisonnées dans les espaces des 10 cm supérieurs d'un lit de gravier initialement propre (diamètre médian des particules 15 mm), formant une barrière à l'introduction d'une plus grande quantité de sable. Une analyse des variables du débit démontre que ce dernier, indexé selon le nombre de Froude, affecte de façon significative (limite de confiance à 90%) les quantités introduites, possiblement en agissant sur la vitesse et la profondeur auxquelles la couche scellante se forme. Les quantités introduites, exprimées en pourcentage du volume total, varient de 2 à 8%. Dans deux essais répétés, nous avons utilisé un sable plus fin (diamètre médian des particules 0,2 mm) qui pénétra davantage et qui, dans un cas, remplit complètement les pores du gravier (25% en volume). C'est une indication de plus que la taille des particules peut avoir une plus grande influence que les variables hydrauliques sur le degré d'intrusion.

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IN small mountain streams, particle size distributions of bed materials are affected by many interrelated factors. Climate, soils, parent materials, watershed characteristics, and land use activities all interact to produce various combinations of sediment sizes. Particularly important are the bed materials approximately 1 mm in diameter or smaller. These fine sediments can significantly affect fish habitat and other instream biota (Gibbons and Salo 1973; Meehan and Swanston 1977).

Although the evidence is not entirely consistent, research results have generally shown that increased levels of fines have detrimental biological effects (N. H. Ringler unpublished data; Moring and Lantz 1974; Iwamoto et al. 1978). In forest streams of the Pacific Northwest, fine sediments can affect anadromous and resident fish populations after the deposition of eggs. Fine sediments may reduce intergravel circulation of water, which decreases the dissolved oxygen available in the spawning gravels (Vaux 1962). Apparently, relatively small percentage increases in the volume of fine sediments may greatly reduce the permeability of stream gravels (Phillips 1971). In addition, sediments may block the movement of fry from the gravels to the stream (Koski 1972). By eliminating intergravel crevices for juvenile fish and other organisms, fine sedi-

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ments alter habitats and decrease fry survival. Iwamoto et al. (1978) identified several studies noting such effects. Furthermore, many common riffle insects are unable to move upstream on sand substrates (Luedtke and Brusven 1976).

Research has also shown that forest harvesting operations (including road building, yarding, and slash disposal) in steep terrain can increase the amount of sediments entering the stream system. Although watershed responses to land use are highly variable, sediment transport rates typically recover, with time, toward pretreatment levels (Anderson 1972; Brown 1972; Megahan 1974; Swanson and Dyrness 1975). The linkage between this temporary increase in sediment production and changes in streambed composition have not been identified.

Unfortunately, little information is available relating the temporal and spatial variability of percent fines in streambed gravels to hydraulic variables, rates, and mechanisms of sediment transport (i.e. suspended load vs. bedload), as well as sediment characteristics. Although sediment transport mechanisms are complex (Lawson and O'Neill 1975), selected hydraulic and sediment variables may be useful for characterizing these mechanisms and the intrusion or deposition of fines.

This study evaluates factors affecting the intrusion of fine sediments into a stable gravel substrate, thereby improving understanding of the process and possible biological impacts.

Methods

The study was conducted during the summer of 1977 at the Kalama Springs Field Laboratory of the Weyerhaeuser Company, approximately 80 km east of Longview, Washington. A wooden flume, 7.6 × 0.71 m, was constructed so that a range of streamflow and sediment transport conditions could be evaluated. Although a flume does not simulate the exact conditions found in mountain streams, this approach was used because flow conditions and sediment transport variables could be controlled and modified as desired. The flume was hinged at the upstream end and supported by hydraulic jacks so the bottom slope could be adjusted. Several glass windows installed along the sides of the flume allowed observations of both the transport and intrusion of fine sediments.

Water from Kalama Springs was used for each test. A headgate upstream from the flume controlled the discharge of water into an artificial channel that led to the flume. At the immediate upstream edge of the flume, a weir and staff gage enabled us to monitor discharge during each test. Water temperatures remained a constant 7°C throughout the study.

A fertilizer hopper 46 cm wide was mounted at the upstream end of the flume to meter fine sediments into the flowing water. Delivery rates of fine sediments were controlled by a calibrated lever that opened the spreader. After each test, actual rates of sediment delivery or inputs were calculated from the volume of fines that had passed through the spreader.

Clean gravels to a depth of 30.5 cm were placed into the test section of the flume for each test. These gravels, leveled and lightly tamped to provide a uniform gravel surface, averaged 15 mm diam and had a particle size distribution as shown in Fig. 1. The particle density and porosity of these gravels averaged 2.67 g/cm³ and 35% (by volume). Sand-size sediments, averaging 0.5 mm diam (Fig. 1) and a particle density of 2.26 g/cm³, were used for most of the intrusion evaluations. Both sands and gravels match those being used by fisheries biologists at the Weyerhaeuser Company to assess survival to emergence ratios of salmonid embryos in artificial redds (J. Fisher, Weyerhaeuser Company, Longview, Wash., personal communication, 1977). The relatively low particle density of the 0.5-mm sands can be attributed to the high occurrence of pyroclastic rock types comprising individual sand grains.

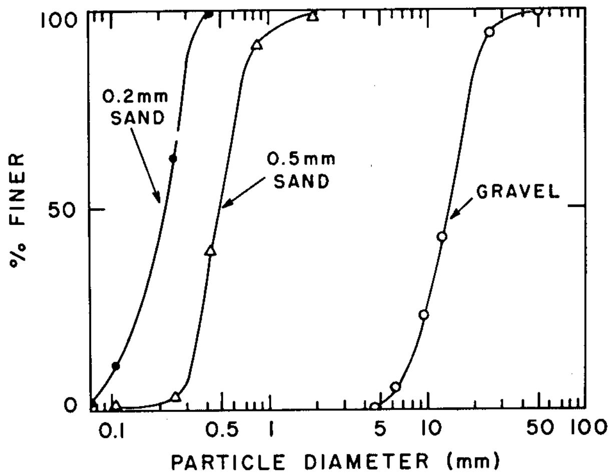

FIG. 1. Size distributions of sands and gravels used to evaluate the intrusion of fine sediments into a gravel-bedded test channel.

During 18 separate runs in the test flume, we evaluated these sediment transport conditions: 1) three discharge rates (≈50, 75, and 90 L/s); 2) three slope conditions (Froude numbers ≈0.6, 1.1, and 1.5); and 3) two (low and high) sediment input rates (≈2500 and 11 500 g/min). These 18 tests (3 × 3 × 2 = 18) were conducted with the 0.5-mm sand. Two additional tests used fine sediments with a particle diameter averaging 0.2 mm (Fig. 1) and a density of 2.53 g/cm³. A final run for an extended period (i.e. 120 min) was made with the 0.5-mm sand at the low input rate to determine maximum levels of intrusion for a given flow condition. All other tests lasted either 30 or 60 min for the high and low input rates, respectively, because field observations and measurements during individual tests indicated that most of the intrusion occurred within the first 15–20 min of a run.

For a given discharge rate, flume slope was adjusted for evaluating a range of Froude numbers. The dimensionless Froude number (F_r), representing the ratio of inertial to gravitational forces in fluid flow (Streeter and Wylie 1975), is expressed as

$$F_r = \frac{\bar{V}^2}{gy}$$

where $\bar{V}$ = mean velocity, m/s
g = acceleration due to gravity, 9.8 m/s²
y = depth of flow, m

The Froude number is a useful way to characterize flow conditions. For subcritical flow ($F_r < 1.0$), stream conditions are typified by relatively deep, slow streamflow. At a critical flow ($F_r = 1.0$), the specific energy ($E = \bar{V}^2/2g + y$) is at a minimum. "Standing waves" in a stream indicate critical flow conditions. Supercritical flow ($F_r > 1.0$) is characterized by relatively shallow, rapid streamflow. Because all three conditions occur in mountain streams, this study evaluated a range of Froude numbers.

For each test run, water velocity and depth were measured at 5-cm intervals across the width of the flume near the center of the gravel test section. Velocities were measured using a pygmy current meter. Vertical velocity profiles were also determined for each flow condition. Two small siphon tubes (0.64 mm diam) extracted water and transported sediment from elevations of 1 and 6 cm above the gravels to determine sediment concentrations immediately above the gravel bed.

After each run, cores of the gravel bed were frozen (Walkotten 1973) for estimating the amount of sand that had intruded into the test gravels. This procedure was also intended to provide information about the positioning of intruded sands within the gravel bed. However, during the first several experimental runs, we had a problem — inserting the probe of the freeze-core sampler disturbed the gravels so that sands deposited among the surface gravels moved downward and were redeposited. Wendling (1978) encountered a similar problem when sampling streambed sediments in Alaskan streams. In our situation, the retrieved frozen core usually had sands concentrated within several centimeters of the flume bottom but few, if any, sand particles above that point. Observations through the flume windows confirmed that the sands settled throughout the gravel profile during sampling. Consequently, after the first three tests (no. 1, 4, and 6), we implemented another method of sampling. We buried two containers (12.5 cm diam, 15.0 cm tall, and open at the top) in the gravels before a run, removed them after completing the run, and then determined the dry weight of sands.

Total intrusion for a given run was calculated by dividing the weight of the sands in each container by the weight of the gravels in a column 12.5 cm diam and 15.0 cm high (even though the total depth of gravels was 30.5 cm). These weights were adjusted to account for differences in particle densities between the sands and gravels. Thus, intrusion amounts were expressed as percent by volume of intruded sands in relation to the total volume of sample. The choice of a 15-cm depth was arbitrary, but made the calculated percentages more representative of average fine concentrations in the gravels.

After each run and the retrieval of sampling containers, the gravels were flushed from the flume and clean gravels were inserted into the test reach.

Results and Discussion

The flume windows proved particularly useful for visually observing sediment transport and intrusion processes. At low Froude numbers ($F_r < 0.9$), sands were transported mostly within 1 cm of the gravel bed surface. The transport process consisted mainly of individual grains rolling and sliding along the tops of gravels and other sand particles. At Froude numbers generally greater than 0.9, transport mechanisms shifted so that the suspension and saltation of sand particles became increasingly important. This was further illustrated by the increased average sediment concentrations (mg/L) measured at both 1 and 6 cm above the gravel bed during runs with 0.5-mm sands. Sediment

Height (cm)	Froude number $0.5 < F_r < 0.9$	$0.9 < F_r < 1.3$	$1.3 < F_r < 2.6$
1	1700	4800	4500
6	120	370	760

concentrations 6 cm above the gravels typically averaged about one tenth of those measured at 1 cm. Dunes, ripples, or other bed forms were not observed during any of the test runs.

Apparently sediment deposition and intrusion into the gravels involves two principal mechanisms: 1) the transport and deposition of sand particles into the surface voids of the gravel bed, and 2) the settling of the particles into deeper gravel voids. The settling process occurs primarily under the influence of gravitational forces, but seems assisted by turbulent pulses at the gravel surface.

At low Froude numbers ($F_r < 0.9$), the 0.5-mm sands quickly established a sand "seal" within the upper 5 cm of the test gravels as the larger sand particles bridged the openings between adjacent gravel particles and prevented the downward movement of additional sands into the gravels. Once this sand seal had formed and the intergravel space above had filled with fines, the intrusion process stopped. Additional sands were transported past the gravel test section. At higher Froude numbers ($F_r > 0.9$), flow characteristics began to alter this process. Observations indicated that most deposition and intrusion now occurred within the upper 5–10 cm of the gravels. Because of the higher velocities associated with higher Froude numbers, these flows were characterized by greater bed shear and the periodic jiggling of the surface gravels. These turbulent pulses seemed to inhibit the formation of a sand seal near the gravel surface. As a result, the sand seal still formed, but generally deeper within the test gravels. Again the formation of a sand seal in a stable gravel substrate would preclude the additional intrusion of fine sediments into the lower gravels.

Table 1 summarizes measurements for each of the 21 experimental runs. For the first 18 runs (using the 0.5-mm sands), an analysis of variance indicated that flow condition, as indexed by Froude number, significantly (90% confidence level) influenced intrusion amounts. Apparently the main influence of flow conditions was to vary, within a relatively narrow range, the depth and rate of formation of the sand seal. Other flow indicators, such as average bed shear stress, unit stream power, or Reynolds number, did not significantly affect the amount of intrusion, making a physical interpretation of the Froude number relationship difficult.

TABLE 1. Test conditions during the intrusion of fine sediments into a gravel-bedded channel.

Test	Mean particle size (mm)	Test duration (min)	Hydraulic variables: Discharge (L/s)	Froude number	Velocity (m/s)	Depth (cm)	Sediment transport and intrusion variables: Input rate (g/min)	c_1[a] (mg/L)	c_6[a] (mg/L)	Total intrusion (% by vol)
1	0.5	60	45.3	0.66	0.75	8.6	1 680	1 010	13	—
2	0.5	60	49.8	1.13	0.92	7.6	2 470	1 430	35	4.6
3	0.5	60	51.0	1.55	1.03	7.0	1 970	1 440	—	6.6
4	0.5	30	44.2	0.89	0.82	7.6	10 000	2 200	72	—
5	0.5	30	53.8	1.08	0.94	8.2	12 850	10 980	260	8.0
6	0.5	30	48.7	2.57	1.20	5.7	8 570	3 770	—	—
7	0.5	60	78.7	0.54	0.84	13.2	1 990	790	70	7.0
8	0.5	60	73.1	1.28	1.09	9.5	2 600	1 500	120	4.5
9	0.5	60	70.2	1.52	1.14	8.7	2 470	1 770	170	6.8
10	0.5	30	73.1	0.53	0.81	12.7	11 860	2 800	200	5.3
11	0.5	30	73.6	1.18	1.06	9.8	11 860	7 210	660	5.8
12	0.5	30	72.2	1.53	1.15	8.9	12 920	7 480	1 200	8.3
13	0.5	60	90.1	0.56	0.88	14.3	2 140	812	70	5.9
14	0.5	60	90.6	1.06	1.10	11.6	3 630	2 450	230	5.0
15	0.5	60	96.3	1.43	1.24	11.1	3 130	1 750	500	7.4
16	0.5	30	91.8	0.66	0.94	13.7	10 830	2 470	280	2.6
17	0.5	30	92.3	0.92	1.05	12.4	11 860	8 970	1 100	5.8
18	0.5	30	92.6	1.52	1.24	10.5	11 860	10 740	1 150	10.4
19	0.2	60	94.0	0.63	0.93	14.1	2 150	970	210	24.9
20	0.2	60	91.5	1.12	1.12	11.5	1 980	860	220	15.3
21	0.5	120	91.8	0.93	1.06	12.2	3 220	1 370	160	6.5

[a]c_1 and c_6 = sediment concentrations 1 and 6 cm, respectively, above gravel surface.

Intrusion amounts by total volume ranged from 2 to 8%.

An increasing Froude number affected intrusion amounts differently at low (Fig. 2) and high (Fig. 3) rates of sediment input. At low sediment input rates, and hence low transport rates across the gravels, intrusion amounts reached a minimum for Froude numbers of approximately 1.0–1.2. However, at high rates of sediment input of the 0.5-mm sand, intrusion amounts generally increased as Froude number increased. These contrasting results further indicated that the intrusion of fines into a stable gravel bed is a complicated and not fully understood process.

Although only 2 of the 21 runs (no. 19 and 20) used 0.2-mm sands, these runs substantially contrasted with those using 0.5-mm sands. The major difference was the absence of a sand seal in the upper gravels. Instead, these finer sands generally moved down through the gravels by gravity and began to fill the test gravels from the bottom up. These observations concur with flume studies conducted by Einstein (1968) where he evaluated the intrusion of silt-size particles into streambed gravels and concluded that silts fill gravels from the bottom upwards.

In comparison to the 0.5-mm sands, more 0.2-mm sands intruded (Fig. 2), further suggesting that particle size is an important variable affecting intrusion into stable gravels. The amount of intrusion by the 0.2-mm sands also substantially decreased as Froude number increased from 0.6 to 1.1. The percentages shown in Fig. 2 are the results of tests that lasted only 1 h. Yet, within that time frame, the 0.2-mm sediments attained intrusion amounts from 15 to 25%. Complete filling of the gravel interstices would be represented by a value of approximately 25% by volume for the 0.2-mm sands.

Although observations indicated that the intrusion and deposition of 0.5-mm sands stabilized before completion of a given run, test no. 21 was implemented to see whether intrusion would continue over a longer period. The measured intrusion of 6.5% for this run was within one percentage unit of that predicted from the curve in Fig. 2. This result seems to substantiate observations that intrusion had essentially stopped by the end of each run.

Particle size distributions for samples obtained during run no. 21 provide an interesting contrast to the transport and intrusion of fine sediment. The particle size distribution of sands from the siphon located 1 cm above the gravels is essentially identical to that of the 0.5-mm sands added at the upstream end of the flume. However, the intruded sands have a mean particle size of <0.3 mm, indicating that the intrusion process into a stable streambed may be selective towards smaller particles (Fig. 4).

Results also illustrate that the mean particle size of sediment in transport 6 cm above the gravels is finer than at 1 cm (Fig. 4). Steep velocity gradients near the gravel surface (Fig. 5) were capable of temporarily

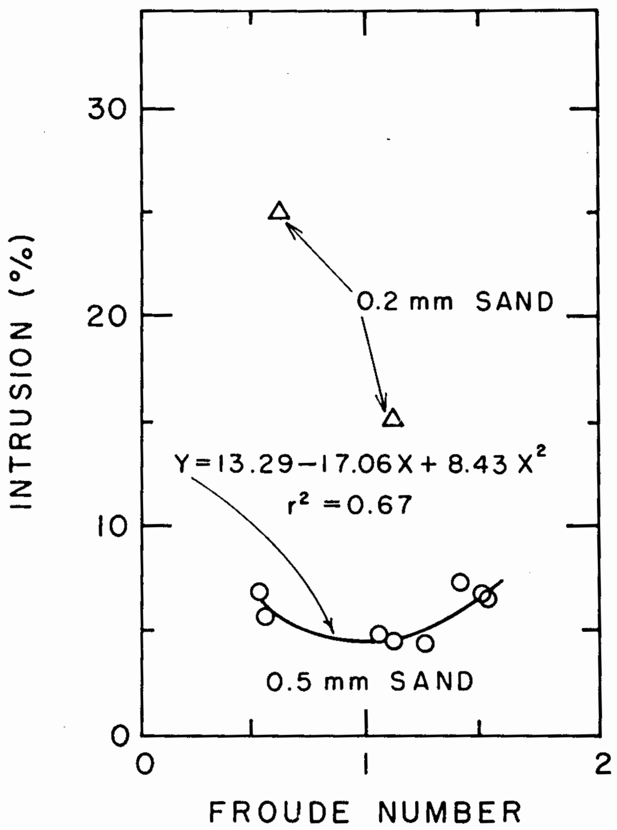

FIG. 2. Total intrusion of sand, percent by volume, in relation to Froude number for low sediment input (≈2500 g/min).

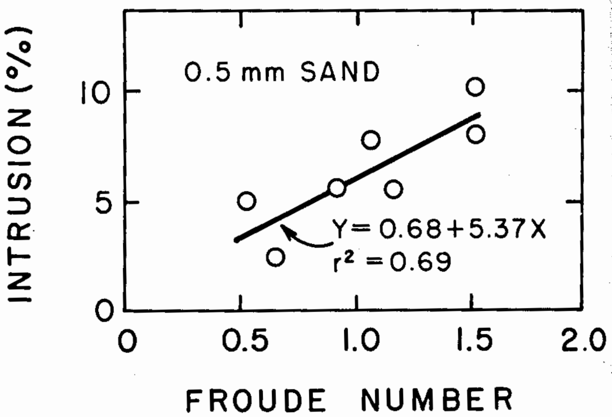

FIG. 3. Total intrusion of sands, percent by volume, in relation to Froude number for high sediment input (≈11 500 g/min).

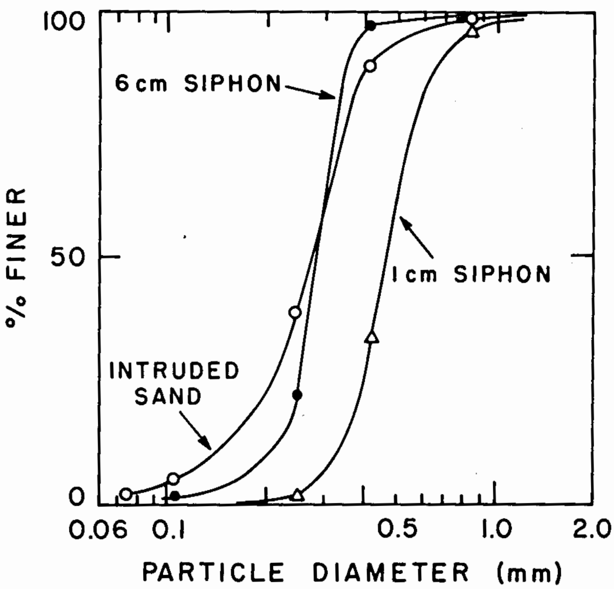

FIG. 4. Size distributions of sands during intrusion test no. 21.

suspending nearly all sand particles at least 1 cm above the bed. At the 6-cm elevation, a reduced velocity gradient and resultant shear stresses (even though average velocities are higher) reduced the transport of larger sediment particles. This is illustrated by the shift in mean particle sizes from 0.5 to 0.3 mm at elevations of 1 and 6 cm, respectively, above the gravels.

An additional experimental run with a flow of approximately 90 L/s and a Froude number exceeding 1.0 was made at the Kalama Springs Field Laboratory; results were not quantified. The 0.5-mm sands intruded until a sand seal formed and the interstices of the gravels had filled with sand. At that point, we stopped adding sand and allowed the flow to clean or flush the fines from the gravels. Observations and photographic evidence showed that fines flushed from the gravels to a depth of about 1 cm. Sands below 1 cm were not entrained by the flowing water, and further flushing did not occur, suggesting stream gravels can be cleansed or flushed only when in motion during high flow events. Without movement of these gravels, apparently no mechanisms are available for flushing. Thus the amount of fines transported during high flow events may have a pronounced effect on the natural quality or composition of gravel substrates in mountain streams. However, McNeil and Ahnell (1964) reported that local fines can be removed from the streambed during spawning.

Flow conditions, sediment transport rates, and sediment particle size all influenced the amount of fines deposited in initially clean gravels. These results indicate the need for improved understanding of the mechanisms of intrusion and the importance of mean particle size in affecting the intrusion of fines. Although several factors were found to influence intrusion, the direct application of these results to natural streams is tenuous. Measurements in small mountain streams must be improved before the ramifications of flume studies

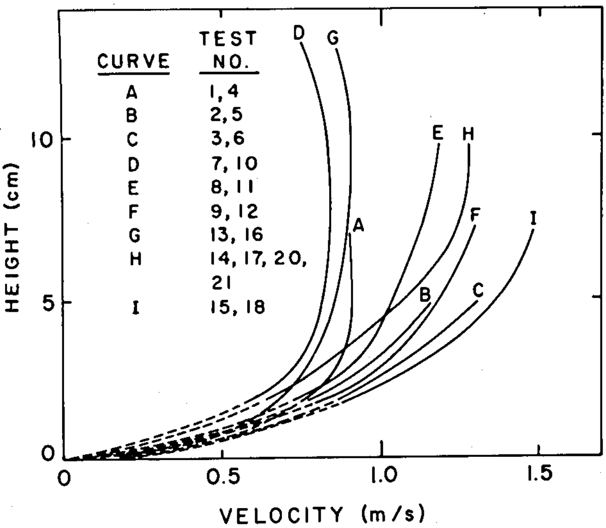

FIG. 5. Velocity profiles measured above gravel test section.

can be fully understood and extrapolated to field situations. For example, flow conditions and transport rates of fine sediments over known spawning gravels must be characterized for a range of high flow conditions. With such information, we may be more able to fully assess the impacts of sedimentation upon the habitats and organisms found in mountain streams.

Even though the quantitative results of this study cannot be directly applied to small streams, the study has several implications concerning land-use activities and stream sedimentation. Fine sediments added when streambed gravels are stable will deposit and intrude into initially clean gravels. If the fine particles are large enough to bridge openings between gravel particles, a seal will form. Then fines will deposit above this seal and fill the upper layers of the gravels. If the fine particles are small enough, they will fill the interstices of the gravels from the bottom up. As long as the gravels remain stable and do not move, adding fines to a stream can only result in the intrusion of fines or perhaps a blanketing of gravel substrates. Biologically, the presence of such conditions will generally have undesirable impacts.

If only intrusion can occur during periods when streambeds are stable, then the question may be posed, How and when are fines flushed in small streams? Apparently, flushing can occur only during periods of relatively high flows that disturb the channel bed and cause bedload transport. Field measurements we have made in Oregon's Coast Range streams indicate that the general transport of bed material (sand size and larger) occurs after discharges exceed ~ 0.15 $m^3 \cdot s^{-1} \cdot km^{-2}$. Based on a frequency analysis of daily streamflow values, flows exceed this level on an average of about 20 d each year. If the amount of fines is increased during high flow periods as a result of land use activities, stream energy may be used to transport and intrude the additional sediment load instead of flushing fines from the gravels. Thus, additional amounts of fines from accelerated surface or hillslope erosion during high flow events may directly influence gravel quality. Consequently, land managers must continue their efforts to control activities that have a high potential to increase stream sedimentation. Increased amounts of fine sediments must be minimized if the natural quality of a stream system is to be maintained.

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Impact Targets versus Discharge Standards in Agricultural Pollution Management
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Impact Targets versus Discharge Standards in Agricultural Pollution Management

John B. Braden, Robert S. Larson, and Edwin E. Herricks

When attempting to protect fish in streams, sediment or erosion targets are inefficient. Use of a habitat suitability target reveals lower cost abatement measures because it accounts for pesticides as well as soil particles. In Lake Michigan case studies, the lower cost measures involve more crop diversity, less use of no-till, and changes on more acres than the solutions based on sediment discharges or erosion rates.

Key words: environmental policy, fish, nonpoint pollution, optimization, targets.

Protecting water quality has been avowed as a major objective of soil conservation programs (U.S. Department of Agriculture). Yet, controlling soil movement continues to be stressed in practice; an example is the Conservation Reserve Program's emphasis on highly erodible lands. The effectiveness of protecting water quality by stabilizing soil is an open question.

This paper quantifies the economic losses from the use of soil movement rather than water quality criteria for the attainment of an important water quality goal—protection of habitat for high-valued fish species. The estimates are based on a case study of Lake Michigan tributaries. The case study also indicates the differences in farming practices when habitat is protected by controlling soil movement alone versus managing both soil and pesticide pollution.

The analysis extends the methods used previously by Braden et al., Crowder and Young, Heimlich and Ogg, Milon, Park and Sawyer, and Park and Shabman. The common theme of those studies is the linking of land management economics to off-site consequences. The usual aim has been to compare various policies for meeting specified levels of pollutant loads.

The present study goes beyond previous work by considering predicted impacts on fish habitat. The costs and management implications of actual environmental damages, as well as of emissions and pollutant loads, then can be assessed. These three performance measures conform to the policy targets defined by Nichols—emissions, exposure, and damages; however, his "damage" category implies economic evaluation of the impacts, while no such evaluation is attempted here. Insight into the inefficiencies introduced by inexact targets is important in evaluating whether to incur the additional costs of measuring and monitoring actual damages.

Economic Model

The economic model portrays a fully informed watershed planner. The planner's objective is to achieve environmental goals at least cost. Difficulties of attaching values to environmental impacts frequently lead to the use of such second-best cost-effectiveness criteria (Baumol and Oates).

The decision context involves stochastic risks of environmental impacts. The environmental consequences of interest depend in part on the severity and timing of weather events in relation to crop cover and chemical use. All of these factors can vary over a planning horizon. Under

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such circumstances, the planner's decision framework follows in spirit the models of Beavis and Walker and Lichtenberg and Zilberman. Let $C(x)$ be a primal cost function defined over choice variables in the vector x, $q(x, \varepsilon)$ be an environmental quality production function, ε be a stochastic disturbance term, Q be a target level of minimum environmental quality, and A be a measure of reliability (the probability of achieving Q). The cost-effectiveness decision is

$$(1) \quad J = \text{Min } C(x), \text{ s.t. } \Pr[q(x, \varepsilon) \leq Q] \geq A.$$

This framework anticipates that higher quality can be achieved only by sacrificing cost or reliability or both. Greater reliability presents similar tradeoffs.

The environmental quality and reliability targets in (1) may be difficult to measure and monitor. Contributing factors, such as emissions or pollutant loads reaching the stream, may provide alternative policy targets. Suppose that the function $q(\bullet)$ can be rewritten as

$$(2) \qquad q = \hat{q}(x, h(\hat{x}), \varepsilon), \hat{x} \subset x, x \not\subset \hat{x},$$

where $h(\bullet)$ is an intermediate environmental quality indicator, such as emissions or pollutant loads. The planner then may consider the alternative policy:

$$(3) \quad J = \min C(x), \text{ s.t. } \Pr[h(\hat{x}) \leq H] \leq B,$$

where the intermediate objectives H and B are based on knowledge about the stochastic relationship between $h(\bullet)$ and final objectives Q and A.

For achieving Q and A, the management choices based on (3) will never be less costly than selections based on (1). Fewer choice variables (instruments) are relevant to the performance measures, and those measures are inexact proxies for the final objectives. The extent of the inefficiency and the nature of the management miscalculations are empirical questions.

Model Implementation

The empirical model has been described in detail by Braden, Herricks, and Larson; and Larson, Herricks, and Braden. It consists of a version of the Sediment Economics (SEDEC) model (Braden et al.; Bouzaher, Braden, and Johnson; Johnson et al.) that is extended to include (*a*) seasonality of sediment loads; (*b*) pesticide losses, toxicity, and concentrations; and (*c*) effects of sediment and pesticides on the habitat requirements of selected fish species.

Briefly, pollutant transport submodels for sediment and pesticides relate farming practices to pollutant delivery to waterbodies. SEDEC places these relationships in a spatial optimization framework, permitting the planner to identify the optimal type and location of interventions to achieve particular goals.

The pollutant loads are translated into habitat quality and reliability through habitat suitability models (U.S. Fish and Wildlife Service). These models are calibrated for individual species, of which more than forty have been characterized. Habitat suitability indices (HSI) are unitless numbers ranging from zero (poor) to one (excellent). They capture the combined effects of relevant habitat parameters, such as temperature, substrate conditions, and concentrations of contaminants.

Timing is extremely important in determining the effects on fish of soil and farm chemicals washed into streams by storm events. Monte Carlo simulations based on historical weather records, planting dates, and chemical application practices are used to capture these stochastic factors. The simulation outputs are probability distributions of pollutant discharges and habitat impacts. The probability distributions identify the likelihood (reliability) of achieving any specific suitability index level with a particular watershed management scenario.

The mathematical expression for the implemented model is a modified and elaborated version of problem (1) above. In addition to the earlier notation, let counting index j, $j = 1, \ldots, J$, refer to subwatersheds and index i, $i = 1, \ldots, I^j$, denote all possible combinations of management practices on the land units within a particular subwatershed. Each i will be called a management path. Variable $x_{ij} \in [0, 1]$ is binary and represents a management path in subwatershed j. Because only one such path can be chosen in each subwatershed, $\Sigma_i x_{ij} = 1$ for all j. PC_{ij} is the probability of pesticide concentrations exceeding a particular suitability level Q^*; PS_{ijg} is the probability of sediment suitability exceeding Q^* in season g; and A is the target level of reliability. Variable a_j is area of subwatershed j and f_{ij} is the predicted median runoff from management path i of subwatershed j. The decision problem is:

$$(4a) \qquad \text{Min } C(x) = \sum_{j=1}^{J} \sum_{i=1}^{I^j} c_{ij} x_{ij}$$

s.t.

$$\text{(4b)} \qquad \frac{\sum_{j=1}^{J}\sum_{i=1}^{I^j} PC_{ij} f_{ij} x_{ij}}{\sum_{j=1}^{J}\sum_{i=1}^{I^j} f_{ij} x_{ij}} \geq A$$

$$\text{(4c)} \qquad \frac{\sum_{j=1}^{J}\sum_{i=1}^{I^j} \mathrm{PS}_{ijg} a_{ij} x_{ij}}{\sum_{j=1}^{J}\sum_{i=1}^{I^j} a_{ij} x_{ij}} \geq A \quad g = 1, \ldots, G$$

$$\text{(4d)} \qquad \sum_{i=1}^{I^j} x_{ij} = 1 \quad j = 1, \ldots J$$

$$\text{(4e)} \qquad x_{ij} = [0, 1].$$

Constraints (4b) and (4c) are weighted-average probabilities of exceeding the target suitability level for pesticides and sediment, respectively. Variables a_j and f_{ij} are weighting terms used to reflect the relative contribution of each subwatershed.

A solution to this problem reveals the management choices that will achieve specified habitat quality and reliability levels at least cost. Varying the constraints will show the trade-offs between cost, quality, and reliability. More important, the model can be modified to constrain emissions or pollutant loads. The resulting solutions can be run through the habitat simulations to reveal how close they come to protecting quality and reliability. Doing so reveals the efficiency losses due to the use of intermediate targets.

Different targets should be good substitutes for policy purposes if they are closely correlated (Nichols). In that case, management prescriptions optimal for one indicator should be nearly optimal (although possibly second-best) for the other indicator.

In the case of agricultural pollution and fish habitat, timing can cause critical differences between emissions, releases, and habitat suitability. The timing effects come through decay in pesticide toxicity and the seasonal patterns of rainfall, erosivity, crop conditions, and fish spawning requirements.

Another source of divergence between targets is oversimplification. Environmental damages are frequently the outcome of complex processes involving many factors. Targets based on a subset of the processes may miss some important contributing factors. For example, erosion rates and sediment loads may not effectively represent the fates of soluble pesticides.

The size of the losses due to the use of proxy targets is an empirical matter. Insights are developed in a case study of sport fish protection in tributaries to Lake Michigan.

Case Study

Active sport fisheries have been successfully developed in Lake Michigan over the past two decades with substantial economic benefits for the near-shore area. Chinook, coho, steelhead, and other salmonids are the most prized varieties.

The salmonid populations have been sustained and enhanced through extensive stocking. Natural spawning has been limited in many tributaries by nonpoint pollution from farmland and by channelization that eliminates habitat while enhancing drainage. These factors not only compel continued stocking, they also reduce the range of seasonal salmon migrations. The migrations are highly valued by individuals and communities near the lake who seek to lengthen and enhance the fish runs.

The model was applied to two agricultural subwatersheds in Berrien County, Michigan, along Lake Michigan's southeastern shore. The Pipestone Creek site drains to the St. Joseph River and on to Lake Michigan. The states of Michigan and Indiana are working together to extend the salmon runs in the St. Joseph River system. The river and its tributaries have been abundantly stocked with juvenile sport fish in recent years. Portions are classified as trout streams. However, the segment of Pipestone Creek chosen for study has been channelized and the silty substrate is poor for spawning and fry development. The 93 hectare (ha) study site contains gently sloping farmland with silty and loamy soils.

The Galien River (east branch) site is part of a smaller river system that also is classified for trout. The habitat conditions are good for salmonids with a meandering channel, cobble and gravel substrate, and pools interspersed with riffle segments. The study site contains 139 hectares of gently sloping farmland with sandy and loamy soils.

Data

Catchments and transects were defined from U.S. Geological Survey topographic maps. Management units were identified from Soil Conservation Service (SCS) soil survey maps, plat maps, and Agricultural Stabilization and Conservation Service aerial photographs. Soils information, including productivity classifications, also came from soil surveys (U.S. Soil Conservation Service). Rainfall distributions were based on a fifty-seven-year record for nearby Eau Claire, Michigan. Basic stream data were compiled through fieldwork.

Coefficients for the Universal Soil Loss Equation (Wischmeier and Smith) and crop budgets, including pesticide application rates and assumptions about the timing of farming operations, were prepared by experts in the Michigan Cooperative Extensive Service and the SCS (J. Black, Dep. Agr. Econ., Michigan State University, personal communication 1988). Corn, grains, and soybeans are the most common farm crops in Berrien County, although orchards, vegetables, and vineyards also are present. The crop-cover (C) factors for the USLE were disaggregated for crop growth phases, and variability was introduced following Thomas, Snider, and Langdale. Twelve possible cropping systems were considered, consisting of combinations of two rotations—wheat-corn (3)-soybeans (WCCCS) and alfalfa (3)-corn (2) (AAACC), three tillage methods—moldboard plowing, till-planting, and no-till, and two mechanical practices—vertical plowing and contour plowing. These options are typical of the area, and the rotations make use of similar pesticides. Three pesticides were selected for study: Atrazine, Furadan, and Bladex. Atrazine and Bladex use does not vary with tillage practices, while Furadan is used in fewer years when tillage is reduced. Assumed crop prices were \$60 per ton for alfalfa hay, \$2.25 per bushel for corn, \$5.40 per bushel for soybeans, and \$2.30 per bushel for wheat.

Chemical toxicity data for salmonids were obtained from Mayer and Ellersieck and incorporated into habitat relationships using the techniques developed by Herricks and Braga. Physical suitability relationships were adapted from existing HSI models (e.g., Raleigh et al.).

Analysis

The SEDEC model was used to determine the economically optimal management practices for meeting sediment targets. The consequences for fish habitat suitability of the practices that optimally control sediment were determined using the extended model (without optimizing for suitability impacts). A similar approach was followed for erosion targets. The analysis also was performed in reverse—the optimal practices were determined with respect to suitability targets and the sedimentation and erosion consequences of those practices were traced. Finally, the subwatershed suitability target was applied to all individual catchments to assess the consequences of imposing uniformity throughout the stream reach.

While any suitability, sedimentation, or erosion levels could be selected for analysis, levels of 0.5, 0.7, and 0.9 were chosen here. These cover average to very good suitability conditions, on the assumption that poor conditions are not relevant environmental targets.

Results

The results are summarized as cost frontiers relating the minimum losses in farming profits associated with attaining particular environmental targets. The cost estimates do not reflect differences in farmer risks that may accompany different management systems; they assume that watershed management can be highly selective; and they assume all farmers would settle for the minimum compensation.

Figure 1 shows the cost curves for the two study sites assuming the extreme habitat suitability targets of 0.5 and 0.9 and allowing reliability to vary. The costs are per hectare for comparison, although the costs are borne unevenly across management units as a result of the optimization.

The curves are quite different for the two sites, and this is attributable to the different background conditions. The Galien site is already highly suitable and reliable for salmonids, while Pipestone is not. Thus, the costs are greater for attaining high reliability levels at Pipestone.

The curves for the 0.5 suitability level extend to higher levels of reliability than do the 0.9 curves. This suggests that the best practices for usual weather circumstances (that dominate the suitability determination) are not the same as the best practices for extreme conditions (that dominate reliability). Furthermore, conservative farming practices alone cannot achieve high levels of suitability with high reliability. The dual extremes would require either land use changes more substantial than those considered here or supplementary measures within the stream channels.

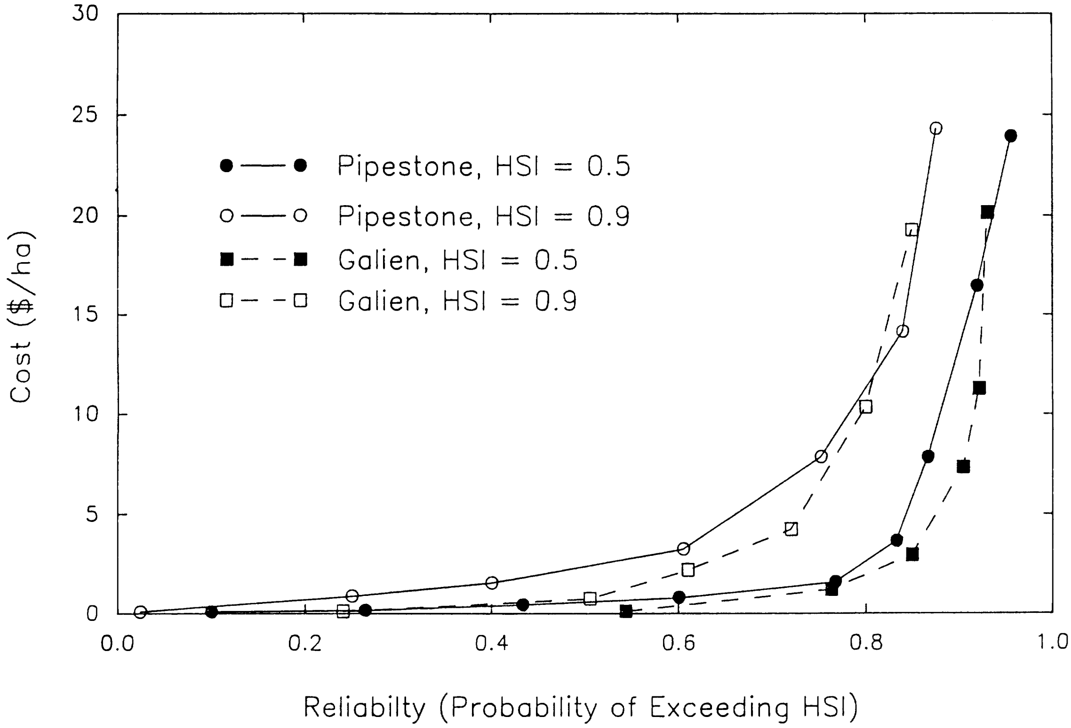

Figure 1. Minimum costs of achieving selected salmonid habitat suitabilities and reliabilities, Pipestone Creek and Galien River sites

The constraint on pesticide suitability is nonbinding at low levels of reliability. The pesticide constraint does not become binding until rather high probabilities of exceeding the target suitability levels are reached, at which point the risk of excessive sediment accumulation is relatively low. (The reliability level at which pesticides become important varies inversely with the suitability level). These findings are consistent with the consensus among fisheries biologists that deteriorated substrate conditions are most responsible for the general degradation of fish populations in midwestern streams (e.g., Smith).

For the comparison of targets, figures 2 through 5 display the cost-suitability frontiers for (*a*) targeting directly on suitability; (*b*) constraining the total sediment load in the watershed; (*c*) constraining the sediment load from each catchment; and (*d*) constraining the soil erosion on each LMU, and the frontier from targeting directly on suitability. The 0.5 and 0.9 levels of suitability illustrate extremes.

The figures show that a sediment target reasonably approximates a habitat suitability target only over a limited range. The approximation grows worse as pesticides play a greater role in suitability determination, i.e., at higher levels of reliability where the pesticide suitability constraint is controlling. Because the critical pesticides are in solution, and because sediment runoff is not necessarily correlated with runoff volume or concentration, "targeting" sediment is a poor way to deal with pesticide effects.

Comparisons of the figures suggests that the range of reasonable approximation shrinks as the suitability target is raised. This shrinkage occurs because the pesticide constraints bind at lower reliability levels when suitability targets are higher.

The sediment and erosion target curves in figures 2 through 5 are not smooth because some strategies (e.g., alfalfa rotations) used to control soil movement also lower pesticides while others (e.g., no-till) can increase pesticide concentrations in runoff. Erosion and sedimentation targets take no account of the pesticide consequences and result in higher costs and greater or lesser reliability depending on the nature of the sediment control regime.

Management Implications

Optimal management scenarios for the HSI target are summarized in table 1, and corresponding results for the alternative targets appear in

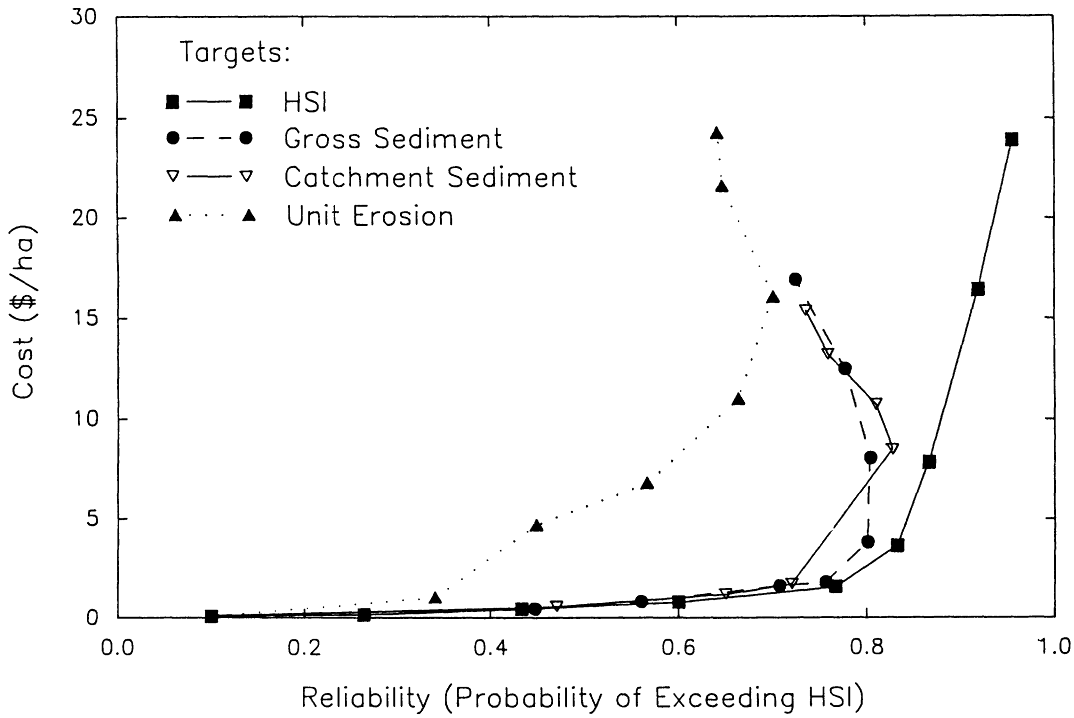

Figure 2. Cost of salmonid reliability with selected discharge targets and impacts, Pipestone Creek, HSI = 0.5

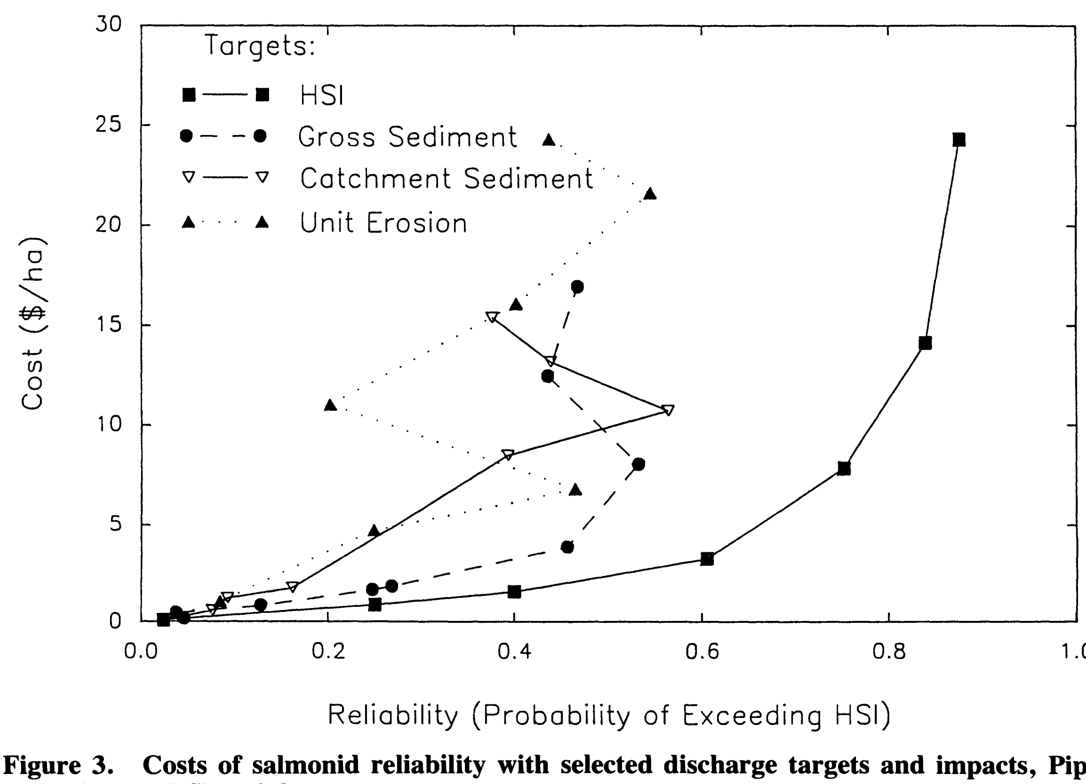

Figure 3. Costs of salmonid reliability with selected discharge targets and impacts, Pipestone Creek, HSI = 0.9.

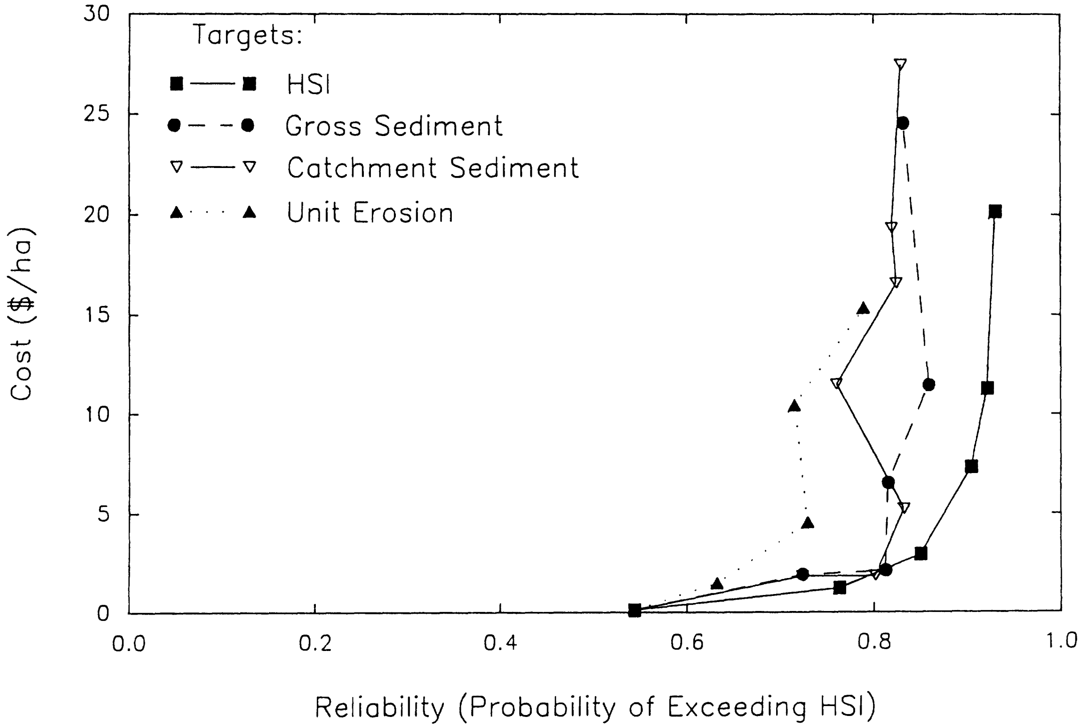

Figure 4. Costs of salmonid reliability with selected discharge targets and impacts, Galien River, HSI = 0.5

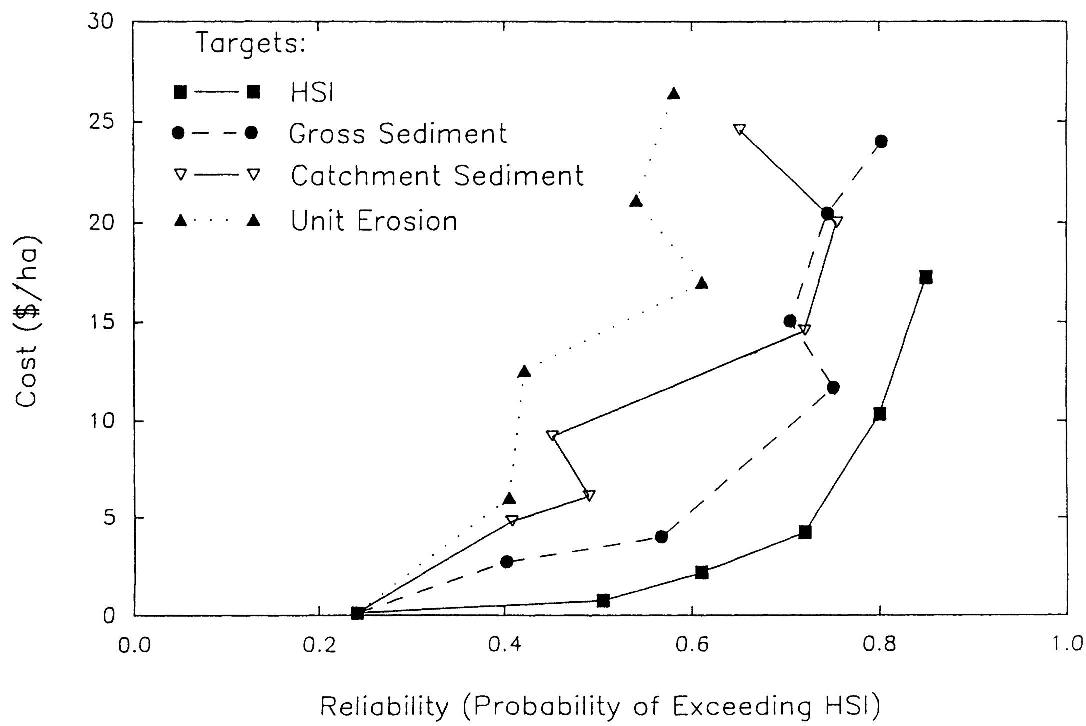

Figure 5. Cost of salmonid reliability with selected discharge targets and impacts, Galien River, HSI = 0.9

Table 1. Optimal Management Summaries for Selected Salmonid Suitability/Reliability Levels, Pipestone Creek and Galien River Sites

Salmonid HSI/Reliability	Management Practices: Rotation	Management Practices: Tillage	Management Practices: Mechanical	Extent of Management Changes		Cost
	(% ha WCCCS)	(% Mold-board)/ (% No-till)	(% Verticle)	(% Area)	(% units)	($/ha)
		Pipestone Creek				
Baseline	100	67/6	93			0.00
0.5/0.40	100	55/17	93	13	5	0.43
0.5/0.80	91	27/28	93	33	11	2.30
0.9/0.40	92	48/28	93	30	26	0.57
0.9/0.80	72	18/31	93	71	80	11.03
		Galien River				
Baseline	100	42/22	87			0.00
0.5/0.40	100	42/22	87	0	0	0.00
0.5/0.80	82	26/26	84	41	17	1.60
0.9/0.40	87	35/24	87	33	18	0.45
0.9/0.80	64	24/17	84	65	73	8.77

table 2. The selection of performance goals for table 2 was limited because some or all of the alternative targets could not achieve reliability of 0.8 at Pipestone with either the 0.5 or the 0.9 HSI, nor at Galien with 0.9 HSI.

In the baseline case, without habitat constraints, the WCCCS rotation and a combination of tillage practices are implemented at both sites. As indicated in table 1, tightening the habitat constraint initially (the 0.5/.40 case) prompts greater use of no-till WCCCS. Requiring reliability of .80 causes a shift away from no-till WCCCS and toward the AACCA rotation. The greater availability and concentration of pesti-

Table 2. Comparison of Watershed Management for Selected Alternative Pollution Abatement Targets and Impacts, Pipestone Creek and Galien River Sites

Impact HSI/Reliability	Target	Management Practices: Rotation	Management Practices: Tillage	Management Practices: Mechanical	Extent of Management Changes		Cost
		(% ha WCCCS)	(% Mold-board)/ (% No-till)	(% Verticle)	(% Area)	(% units)	($/ha)
			Pipestone Creek				
0.5/0.40	Gross Sed.	100	55/17	93	5	2	0.43
	Catch Sed.	100	53/19	93	7	3	0.50
	Unit Erosion	100	51/26	93	13	8	4.39
0.9/0.40	Gross Sed.	92	51/26	93	21	15	2.51
	Catch Sed.	93	53/31	93	24	19	7.87
	Unit Erosion	96	49/44	93	36	25	5.47
			Galien River				
0.5/0.80	Gross Sed.	100	42/20	87	2	1	1.73
	Catch Sed.	100	40/20	87	5	2	1.84
	Unit Erosion	100	35/28	87	12	21	2.63
0.9/0.40	Gross Sed.	89	44/32	87	27	8	2.72
	Catch Sed	90	41/33	87	31	12	4.81
	Unit Erosion	94	33/41	87	37	33	6.02

cides with no-till accounts for this shift. Tightening the constraints also requires that changes be made in more management units and more acres.

For each site, the mechanical practices change little or not at all with different constraints because contour and vertical plowing perform very similarly on the long gentle slopes of the sites.

In comparing tables 1 and 2, the erosion and sediment targets generally lead to more acreage in the WCCCS rotation and more no-till. (An exception to the no-till result appears in the Galien 0.5/.80 case, but more use of conservation tillage with the HSI/Reliability target accounts for this apparent anomaly.) These results are as expected and are more pronounced, respectively, for gross sediment, catchment sediment, and erosion, that is, as the target becomes further removed from fish habitat.

An unexpected result, at least for the sediment targets, is that less area and fewer management units are involved in the solutions, albeit at higher overall costs. An interesting implication is that if administrative costs increase with the area and number of farms involved in abatement actions, the ostensible efficiency gains of using suitability/reliability targets could be offset.

Conclusions

This study suggests that protecting fish habitat can be quite distinct from reducing agricultural erosion or sediment discharges. Policies that address sediment or erosion effectively are less effective in protecting habitat, especially at high suitability and reliability levels. This is because soluble pesticides dominate extreme suitability and reliability conditions, and the correlation to sediment loads is not high. This result is not surprising because fish respond to multiple qualities of the stream channel. Single-dimensional policies will be effective only if the dimension chosen is highly correlated with overall suitability.

A specific policy concern involves no-till farming. No-till has been widely encouraged. At least in the cases studied here, this approach appears sound with respect to erosion and sedimentation. But the consequences for fish, and perhaps other wildlife, may be perverse; no-till sometimes involves greater use of pesticides, which are not as fully incorporated, while it also reduces runoff volume. Nonincorporation means that less water will move more chemicals. The results point toward the desirability of no-till systems that better control pesticide releases.

Another policy issue involves the apparent desirability of heterogenous cropping systems in a watershed. When suitability and reliability goals are high, changes in tillage and mechanical practices are inadequate. Crop changes are needed (unless stream channel measures are undertaken), and the changes entail more diversity. More diversity reduces the probability of any one chemical exerting influence in a particular weather event. Agricultural policies favoring the cultivation of fewer crops may hamper efforts to attain high quality stream fisheries in some areas.

The results suggest that less area and fewer farms are affected by targeting on sediment than on suitability. Thus, the apparent disadvantages of sediment targets may be less pronounced when administrative costs are considered.

Finally, the differences between targets in the costs of attaining particular quality/reliability goals are the potential gains from intensive water quality monitoring and measurement to fine tune abatement efforts. Intensive programs will undoubtedly be quite costly. There is apparently little to gain when quality/reliability goals are modest or when existing conditions are generally good. The emissions or exposure targets perform reasonably well. For fish habitat purposes, the gains from intensive programs probably warrant the costs only where they stand a good chance of greatly improving habitat for high-value species.

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Colmation and Depth Filtration within Streambeds: Retention of Particles in Hyporheic Interstices

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Colmation and Depth Filtration within Streambeds: Retention of Particles in Hyporheic Interstices

key words: colmation, depth filtration, sediment, fine particles, streambed

Abstract

Colmation refers to the retention processes that can lead to the clogging of the top layer of channel sediments and decolmation refers to the resuspension of deposited fine particles. Internal colmation, clogging of the interstices directly below the armor layer, may form a thin seal that disconnects surface water from hyporheic water by inhibiting exchange processes. The settling of particles under low flow conditions can cause external colmation. Colmated channel sediments are characterized by reduced porosity and hydraulic conductivity as well as by a consolidated texture. The term 'depth filtration' refers to the transport and storage of fine matrix sediments in interstitial layers. Depth filtration is of significance for the transport of colloidal and fine particulate inorganic as well as organic matter within the hyporheic interstices and into the alluvial aquifer. The role of depth filtration is assessed for the content (given in mg per liter) of matrix fine particles retained in the coarse framework sediment of a gravel-bed river in Switzerland. Sediment samples were taken by freeze-coring with liquid nitrogen down to 70 cm depth and by piezometers down to 150 cm depth. Seventy-two percent of the mobile matrix fine particles were smaller than 0.1 mm and 50% were smaller than 0.03 mm. The content of fines tended to increase with depth, although higher accumulations were found at intermediate depths in sediments influenced by exfiltrating ground water. Interstitial detrital particles >90 μm showed vertical distribution patterns opposing those of total particles. These relationships revealed a differential significance of import, storage, and transport within three types of hydrological exchange zones (infiltration, horizontal advection, exfiltration) in the cross-section of the stream.

1. Introduction

Hyporheic interstices are the connecting ecotone between river and groundwater ecosystems and mediate the exchange processes between both of them. The permeability of this ecotone depends on the hydraulic conductivity of the sediment layers (BRUNKE and GONSER, 1997). The sediment of gravel-bed rivers can be separated into two components, the framework gravel and the fine inorganic matrix particles (PETTS, 1988). The proportion of these fine inorganic particles (<2 mm: sand, silt, clay; PETTS, 1988; LISLE, 1989), henceforth referred to as 'fines', is decisive for the hydraulic conductivity.

Fines suspended in the flowing surface water may intrude into stable gravel-beds and progressively reduce pore spaces, thereby causing decreasing seepage rates. This affects the metabolism of fluvial hydrosystems and the habitat quality for fish and aquatic invertebrates (BRUNKE and GONSER, 1997; MILAN and PETTS, 1998).

The local retention and transport of particles <2 mm in rivers are determined by flow conditions (shear stress, depth, kinematic viscosity, density, hydraulic gradient), by properties of the suspended load (grain size distribution, concentration, shape, settling properties, cohesivity), and by the channel sediment structure (grain size distribution, texture; GELDNER, 1982; REYNOLDS *et al.*, 1990; BETTESS, 1992; CARLING, 1992). According to JOBSON and CAREY (1989) it is usually assumed that a freely flowing stream is capable of transporting all particles <62 μm which are imported to it.

Sediment in suspension can interact with the streambed in several ways (Fig. 1):

1 Sediment may settle on the top of the streambed in areas of low water velocity (e.g. in pools, or between coarser gravels) under low flow conditions. Silt may be trapped within the structural matrix of epilithic periphyton even in turbulent water (GRAHAM, 1990).

2 A thicker layer of fine particulate matter, that reduces the permeability of the streambed (external colmation), may develop after an extended period of low current velocity (BEYER and BANSCHER, 1975). These clogging processes can also be induced by algal mats in eutrophic streams as well as by cohesive depositions in rivers receiving sewage effluents (KUSTERMANN, 1962; BEYER and BANSCHER, 1975).

3 Fine sediment that passes the coarse armor layer may accumulate beneath the armor layer; if low discharge continues a compact layer that reduces the porosity and hydraulic conductivity of the streambed may develop and stabilize the streambed against erosion (internal colmation) (BEYER and BANSCHER, 1975; SCHÄLCHLI, 1993).

4 Fine particles that penetrate the armor layer but do not contribute to the clogging of the top layer may undergo alternating phases of deposition and resuspension within the interstices (depth filtration).

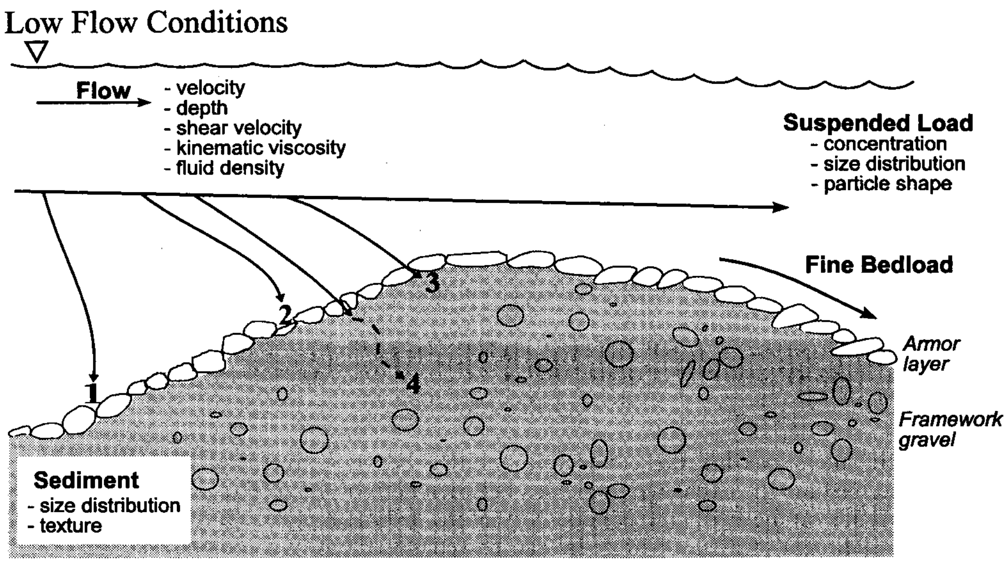

Figure 1. Factors that influence the retention of fine particles in streambeds. Numbers 1–4 are described in the text.

The objectives of this article are two-fold. The first part provides an overview on the ecological significance of fine matrix particle transport within fluvial sediments and on the mechanisms underlying deposition and transport processes. The second part focuses on the characterization and distribution of mobile matrix fines subjected to depth filtration within the interstices of the Töss River, Switzerland.

1.1. Ecological Significance of Colmation and Depth Filtration

1.1.1. Colmation

In general, colmation lowers the exchange processes between a river and the adjacent ground water. The alternating phases of colmation during low flow conditions and the decolmation induced by a high discharge are natural processes of sedimentation and erosion. However, the balance between colmation and decolmation may be altered anthropogenically towards an enhanced siltation, e.g. by flow regulation (PETTS, 1988). In the Rhine River a mechanical opening of a colmated reach near a drinking water filtration site induced a 1 m rise of the ground water table near the river, but after a few weeks the opened section was sealed again (GÖLZ *et al.*, 1991). River bank storage is a component of the ground water budget as well as of the riverine discharge regime (BAUMGARTNER and LIEBSCHER, 1990; SQUILLACE, 1996). Therefore, if infiltration is inhibited, floodpeaks that cannot erode consolidated channel beds (REID *et al.*, 1985) are not diminished. A sealed bed can act as an intrusion barrier that prevents the contamination of ground water by polluted surface water (YOUNGER *et al.*, 1993; KOMATINA, 1994).

Increased clogging threatens the reproductive success of fish spawning on gravel (BERKMAN and RABENI, 1987; CHAPMAN, 1988; ZEH and DÖNNI, 1994). Sealed interstices cannot function as nurseries for aquatic insects (GAMMETER, 1996). Furthermore, siltation reduces the refugial space available to invertebrates, and thus the impacts of natural and anthropogenic disturbances, such as urban stormwater runoff, are magnified (BORCHARDT and STATZNER, 1990). Increased loads of sand may affect the components of benthic communities differentially. It has been shown that small sized animals and the taxonomic groups Ephemeroptera, Diptera, and Coleoptera exhibit stronger declines in abundance compared to other taxa as the proportion of sand increases (ALEXANDER and HANSEN, 1986).

1.1.2. Depth Filtration

The fine matrix sediment proportion of total bed sediments is increased by depth filtration. The fines enlarge the surface area that can be colonized by a biofilm. On the other hand, the interstitial throughflow and the concomitant delivery of resources, as well as the usable pore space for the interstitial fauna can be reduced. Excessive siltation has impacts on the colonization dynamics of interstitial animals (RICHARDS and BACON, 1994; GOVEDICH *et al.*, 1996; MARIDET *et al.*, 1996).

Colloidal particles contribute significantly to subsurface transport processes as carriers (DVWK, 1992) since they are important adsorbents for metals, phosphates, humic acids and organic compounds (EGGLESTON *et al.*, 1991). Trace metals adsorped to particles may accumulate in the interstitial habitat (PETTS *et al.*, 1989; GIBERT *et al.*, 1995). Suspended bacteria may be interpreted as living colloids (VAN LOOSDRECHT *et al.*, 1990). Their transport and initial adhesion is controlled by the structure and physicochemical characteristics of the surfaces, especially by the hydrophobicity (LINDQVIST and BENGTSSON, 1991; BOSMA and ZEHNDER, 1994). Algae that are transported into the subterranean water may remain viable; they are protected during unfavorable periods and may even reproduce in some cases (POULÍCKOVÁ, 1987; WASMUND, 1989).

1.2. Overview of Colmation and Depth Filtration

1.2.1. Mechanism of Filtration

According to HERZIG *et al.* (1970) and BEYER and BANSCHER (1975) two types of filtration can be defined in natural systems: Large fine particles with diameters >30 µm are subjected to a mechanical filtration, whereby size and shape prevail in importance over surface phenomena such as positive and negative charges. Smaller particles (diameter about 1 µm) are mainly subjected to physicochemical filtration; surface phenomena prevail over volume phenomena. Medium size particles between 3 and 30 µm are retained by both filtration mechanisms. Adhesion of colloidal particles and bacteria is exclusively due to physicochemical processes (HERZIG *et al.*, 1970; VAN LOOSDRECHT *et al.*, 1990) (Tab. 1).

Table 1. Mechanisms of filtration according to HERZIG *et al.* (1970) and BEYER and BANSCHER (1975).

Particle Size	>30 µm	~1 µm	<0.1 µm
Filtration Type	mechanical	physicochemical	colloidal
Retention Sites	constrictions, crevices, caverns,	surface sites	surface sites
Retention Forces	friction, fluid pressure,	Van der Waals forces, electrokinetic forces,	Van der Waals forces, electrokinetic forces, chemical bonding
Capture Mechanism	sedimentation, interception,	interception	interception, diffusion
Remobilization	alterations in flow direction	increase in flow rate	increase in flow rate

The following formula describes the effectiveness of mechanical depth filtration in relationship to the *effective pore diameter* of a filter medium, which is about *D15/5*[1]:

$$\mathrm{D15_{particle}} < D15_{filter}/5 < \mathrm{D85_{particle}}$$

For a retention of fine particles the effective pore diameter of the filter must be smaller than D85 of the fines, otherwise fines will pass through the filter. Furthermore, for a retention within the filter column, the effective pore diameter must to be larger than D15 of the fines (SOWERS and SOWERS, 1970; SHERALD *et al.*, 1984). If the effective pore diameter is smaller than D15 of the fines, they would accumulate on the top of the filter.

1.2.2. The Evolution of Colmation

The formation of clogged interstices depends on the size distributions of suspended sediment, fine bedload, and channel sediments. SCHÄLCHLI (1993) developed a model of the evolution of colmation in which he distinguished three phases by using the different filtration mechanisms of HERZIG *et al.* (1970) (Tab. 2). During each of these phases the retention of

[1] D15 represents the particle diameter for which 15% of the sediment is smaller

a different particle size fraction is decisive for the process. Larger fine particles (>30 μm) are essential for an initial bridging of the pores, but during this first phase the hydraulic conductivity remains largely unaffected. This phase can be rapidly completed. Fine sand may be under-represented during low flow conditions with a minor transport capacity, and therefore a depth filtration of particles may prevail. However, in most streambeds a sandy fraction exists, which has been deposited contemporaneously with coarser framework particles (LISLE, 1989).

Table 2. Processes and phases of internal colmation according to SCHÄLCHLI (1993).

	Large Particles (∅ > 30 μm)	Medium Particles (∅ 3–30 μm)	Small Particles (∅ < 3 μm)	Permeability of the Streambed
Phase **I**	*decisive process:* clogging of larger pores directly between and below the gravels and stones of the armor layer	deposition mainly in the pores of the upper subarmor layer; transport into deeper strata is possible (depth filtration)	partial deposition on surfaces due to physicochemical interactions; transport into framework gravel (depth filtration)	minor reduction of hydraulic conductivity
	↓	↓	↓	
Phase **II**	filling of some voids, limited deposition on the armor layer; role of large particles is largely terminated	*decisive process:* mechanical clogging of fine pores; sedimentation in voids with low current	attachment on substrate of the filter layer effects further narrowing of pore channels	substantial reduction of hydraulic conductivity
	↓	↓	↓	
Phase **III**	only minor deposition	lessoned deposition between larger particles	*decisive process:* further attachment effects a reduction of the interstitial velocity to a lower limit due to a decreased import of particles into the filter layer	hydraulic conductivity reaches a minimum value

Commonly fluvial sediments have a bimodal grain size distribution (PETTS *et al.*, 1989) characterized by the absence of the fine gravel and coarse sand. The fine material is generally missing in unimodal open framework gravel (HUGGENBERGER *et al.*, 1988), e.g. of fresh spawning redds (KONDOLF *et al.*, 1993). Such sediments are rapidly filled by fines from the bottom up (EINSTEIN, 1968; BESCHTA and JACKSON, 1979; CARLING, 1984).

During the second phase the straining and sedimentation of intermediate sized particles (3–30 μm) effects a significant reduction of the streambed's hydraulic conductivity, while the contribution of larger particles on the clogging process decreases continuously. The small particles (<3 μm) still penetrate the sediment layers in which larger particles are strained (filter layer) or may adhere to surfaces due to physicochemical interactions. Below this filter layer the small particles are subjected to depth filtration.

In the third and final stage of colmation the small particles settle in the filter layer and the pore spaces are reduced to a minimum. Then the hydraulic conductivity reaches a certain minimum value (BANSCHER, 1976), due to an inhibited import of fines in the filter layer by lowered advection. The minimum value of hydraulic conductivity is maintained by an equilibrium of deposition and resuspension of particles at the top of the filter layer (SCHÄLCHLI, 1993).

1.2.3. Factors that Influence Colmation

The temporal development of colmation, the clogging depth, and the lower limit of the decreased hydraulic conductivity are influenced by interactions of several factors (Tab. 3). Low flow conditions, suspended particles and fine bedload are requirements for the clogging process. A key parameter is the *dimensionless shear stress* (Θ, Theta) (often termed Shields factor) (SCHÄLCHLI, 1993) as the force ratio between shear stress promoting entrainment of particles and the particle size, density and gravity resisting entrainment (CARLING, 1992):

$$\Theta = \tau / (p_s - p)\ g\ D$$

and

$$\tau = p\ g\ R\ S$$

where Θ is the dimensionless shear stress; τ is the shear stress; p_s is the density of the sediment; p is the density of water; g is the accelaration due to gravity; D is the grain size; R is the hydraulic radius and S is the slope.

A dimensionless shear stress below a *critical* dimensionless shear stress (Θ_k of the beginning of decolmation, which is about 0.05) is a prerequisite for the occurrence of colmation. If Θ is below a certain minimum value, all suspended particles will be deposited. When Θ is above this minimum value only a small fraction of the fines will settle and the rest remain in suspension. At a Θ near the Θ_k, colmation converts to decolmation. Thus, for a theoretically constant Θ the concentration of suspended particles and their deposition attain an equilibrium. Below a Θ_k increasing Θ effect a more dense packing of the sediment due to higher turbulent pulses and vibrations of the framework gravel (SCHÄLCHLI, 1992, 1993). Therefore, higher Θ (below Θ_k) accelerate the development of sealing and lower the minimum of the hydraulic conductivity.

The grain size distribution is of overriding relevance for the intrusion of fines into the bed sediments (BESCHTA and JACKSON, 1979). The transport into the interstices of large and medium sized particles, once they have passed through the surface, is largely determined by the pore sizes (FROSTICK *et al.*, 1984). Well-sorted gravel with large pores promote a deep intrusion of fines into the bed sediments, and thus more matrix sediment can accumulate increasing the clogging depth and lowering the minimum of hydraulic conductivity (BEYER and BANSCHER, 1975). In contrast, poorly sorted framework sediment is characterized by small pore sizes, which induce a straining of fines within a thin layer. Therefore, in such sediment less fine material is deposited, the clogging depth is comparatively shallow and the minimum of hydraulic conductivity is not raised compared with well-sorted gravel (SCHÄLCHLI, 1993). However, even in poorly sorted sediment macropores may exist which can enable the transport of fines into deeper layers. In streams with a coarse armor layer and a finer subarmor layer internal colmation develops directly below the armor layer in a thin stratum due to clogging of the small pore spaces of the subarmor layer (FROSTICK *et al.*, 1984; CUNNINGHAM *et al.*, 1987).

Table 3. Summary of factors that influence the temporal course of colmation, the clogging depth, and the minimum value of decreased hydraulic conductivity (after (BEYER and BANSCHER, 1975; SCHÄLCHLI, 1993) (C is the concentration of suspended particles, Re is the Reynolds number).

Primary Influence	Variable	Key Parameter	Relevance
Flow	– current velocity – dimensionless shear stress	Θ	requirement: $\Theta < \Theta_{k;}$ low
Suspended Particles	– concentration – size distribution – shape – adhesion, cohesion	C/p_w	requirement: $C > 0$
Sediment	– size distribution – armor layer – texture	d_{10}/d_m	high
Hydraulic Gradient	– infiltration – exfiltration	– VHG + VHG	medium decolmation at $+VHG > +VHG_k$
Temperature	– kinematic viscosity	Re	low
Secondary Influence	**Variable**		
Morphology	– riffle-pool sequences – longitudinal and cross profile – zones of preferential bedload movement	VHG, Θ, D_{10}/D_m	variable

In the study by CUNNINGHAM *et al.* (1987) the suspended particle concentration in a range of 200–1600 ppm did not have a significant effect on the degree of colmation. This is because the particle concentration does not influence the location of deposition within the filter layer (SCHÄLCHLI, 1993). However, suspended particles and fine bedload are a prerequisite for the clogging process and higher concentrations of such material can accelerate the temporal development of colmation (BANSCHER, 1976; CARLING, 1984; DIPLAS and PARKER, 1992).

Compared with other factors the vertical hydraulic gradient (VHG) is of intermediate relevance for the development of a colmation (SCHÄLCHLI, 1993). Influent conditions induce higher interstitial current velocities and particles may be transported deeper into the bed sediment. Thus, more fines can be deposited resulting in a thicker colmated layer, and thereby effect a low hydraulic conductivity (SCHÄLCHLI, 1993). Effluent conditions reduce the deposition of very fine particles and only large particles intrude into interstices, which exert a minor effect on hydraulic conductivity. However, the permeability of an already clogged streambed can only be re-established locally when the exfiltration reaches site-specific threshold levels, whereby fines are flushed out from local pore channels (BANSCHER, 1976).

Increasing water temperature affects colmation in a similar way as infiltration; the reduction of the cinematic viscosity promotes higher interstitial velocities. However, the relevance of temperature for colmation is comparatively low (SCHÄLCHLI, 1993).

In conclusion, the colmation depth depends mainly on the grain size distribution of the bed sediment. SCHÄLCHLI (1993) derives an empirical formula from his flume experiments, which can be used to roughly estimate the depth of the clogged layer (D_c) in gravel-bed streams:

$$D_c = 3D_m + 0.01 \text{ [m]}$$

where D_m is the mean grain size.

A variable fluvial geomorphology integrates all these factors on a reach scale. Riffle-pool sequences induce variable flow patterns with alternating local up- and downwelling zones (SAVANT *et al.*, 1987; HARVEY and BENCALA, 1993), and heterogeneous grain size distributions and textures (CARLING, 1992). Complex flow patterns with high variability due to discharge fluctuations create different scour and depositional areas in a longitudinal and cross-sectional profile (CHURCH, 1992). Thus, siltation may be highly variable locally resulting in a three-dimensional mosaic of differentially colmated areas within the streambed. The import of larger fine particles is increased in areas of preferential bedload movement and higher surface velocities (FROSTICK *et al.*, 1984), presumably because of fine bedload (i.e. larger fine particles) that has a higher probability of intrusion into the interstices than the suspended load (LISLE, 1989) and an increased flushing of small particles relative to areas not situated in the thalweg (FROSTICK *et al.*, 1984). However, the effect of imported larger fine particles may be small, since the hydraulic conductivity of the streambed is not reduced significantly.

Low flow conditions enable external colmation processes, characterized by a deposition of fines directly on the streambed. External colmation prevents an intrusion of fines into the channel sediments and thus constrains the development of internal colmation (BANSCHER, 1976).

1.2.4. Decolmation

Decolmation refers to all processes that contribute to an increase of hydraulic conductivity and to a breaking up of the bed sediment structure. SCHÄLCHLI (1992, 1993) distinguished different phases of decolmation according to dimensionless shear stress: (I) At $\Theta > 0.05$ fine bedload transport begins, which may induce a partial decolmation by jostling deposited fines and removing them or, depending on the stage of colmation, fill up larger pores, e.g. of open framework gravel. (II) During a transitional phase of Θ values between 0.06 and 0.072 the hydraulic conductivity of the top layer can increase by a factor of 10 due to a flushing out of fines below and beside the coarser gravels of the armor layer and by removing individual components of the armor layer. (III) During the flushing phase (Θ between 0.072–0.078) at high flow conditions the armor layer breaks up locally and hydraulic conductivity increases to a maximum value. However, the development of new colmation is possible when discharge remains constant and the suspended load is high. (IV) At peak flows when $\Theta > 0.08$ the whole river bed is mobilized and consolidated channel beds are broken up.

1.2.5. Depth Filtration

Depth filtration refers to particle separation by selective straining and transport within porous media. In riverine sediments and aquifers it means transport and retention of fine inorganic and detrital particles. Moreover, fine inorganic particles may be colonized by microorganisms and contain adsorbed compounds on their surfaces. The mechanisms underlying depth filtration correspond to those occurring during colmation. It must be stressed that the role of the cohesive properties of organic matter for filtering are difficult to assess and have rarely been addressed in experiments. Since organic layers have a porosity of about

90% (CHARACKLIS, 1984), they can increase the particle size of fines enormously. However, organic layers on fines and detrital particles may have a high plasticity in form and thereby pass pore channels differentially depending on interstitial throughflow.

2. Distribution of Fine Particles within the Interstices of a Calcareous Gravel-Bed River

Very little ecological research has been conducted on fine matrix sediments, regarding their influence on interstitial organic matter and organisms, even though they constitute the ecologically most important inorganic sediment fraction in gravel-bed rivers (JOBSON and CAREY, 1989; BRETSCHKO, 1991; WARD *et al.*, 1998). For example, in two gravel-bed streams in Austria and Switzerland particles smaller than 1 mm contributed to only 6–9% of the total sediment, yet up to 88% of the total organic carbon (TOC) and total organic nitrogen (TON) content was associated with this component (LEICHTFRIED, 1988; EGLIN, 1990). The percentage of particles <1 mm can be used as an indicator of habitat quality (ADAMS and BESCHTA, 1980).

For a characterization of the mobile matrix fines the size distribution of this fraction must be examined separately from the total bed sediment. Theoretically, the interstitial content of the mobile matrix fines should increase with sediment depth simply because of gravity. It is hypothesized that the proportion of large particles (>30 μm) should decrease with depth due to a straining in the topmost layer of the streambed, whereas the content of small particles (<2 μm) should not change with depth, because similar physicochemical surface properties of the sediments were assumed. Therefore, the proportion of small particles should increase with depth.

Interstitial detrital particles >90 (DPs) of allochthonous origin (i.e. mostly plant material) were used as tracers of interstitial transport of organic matter. It was predicted that the interstitial content of DPs decrease with depth, because of continuous mineralization and ingestion by interstitial animals. Furthermore it was tested if the DP content differs between three types of hydrological exchange between the stream and the interstices, i.e. infiltration of surface water, exfiltration of ground water, and prevailing subsurface flow along the channel (horizontal advection).

2.1. Objectives

Specific objectives were: (1) to characterize the mobile matrix fines by their grain size distribution, (2) to examine the vertical distribution of three grain size classes differing in their type of filtration, (3) to examine the vertical distribution of detrital particles, and (4) to test the influence of infiltration, horizontal advection, and exfiltration on the vertical distribution of detrital particles and all fine patricles.

3. Study Site, Materials and Methods

The study was carried out in a transect across a calcareous prealpine gravel-bed river (Töss River, 460 m a.s.l., Switzerland). Due to hydro-engineering the former braided channel has been straightened and no riffle-pool morphology exists; the present channel width is about 20 m. At the study site distinct exfiltration zones (EZ) are located on the stream's left side, whereas on the right side surface water infiltrates into the alluvial sediments (infiltration zone; IZ). In midstream neither infiltration nor exfiltration prevail, but a horizontal advection dominates the throughflow within the sediments (horizontal advection zone; HZ).

Interstitial water was extracted by steel piezometers (internal diameter = 50 mm), from 4 sediment depths (20, 50, 100, 150 cm). Piezometers were spatially organized as clusters located in each of the three hydrological exchange zones. The piezometer nests within the IZ were located 3 m from the right bank, those within the HZ were 10 m from both banks in midstream and those within the EZ were 3 m from the left bank. Total matrix particles (i.e. inorganic fine particles and particulate organic matter, given as total particle content in mg per liter of interstitial water) were collected with a submersible electric pump after taking a sample with a hand pump for faunistic investigations (BOU and ROUCH, 1967; HUSMANN, 1971). Sampling dates were on 15 occasions between May 1995 and November 1996. The samples were filtered through Whatman GF/F filters. The filters were dried to constant weight at 60 °C and weighed. The grain size distribution of the fine particles from 12 samples were measured with a SediGraph 5100 (Micrometrics). The detrital particles (DPs) were collected on 3 occasions in July, September and November 1996. Ten liters of interstitial water were taken with the hand pump and filtered through a 90 μm net. The DPs were extracted from the sample by a flotation technique along with the interstitial animals (DANIELOPOL, 1976; DOLE-OLIVIER and MARMONIER, 1992a). In another step DPs and animals were separated by removing the animals individually. The DPs were separated into size classes >300 μm and 300–90 μm. The allochthonous origin of the DPs from plant material was determined by means of cellular structures with a microscope. The DPs were measured as ash free dry mass by loss on ignition at 500 °C for 3 hours.

Additionally, 9 freeze cores (BRETSCHKO and KLEMENS, 1986) were taken on November 1996, three of them in each hydrological exchange zone to measure the size distribution of framework gravel and to assess sediment porosity and hydraulic conductivity. Cores were taken to a sediment depth of 70 cm and cut into 10 cm sections. Framework grain size distribution was determined for each depth layer by passing the sediment through a set of seven sieves (mesh sizes >20, 12.5, 8, 6.3, 2.5, 1, 0.063 mm). Rocks greater than 50 mm were excluded from the analyses as recommended by ADAMS and BESCHTA (1980) to reduce the effect of extreme values for relatively small sediment samples. The sorting coefficient $(D75/D25)^{-0.5}$ was calculated according to SCHWOERBEL (1994). Porosity was calculated as (total volume – grain volume) × 100/(total volume) according to EGLIN (1990). Hydraulic conductivity was calculated according to BEYER (1964) and HÖLTING (1989). A few depth layers did not freeze on some cores presumably due to high interstitial currents. Therefore, it was not possible to statistically compare framework characteristics of depth layers between hydrological exchange zones. The grain size distribution of the armor layer was assessed with two methods: (a) by measuring the intermediate axis (b-axis) of 200 mineral particles >1 cm along a straight line in flow direction in three replicates according to FEHR (1987) and (b) by taking 6 sediment samples down to a depth of 10 cm using a Surber sampler (400 cm²) with a 90 μm net.

4. Results and Discussion

4.1. Bed Sediment Characteristics

The porosity of the bed sediments of the Töss River tended to increase with depth (Fig. 2), though this was not statistically significant. Porosities ranged between 6.2% and 32%, with an average of 18.8% (± 6.6 SD). The Thur and Neckar Rivers, two other calcareous gravel-bed rivers in Switzerland, had similar sediment porosities (EGLIN, 1990; NAEGELI, 1997). In the Lunzer Seebach (Austria) porosity is about 24% (BRETSCHKO, 1991).

The D50 of the surface sediment (i.e. the grain diameter at which 50% of the sediment in weight is smaller) was 31.0 mm (±2.6 SD) measured by b-axis and 32.2 mm (±6.1 SD) measured by sieving. It was coarser at the stream margins than in midstream (Tab. 4). Not shown by the grain size analyses was that the grains in midstream tended to be more rounded and those at the margins tended to be flatter. The reason for this difference was probably that the midstream sediments were more often affected by bedload movement, where water velocities were highest in the straightened channel. Furthermore, the layering of the flatter gravels at the margins resulted in a more erosion resistent armor layer. River engineering and lateral embankments in the whole catchment of the Töss River have reduced sediment supply, which leads, in conjunction with an increased transport capacity by the straightening of the river course, to a deficiency of bedload. In flume experiments DIETRICH *et al.* (1989) found that a reduction in sediment supply resulted in an expansion of lateral coarser 'inactive' zones in which little or no transport took place compared to finer 'active' zones in midstream. In the Töss River this is presumably a self enhancing process, since the longer a surface layer is not mobilized, the more it consolidates and becomes erosion resistent (REID *et al.*, 1985).

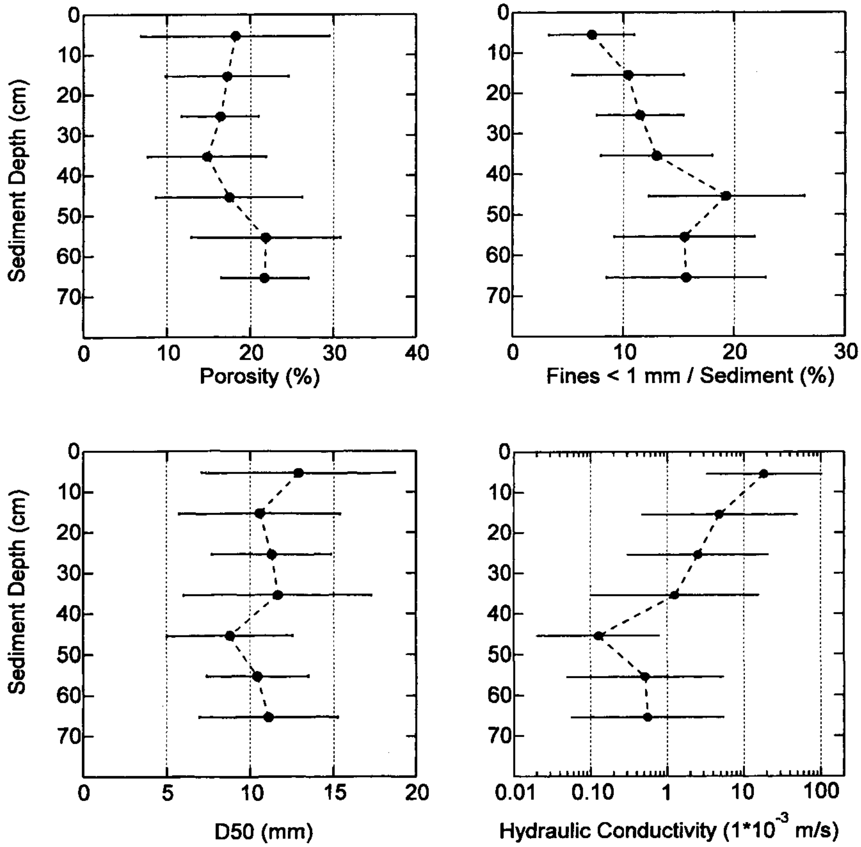

Figure 2. Depth profiles of sediment porosity, the proportion of fine particles <1 mm in the sediment, the D50 (mm) and the hydraulic conductivity. Means (±SD). n = 38.

Table 4. Sediment characteristics of the three positions within the channel cross-section. Means (±SD) for three freeze cores in each zone. Grain size metrics in mm.

	Infiltration zone (right margin)	Horizontal advection (midstream)	Exfiltration zone (left margin)
$D50_{surface}$ (b-axis)	36.5 ± 2.9	24.9 ± 1.8	32.2 ± 2.8
$D50_{surface}$ (sieving)	38.5 ± 6.3	26.5 ± 0.7	31.5 ± 0.7
$D75_{subsurface}$	16.5 ± 2.7	15.5 ± 2.3	16.2 ± 2.5
$D50_{subsurface}$	11.0 ± 3.6	10.3 ± 3.1	10.9 ± 4.1
$D25_{subsurface}$	3.4 ± 3.4	3.1 ± 1.4	4.4 ± 2.9
$D15/5_{subsurface}$	0.41 ± 0.33	0.21 ± 0.16	0.31 ± 0.45
So $(D75/D50)^{-0.5}$	2.4 ± 1.0	2.3 ± 0.45	2.9 ± 1.6
Porosity (%)	18.8 ± 7.8	19.7 ± 6.9	18.0 ± 5.7
$kf_{0-70\,cm}$ (1×10^{-3} m/s)	16.9 ± 34	3.7 ± 6.8	11.2 ± 28
$kf_{0-20\,cm}$ (1×10^{-3} m/s)	48.5 ± 63	9.2 ± 6.4	29.3 ± 51
$kf_{20-50\,cm}$ (1×10^{-3} m/s)	6.2 ± 8.4	5.5 ±10.6	1.3 ± 1.4
vertical $vf_{0-20\,cm}$ (1×10^{-3} m/s)	−0.92 ± 63	0.12 ± 6.4	0.44 ± 51
vertical $vf_{20-50\,cm}$ (1×10^{-3} m/s)	−0.94 ± 8.4	−0.09 ± 10.6	0.04 ± 1.4

The subsurface sediment showed a finer grain size composition than the surface sediment, with a D50 = 11 mm (±3.6 SD), but ranging between 4.5 and 17 mm. The difference in D50 values between midstream and marginal sediments is reduced (Tab. 4), due to a higher proportion of sand in the subsurface sediments. SCHÄLCHLI (1993) collected larger amounts of sediment with a different technique from an upstream site of the Töss River and calculated a D50 of 16 mm, presumably for a mixture of surface and subsurface sediment. The technique of freeze coring underrepresents the coarse fraction where there are cobbles and gravels (discussion by G. E. PETTS in CHURCH *et al.*, 1987). SCHÄLCHLI (1993) calculated a porosity of 25% and hydraulic conductivities ranging between kf = 1.5×10^{-3} and 2.5×10^{-4} m/s. In this study, hydraulic conductivities ranged between 1.5×10^{-5} and 1.2×10^{-1} m/s and changed significantly with depth (Kruskal-Wallis test, $p < 0.05$) (Fig. 2). Hydraulic conductivity decreased continuously down to the 40–50 cm depth stratum, thereafter it increased again. On average the sediment down to 40 cm was 'highly permeable', whereas the deeper strata were 'permeable' according to HÖLTING (1989). The specific flux (vf) was calculated for the upper sediment strata by using the average kf-values and the average vertical hydraulic gradients measured between April 95 and November 96 on 22 occasions (BRUNKE and GONSER, in prep.) (Tab. 4). These calculations indicate that in the IZ water can infiltrate down to a depth of 20-cm with an average velocity of 5.5 cm/min and in the EZ water can exfiltrate with an average of 2.6 cm/min. Therefore, by using the porosities for the IZ and EZ, 10 liters per minute could infiltrate and 4.7 liters per minute could exfiltrate on a square meter. At the sampling site the IZ and EZ are both approximately 4 m x 24 m in size, each. Therefore, about 47 liter m^{-1} s^{-1} are exchanged throughout the sampling site, contributing to 0.67% of the annual mean surface discharge of 7 m^3/s. Assuming hydrological exchange along the river course is the same as at the study site, the entire stream water would be exchanged on a channel length of 3.57 km. With an average flow velocity of 0.67 m/s complete exchange would take 1.5 hours. However, these calculations are just rough estimations, since they are based on averages and approximate calculations of hydraulic conductivity. Data from the HZ were not included in the calculations, though it contributed actively to the exchange since temperature variations can be detected down to a depth of 150 cm. Therefore, these estimates are likely to underestimate the water exchange between the stream and the subsurface water.

According to SCHÄLCHLI's (1993) formula a thin colmated layer of the topmost streambed would be located at a depth of about 10 cm. Severe colmation only occurred at the channel margins that were rarely affected by bedload movement.

The proportion of fines <1 mm in the sediment increased with depth, from 7.3% (±3.2 SD) at 10 cm to 16% (±6.7 SD) at 70 cm (Fig. 2). ADAMS and BESCHTA (1980) and MILAN (1996) also found lower levels of fines near the bed surface. ADAMS and BESCHTA (1980) assumed that this might reflect a paucity of fines or the presence of an armor layer. In their study they had an average of fines of 17.4% at 0–10 cm, 22.3% at 10–25 cm, and 22.2% at 25–40 cm sediment depths, which were clearly higher percentages of fines than in this study. The vertical trend for fines was not reflected in the porosity; porosity and fines were not correlated. MARIDET *et al.* (1992) concluded in their study, that only a high content of fines always leads to a low porosity, whereas porosity may be high or low when the content of fines is low. However, if the infiltrated sediments are a clay then the porosity can remain high.

The sorting coefficient was highly correlated to the percentages of fines <1 mm and <2 mm ($r_s = -0.86$, $p < 0.001$ and $r_s = -0.95$, $p < 0.001$, respectively). Therefore, the depth profile of the sorting coefficient resembles the depth profile of the percentage of fines <1 mm in the sediment (Fig. 2).

4.2. *Characterization of Mobile Matrix Sediment*

For various reasons the interstitial fines fraction is a critical factor for hyporheic biota in the Töss River (BRUNKE and FISCHER, in press.; BRUNKE and GONSER, in prep.). On average it contributed 13.9% (± 6.0 SD) of the bed sediment down to a depth of 70 cm. About 72% (±23 SD) of these fines were smaller than 0.1 mm and the median size was 0.03 mm (Fig. 3a).

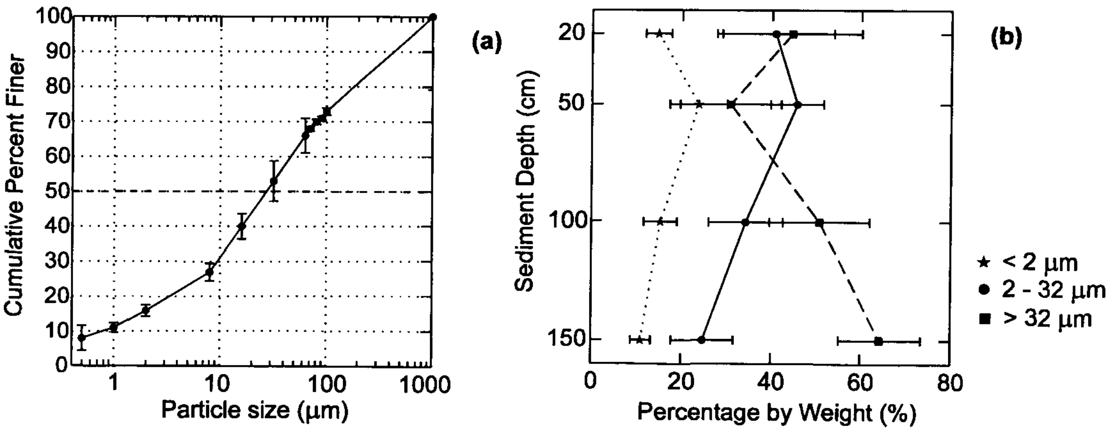

Figure 3. (a) Cumulative size distribution curve for mobile matrix fines. (b) Depth profiles for three size classes which are differentially subjected to physicochemical and mechanical filtering (see text and Tab. 1 for further explanations). Means (±SD). n = 12.

The distribution of particles <2 μm, which is exclusively filtered by physicochemical forces, did not change much with depth except for slightly higher portions at 50 cm depth (Fig. 3b). Its proportions ranged between 8.5% and 32.7% with an average of 14.5 (±6.6 SD). The depth profile of the size fraction 2–32 μm was similar in shape to the depth profile of the fraction <2 μm ($r_s = 0.88$, $p < 0.001$), but it contributed to higher proportions ranging between 16.7% and 60.4% with an average of 36.4% (±12.9 SD). Consequently, the proportions of the fraction > 32 μm were inversely related to the fractions of 2–32 μm and < 2 μm ($r_s = -0.82$, $p < 0.001$ and $r_s = -0.81$, $p < 0.001$, respectively). The proportions of this fraction ranged between 5.1% and 74.9% with an average of 43.7% (±6.4 SD). The hypothesis that the proportions of the large size fraction are greater in upper sediment strata, because of an exclusive filtering in the top layers by mechanical straining is not supported. Instead, this fraction might be responsible for the general trend that fine particles accumulate with depth (LEICHTFRIED, 1988, 1994; PANEK, 1994). In this study the content of the fine particles also increased with depth, though strong temporal variations occurred (Fig. 4). Only in the 50-cm depth stratum of the exfiltration zone was the content of fine particles higher than in deeper strata. This is probably due to the upwelling forces of exfiltrating ground water, which likely prevented the sedimentation of particles into the deeper strata.

As discussed above, bedload movement was more frequent in midstream than at the margins because of the straightening of the channel and bedload deficits. In these midstream sediments the content of total mobile fine particles tended to be lower. In contrast, the vertical patterns of the detrital particles (DPs) (Fig. 5) were opposed to those shown by the content of total fine particles (Fig. 4). Considering the continual degradation of DPs, higher contents of DPs in sediment depths below 50 cm in midstream sediments indicate higher import of fine particles in these sediments. Therefore, it appears that the transport of fines within

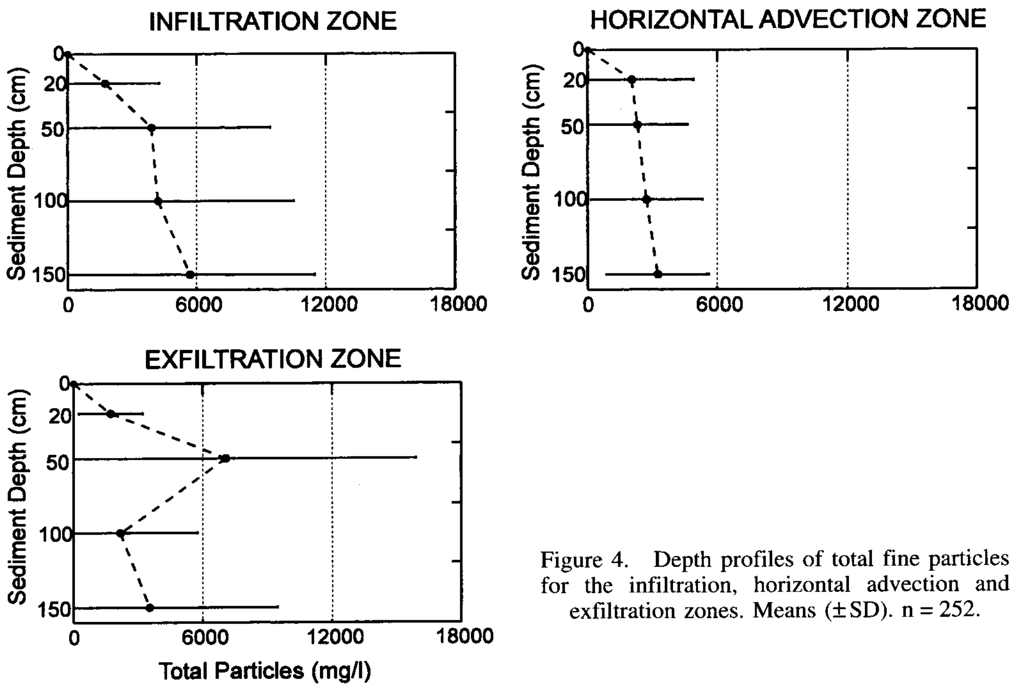

Figure 4. Depth profiles of total fine particles for the infiltration, horizontal advection and exfiltration zones. Means (±SD). n = 252.

midstream sediments is increased relative to the deposition when compared with sediments at the margin. In a similar way FROSTICK *et al.* (1984) found a higher ingress of matrix fines in thalweg sediments in a field experiment. Higher interstitial currents promote the transport of fines, provided there is no straining effect (HERZIG *et al.*, 1970). In contrast, in marginal areas where flow is reduced accumulation may prevail until the next flood that breaks up the sediment across the entire channel. In this context, the contrasting low contents of DPs in the exfiltration zone at a depth of 50-cm compared with the high contents of total particles are remarkable (Figs 4, 5). This relationship supports the interpretation that the import of particles to this stratum is reduced, but the temporal accumulation is of long duration.

The proportion of DPs > 300 μm to DPs > 90 μm ranged between 22% and 60% and showed no clear trend with depth (Fig. 5). This suggests that the straining of DPs >300 μm was not affected differentially by pore size than that of DPs >90 μm. The DPs were certainly flushed (and not deposited during bedload movements) into the sediment down to depths of 100-cm and 150-cm and probably also into the upper strata, because bedload movement of a magnitude which could have buried the DPs to depths below 100-cm would have removed the piezometers.

Calculations using the formula by SOWERS and SOWERS (1970) demonstrate the potential mobility of most matrix fines in the Töss River streambed. The D15/5 of the subsurface sediment (Tab. 4) is about the size of the D85 of the fine particles ($D85_{fines}$ = 0.28 mm) and much higher than the D15 ($D15_{fines}$ = 0.002 mm).The grain size distribution of the matrix fines reflects the absolute sizes of particles. It is likely that the effective grain size distribution is somewhat coarser due to formation of aggregates. However, even with an estimated median grain size an order of magnitude greater (i.e. about 0.3 mm), most particles could still pass the framework. This becomes even more evident considering the sampling method which in general underrepresents the gravels and cobbles, the exclusion of rocks greater than 50 mm, and the fact that the sand and silt fractions were also included in the grain size analysis of the freeze core samples.

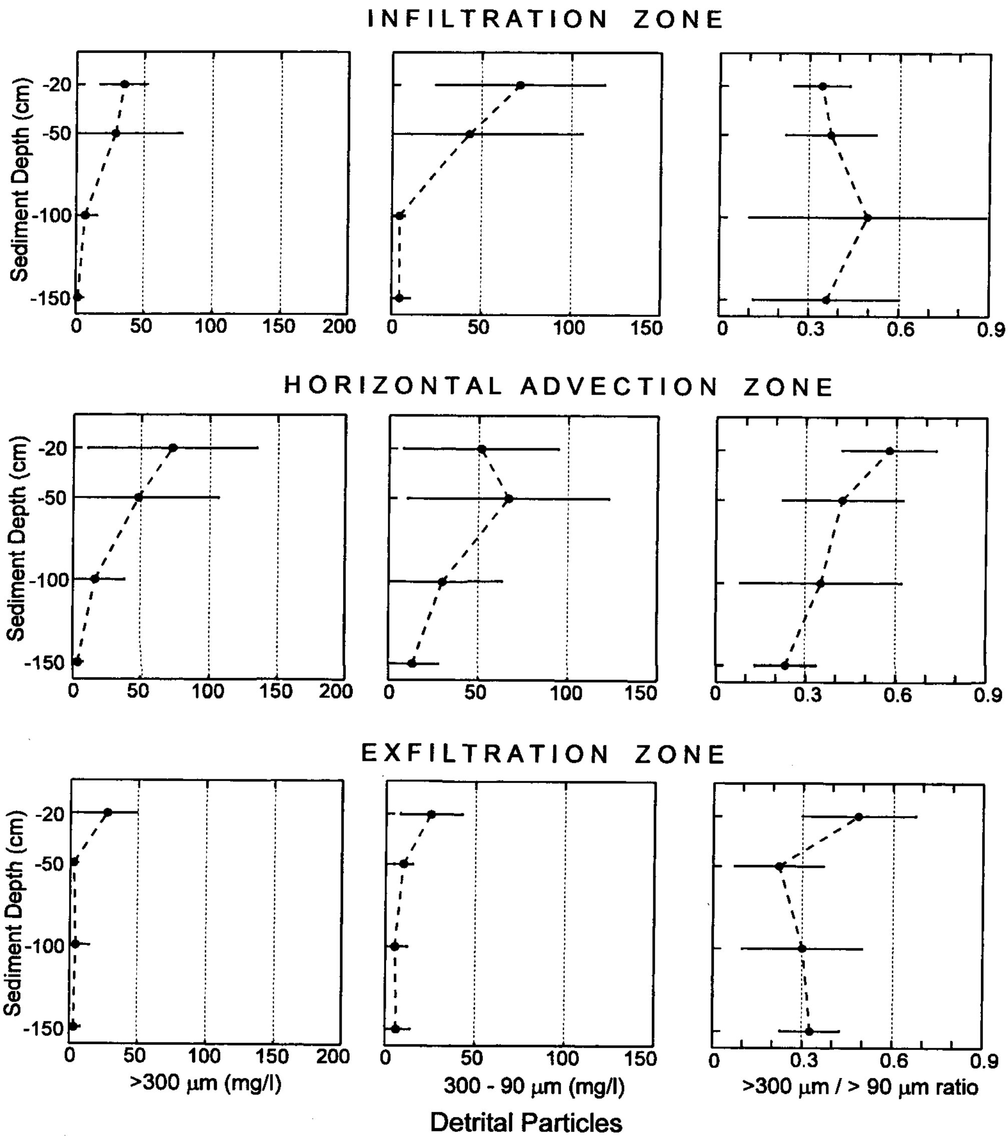

Figure 5. Depth profiles of detrital particles >300 µm, 300–90 µm, and the proportion >300 µm to DPs > 90 µm for the infiltration, horizontal advection and exfiltration zones. Means (±SD). n = 74.

5. Conclusions

Field studies in gravel-bed rivers with unregulated discharge regimes (LEICHTFRIED, 1988; 1991; PANEK, 1994; NAEGELI *et al.*, 1995; PUSCH, 1996; MILAN and PETTS, 1998; this study) do not support the results from flume experiments in which fine particles were retained exclusively in the topmost layer of the channel bed and that further transport into deeper layers was negligible (e.g. BEHNKE, 1969; BESCHTA and JACKSON, 1979; DIPLAS and PARKER, 1992; SCHÄLCHLI, 1992). The presence of detrital particles >300 μm even at sediment depths of 150 cm demonstrated the importance of transport by interstitial throughflow and the interconnectedness of voids. In the Töss River the large pore sizes of the framework sediment probably limited the significance of mechanical straining of mobile fine particles. Rather, the interplay between direction and intensity of interstitial throughflow with sedimentation in subsurface dead zones and particle inertia, as described by HERZIG *et al.* (1970), appear to be more important for depth filtration. Finally, the intrusion of fines appears to be influenced by the location in the channel relative to the frequency of bedload movement, since it controls the composition and shape of framework gravel.

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Detecting the structural and functional impacts of fine sediment on stream invertebrates

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ABSTRACT

Fine sediment is one of the major sources of stream physical and ecological impairment worldwide. We assessed the ecological effects of fine sediment in an otherwise undisturbed catchment (the Isábena, NE Spain). Using data from sites across the catchment we describe the spatial variability and nestedness of invertebrate assemblages and evaluate the effectiveness of compositional (taxon-based) and functional (trait-based) metrics for detecting sediment impacts on these assemblages.

Invertebrate assemblages were relatively taxon poor and had low densities in those locations with high fine sediment content. Assemblages showed significantly nested patterns, with those in sediment-rich locations consisting of a subset of those in locations with little fine sediment. A number of biological traits appeared to promote this nestedness, particularly those conferring resistance and resilience to fine sediment (polivoltinism, short live cycles and small body sizes).

Generalised Additive Models indicated that most metrics were able to detect ecological responses to sedimentation. Some taxon-based metrics (%EPT and evenness) performed less well, with values showing only a weak relationship with fine sediment. Results are consistent with previous studies which have highlighted the limitations of taxon-based metrics and suggest that indices of functional diversity are capable of detecting sediment related impairment.

Overall, the study suggests that fine sediment in the Isábena was selecting for specific life-history traits, and that this selection resulted in clear differences in assemblage structure across the catchment. The use of biological traits in studies of sediment related disturbance may help identify extinction-prone species (e.g. those with univoltine and/or long life-cycles), while trait-based monitoring and assessment metrics, because they reflect the ecological mechanisms underlying observed patterns, should prove useful to help guide management in catchments subjected to excessive fine sediment. More broadly, the study indicates that nestedness in assemblage structure can be driven by local habitat changes, and not only by large scale biogeographical processes.

1. Introduction

Fine sediment (material <2 mm) has long been recognised as one of the major causes of impairment and ecological degradation affecting freshwaters globally (Hynes, 1970; USEPA, 2000; Harrison et al., 2007). While sediment plays an important functional role in river ecosystems, providing a substrate for biological and chemical processes, excess quantities of fine material can cause a range of impacts (reviews by Graham, 1990; Peckarsky, 1984; Waters, 1995; Doeg and Koehn, 1994; Wood and Armitage, 1997; Bilotta and Brazier, 2008). Fine sediment accumulation smothers riverbed micro-topography and clogs interstitial space. It also affects bed stability, and hence disturbance regimes, as a function of changes in cohesion and entrainment thresholds (Hjulström, 1935; Grabowski et al., 2011). Such changes affect aquatic organisms across all trophic levels, through mechanisms that include: (1) modification of habitat availability and suitability for some taxa; (2) increases in turbidity and reduction of primary production, (3) impairment of feeding due to a reduction in the energetic value

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of periphyton and prey density; and (4) impairment of respiration due to low oxygen concentrations in sediment deposits.

Diversity indices or metrics have classically been implemented in ecology to identify and assess ecological impacts and evaluate the ecological integrity of aquatic ecosystems (Coleman et al., 1997; Pires et al., 2000). The most commonly used metric to detect changes and summarise assemblage composition has been taxonomic richness or the number of taxa found in a sample (α-diversity, Magurran, 1988). Other metrics also take into account the assemblage structure and incorporate a measure of the relative abundance of taxa, such as the Shannon index (Shannon and Weaver, 1949) and its associated evenness. While some studies have found these taxonomically based metrics to be useful for detecting anthropogenic disturbance (Robertson, 1981; Erman and Mahoney, 1983), others have reported a poor performance (Wood, 1977; Mouillot et al., 2006). A specific limitation is that they cannot necessarily differentiate the cause of the assemblage change or discriminate between natural and anthropogenic stressors (Reizopoulou et al., 1996; Mouillot et al., 2006; Gallardo et al., 2011).

To overcome the limitations of taxonomic-based methods, the incorporation of species biological traits into alternative metrics has increased in recent years. Biological traits refer to the functional attributes of the species (i.e. morphological, physiological, behavioural and ecological characteristics) and offer the main advantage that they can be applied broadly across biogeographic boundaries (Mcgill et al., 2006). Trait approaches are based on the habitat template model proposed by Southwood (1977), which states that habitat selects for characteristic life history traits through natural selection; species are expected to respond to environmental gradients and we should therefore find a correlation between species traits and habitat characteristics over an evolutionary time period (Poff et al., 2006). Trait patterns can be indicators of the source of impairment because anthropogenic disturbances will select for well adapted species and, as a result, only those possessing relevant adaptive traits are likely to remain (Statzner et al., 2004).

Trait-based approaches have proved useful in understanding environmental change and have been widely applied in biological monitoring and restoration (Gayraud et al., 2003; Statzner et al., 2007). Pollard and Yuan (2010) studied the consistency of the response of trait- and taxon-based measures and found a better performance of the former, regardless of the geographic location and spatial scale considered. Bêche and Resh (2007) also reported consistent patterns in trait diversity and richness over time, despite high taxonomic turnover.

In parallel with the use of trait-based metrics, the assessment of nestedness in species composition has developed to provide insights into the effects of environmental stressors on patterns of species loss. The term *nestedness* was first described for insular faunas to describe patterns of species composition within continental biotas and among isolated habitats such as islands and landscape fragments (Darlington, 1957; Patterson, 1987; Atmar and Patterson, 1993). Perfect nesting occurs where rare species are exclusive to species-rich locations, while reduced nesting occurs where rare species are distributed more evenly (Wright et al., 1998). Species distribution patterns in naturally fragmented habitats often exhibit nested patterns as a result of colonisation and extinction processes over long time scales, while they can also be a consequence of human disturbance and habitat alteration (Fernandez-Juricic, 2002).

Few studies have compared the sensitivity of taxonomic, trait-based and nestedness metrics to fine sediment impacts (Angradi, 1999; Relyea et al., 2000; Zweig and Rabeni, 2001). The limited published work has beed contradictory. For example, Angradi (1999) found a weak correlation between taxonomic metrics and deposited sediment, while Zweig and Rabeni (2001) reported a strong relationship between them. Studies have generally been based either on experimental sediment additions (Suren and Jowett, 2001; Vasconcelos and Melo, 2008; Larsen et al., 2011; Reid et al., 2011) or were carried out in catchments with a high human impact (Leana et al., 2001; Suren et al., 2005; Kreutzweiser et al., 2005), where sedimentation is often associated with multiple stressors (e.g. nutrients, pesticides, alteration of thermal regimes), which result in complex ecological responses and a difficulty in disentangling individual effects.

The present study aimed to assess evidence of patterns in the structure of invertebrate assemblages and functional traits related to fine sediment, and evaluate the extent to which a number of metrics were able to detect these patterns. It is based on data collected in the river Isábena (Central Pyrenees, NE Spain), where naturally high sediment transport rates occur as a result of the presence of highly erodible badland formations in some of its tributaries. Specific objectives of the paper are to (i) describe the invertebrate assemblages across the entire Isábena catchment and assess whether nested assemblages were associated with high sediment loads, (ii) examine the trait composition of assemblages to test whether sites impacted by high sediment loads show a distinct trait structure, and (iii) assess the ability of a set of taxonomic and trait-based metrics to detect the impacts of fine sediment on invertebrate assemblages.

2. Methods

2.1. Study area

The river Isábena is a mesoscale mountainous catchment located in the Central Pyrenees (NE Iberian Peninsula). It belongs to the Ebro basin and occupies and area of 445 km^2 (Fig. 1). The basin is characterised by a strong heterogeneity in relief, vegetation and soil characteristics, with elevations ranging from 450 m to 2720 m.a.s.l. in the northern parts. Woodland occupies the major part of the catchment, with agriculture (on floodplains) and urban areas comprising only 13% and 1% of its area, respectively. An assessment of the physical and chemical habitat conditions in the Isábena was undertaken by the Ebro Water Authorities (United Research Services – URS, 2002). Overall, the river was classified as largely unaffected by human activities, with more than 71% of the reaches they studied classified as having physical and chemical conditions that were "good" or "very good" and none classified as "bad" or "deficient".

The hydrology and sediment dynamics of the Isábena have been described in numerous studies (e.g. Verdú, 2003; Francke et al., 2008; López-Tarazón et al., 2009, 2010). Hydrology is characterised by a rain-snow fed regime with flows showing a marked seasonality, although the river has never dried up within the period of record. Floods typically occur in spring as a result of rainstorms, which tend to be scarce in summer and variable in autumn and winter. Mean annual discharge at the basin outlet is 4.1 m^3/s, with minimum flows ($\sim$0.20 m^3/s) typically occurring in summer and maximum flows recorded during autumn (e.g. 290 m^3/s on 9th November of 1966, 192 m^3/s on 18th December 1997).

Miocene continental sediments (marls, sandstones and carbonates) make up the western and middle parts of the catchment, leading to the formation of badlands. Although they represent less than 1% of the total area of the basin, badland areas, being highly erodible, have proven to be the major source of sediment, contributing more than 95% of the annual sediment load of the catchment. In addition, these areas are very well connected to the drainage network, facilitating sediment delivery to the channel. As a consequence, suspended sediment concentrations (hereafter SSCs) show great variability (they can span five orders of magnitude

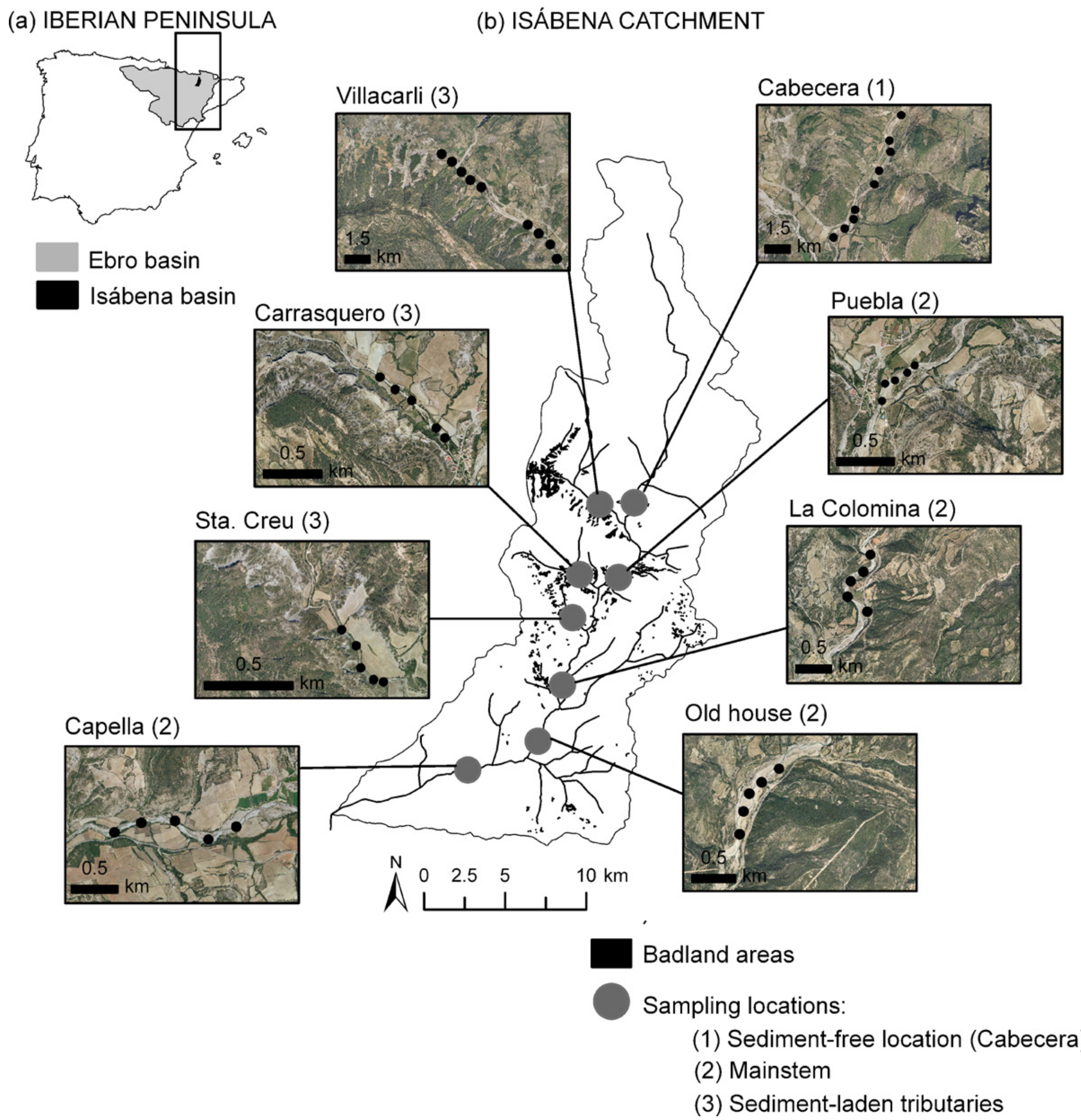

Fig. 1. Location of the Isábena catchment in the Iberian Peninsula (a) and sampling sites (b). Badlands are shown as black areas in the map.

for the same discharge) and may reach maximum instantaneous values above 300 g/l (López-Tarazón et al., 2009).

The relative location of badland areas gives rise to a wide range of sedimentary conditions along the river channel. In some headwater areas where badlands are virtually absent, fine sediment accumulation in the channel is minimum, whereas in areas with extensive badlands, it is extremely high. These contrasts, together with the fact that the river is not hydrologically regulated and shows limited human impact on landcover, make this catchment a suitable location for assessing the effects of fine sediment.

2.2. Sampling design

2.2.1. Sampling locations

In order to account for the spatial variability in both macroinvertebrate assemblage composition and fine sediment accumulation, eight reaches were selected across the Isábena (Fig. 1). One reach was located in the headwaters of the catchment (Cabecera) where badlands are virtually absent and thus represents an area free from fine sediment. Three reaches were located in the sediment-laden tributaries in the western and northern parts of the catchment (Villacarli, Carrasquero and Sta. Creu). The other four reaches (Puebla, La Colomina, Old house and Capella) were located along the mainstem, to detect any downstream change in invertebrate assemblages associated with increasing distances from the sediment sources. To minimise inter-habitat variability, all reaches were pool-riffle morphologies, with all invertebrate samples being collected from riffles. In most reaches 5 samples were collected, with a sample taken from a randomly allocated point within each riffle. In two reaches (Cabecera and Villacarli), 10 samples were collected. This increased number was due to the fact that these two tributaries occupy a larger area than the others, provide the majority of the runoff and sediment to the catchment (Verdú, 2003) and, in the case of Cabecera, provides a reference free from fine sediment influences. Upstream catchment characteristics for each reach, as well as summary hydrologic information and suspended sediment concentrations, are given in Table 1. Sampling was completed under stable, base-flow conditions within 1 week (August 2010); which minimised sedimentary and geomorphic differences related to flow change.

Table 1
Summary information on study reaches. Landcover statistics were determined using GIS and are for the total area upstream from each reach. %Bad (percentage of badlands in the catchment), %Wood (percentage of catchment area occupied by badlands). %Agr (percentage of agriculture) and %NP (percentage of non-productive land, includes all those areas which are not under agriculture, e.g. urban areas).

Reach	Area (km^2)	%Bad	%Wood	%Agr	%NP	SSC[a] (g/l)	Vel ± SD (m^3/s)[b]	Depth (m)[b]	Q (m^3/s)[c]
Cabecera	146	<0.01	97.7	0.8	1.4	0.005	0.65 ± 0.02	0.15	4.1
Villacarli	40	6.7	81.2	4.3	14.5	0.007	0.61 ± 0.11	0.17	0.7
Sta. Creu	10	4	83.8	12	4.25	0.039	0.03 ± 0.01	0.11	0.2
Carasquero	25	2	86	10.5	3.5	0.009	0.47 ± 0.08	0.15	0.4
Puebla	215	1.3	93.8	2.8	3.4	0.006	0.56 ± 0.03	0.13	3.2
La Colomina	270	1.5	92.7	4.4	2.8	0.003	0.82 ± 0.04	0.15	5.9
Old house	309	1.5	91.9	5.4	2.6	0.025	0.68 ± 0.04	0.17	5.5
Capella	395	1.2	87.5	10.3	2.1	0.011	0.58 ± 0.02	0.14	6.2

[a] Values of SSC are for the day of sampling.
[b] Depth and velocity data are based on measurements taken at each of the invertebrate sampling locations ($n = 10$ in Villacarli and Cabecera, $n = 5$ at the remainder).
[c] Discharge for each reach was calculated empirically using depth and velocity data collected on the day of invertebrate sampling.

2.2.2. Invertebrate sampling

The combined total of 50 benthic samples was collected using a standard surber sampler (300 μm mesh size, 0.09 m^2 area). Invertebrates were preserved in 4% formaldehyde for later analysis. In the laboratory, they were sorted and identified to genus level according to Tachet et al. (2002). Diptera and Oligochaeta were identified as far as possible according to the key (subfamily or tribe level).

2.2.3. Fine sediment determination

Deposited fine sediment was assessed in the patches of bed immediately adjacent (laterally) to where invertebrates had been collected, using the re-suspension technique first proposed by Lambert and Walling (1988). This technique had been previously used in the Isábena to estimate fine sediment storage in the main-stem channel (López-Tarazón et al., 2011a,b) and in a preliminary study of fine sediment effects on benthic invertebrates (Buendia et al., 2011). Full details of this methodology can be found in these papers. Briefly, a metal cylinder (0.5 m diameter, and 0.6 m height) was carefully pushed into the river to delimit the sampling area. The surface 10 cm of the river bed were vigorously disturbed using a rod to resuspend deposited fine sediment. Two 500 mL water samples collected just after the disturbance to provide a mean of the suspended sediment concentration in the cylinder and calculate the volume of fines contained in the gravel-bed matrix isolated by the cylinder.

2.3. Data analysis

2.3.1. Spatial patterns in invertebrate assemblages

Non-metric Multidimensional Scaling (NMDS, Kruskal, 1964) using Sorensen's coefficient (Heino et al., 2003) was used to examine patterns in invertebrate assemblage composition. This ordination method calculates a distance matrix (D) and visualises this matrix in a low dimensional configuration (2 or 3 dimensions). It has advantages over other ordination techniques in that it makes few assumptions about the nature of the data and allows the use of any distance measure (McCune and Mefford, 1995). Stress was used to judge the goodness of fit (i.e. how well the ordination summarises the observed distances among the samples), as it indicates the degree of correspondence between the distances among points in the ordination plot and the original distances measured. Sorensen's distance values were used for the NMDS. Scree plots (i.e. stress by dimension) were used to select the dimensionality of the analysis. A vector representing fine sediment was fitted to the final plot to help interpretation of the ordination. Squared correlation coefficients (r^2) for the vector and the significance of these values were assessed by 1000 permutations of the sediment values.

To test the significance of patterns of clustering found in the NMDS, a Multi-Response Permutation Procedure (MRPP) was run using the Sorensen's distance values. MRPP is a nonparametric procedure for testing the hypothesis of difference between two or more pre-defined groups (Zimmerman et al., 1985). Chance-corrected within group agreement (A) was used to describe within group heterogeneity. Values of A range from 0 to 1, showing lower values when the groups are formed by identical items. The significance of the null hypothesis of no difference among groups was assessed using a Monte-Carlo permutation procedure ($n = 1000$).

The nestedness temperature of presence-absence matrices was used to quantify the level of nestedness in invertebrate assemblages across the Isábena catchment. The temperature method was proposed by Atmar and Patterson (1993) and is based on a measure of the order and disorder apparent in the nested patterns. It uses Euclidean distances of unexpected species absences and presences in individual locations from the isocline (or extinction curve) which separates presences from absences in a perfectly nested matrix. The isocline is a curvature of maximum packing, given the species-by-site matrix size and fills, and does not depend on the structure of the data (Atmar and Patterson, 1993). In a perfectly nested matrix, all presences will be in the upper left corner of the theoretical isocline and the matrix temperature will be 0° (cold systems where extinction order does not change). The maximum value of this metric is 100° and corresponds to systems with a high influence of random processes acting on populations, which therefore are less predictable. These "hot" systems are represented by a matrix with many unexpected absences above the isocline and unexpected presences below the line.

Nestedness analysis used the BINMATNEST algorithm developed by Rodriguez-Girones and Santamaria (2006). This algorithm has been argued to be more robust and efficient than others (Azeria and Kolasa, 2008; Larsen and Ormerod, 2010). BINMATNEST reorders rows and columns from the presence-absence matrix to minimise unexpectancy and maximise nestedness and calculates the matrix temperature (T). In order to test the statistical significance of the maximised nested matrix (i.e. that the observed T is not lower than expected by chance), the observed T was compared with that of null models generated by randomising the original matrix in 400 Monte-Carlo simulations. Following Larsen and Ormerod (2010) the null model III was used, as it is less sensitive to species richness and occurrences and, thus, is considered the most conservative and reliable. Moreover it has been shown to perform better (smaller type I error) than the other null models available (Rodriguez-Girones and Santamaria, 2006). This model calculates the proportion of presences for each row (ρ_{ri}) and column (ρ_{ci}), and, for every random matrix, the probability of a cell being filled is set equal to the average probabilities of occupancy of its row and column $(\rho_{ri} + \rho_{ci})/2$ (Bascompte et al., 2003).

Spearman-rank correlations were used to test whether nested patterns observed could be driven by the fine sediment content of the riverbed and by the biological traits of the assemblages. For this, the ranking of sites in the maximally packed matrix was related to the proportions of each trait category (as per Larsen and Ormerod, 2010). A significant correlation would suggest that assemblages are packed in a particular order due to fine sediment influences and the selection of certain traits in sediment-rich (most nested) locations.

2.3.2. Biological trait data

Information on 12 traits (Table 2) was collected for a total of 47 taxa, using databases for European rivers (Tachet et al., 2002; Statzner et al., 1994, 2007). For some taxa (i.e. Oligochaeta), information on biological traits was not available and these were excluded from the analysis.

A fuzzy coding procedure (Chevenet et al., 1994) was used to quantify the affinity of each taxon for each trait category. Scores ranged from 0 (indicating no affinity) to 3 or from 0 to 5, depending on the number of categories within each trait. This approach provides information on the possible differences among functional characteristics of species belonging to the same genus and the possible variation of some habits throughout the different life stages of a species (Chevenet et al., 1994; Usseglio-Polaterra, 1991). To give the same weight to each taxon and each trait, affinity scores were rescaled so that the sum of a given taxon and a given trait equalled one (Doledec and Statzner, 2008; Van der Linden et al., 2012). Trait categories were then multiplied by the $\log(x+1)$ abundance of each taxon at each site to obtain values used to create a site × trait abundance matrix (Larsen and Ormerod, 2010; Feio and Doledec, 2012). Principal Component Analysis (PCA) on the covariance matrix of the fuzzy coded data was performed to determine patterns of functional trait composition of the invertebrate assemblages.

All statistical analyses were performed using the following packages within the R environment (R Development Core Team, 2009): *vegan v2.0-3* (Oksanen et al., 2012) for MMS, MRPP and PCA and *bipartite* (Dormann et al., 2008) for the nestedness analysis.

2.3.3. Metrics

Invertebrate density (number of individuals per unit area) and six taxon-based diversity metrics were calculated for each invertebrate sample. The diversity metrics included taxon richness (total number of taxa), diversity (Shannon's index; Shannon and Weaver, 1949) and evenness (calculated as the ratio of observed diversity to maximum diversity; Pielou, 1966). These metrics were selected because they have a long history of use in aquatic ecology, with each summarising key properties of the assemblage (Magurran, 1988). In addition, three EPT metrics were computed to assess their ability to detect fine sediment impacts: EPT density (combined abundance of Ephemeroptera, Plecoptera and Tricoptera per unit area), EPT richness (number of EPT taxa) and %EPT (proportion of the assemblage consisting of EPT taxa). These metrics have been widely used in detecting human impacts, as EPT taxa are considered to be relatively sensitive to disturbance (Angradi, 1999; Zweig and Rabeni, 2001; Kaller and Hartman, 2004; Larsen et al., 2009, 2011).

Two trait-based metrics were computed for each sample to assess the functional attributes of invertebrate assemblages. Trait richness was calculated as the number of trait categories present. Functional diversity (FD) was calculated as the Rao diversity coefficient, following the methodology developed by Champeli and Chessel (2002). This index takes into account the dissimilarity in the trait space among species and each species' abundance in the quantification of functional diversity (i.e. by weighting the pairwise species dissimilarity by the product of relative abundances of the two species):

$$\mathrm{FD} = \Sigma_{i=1}^{S}\Sigma_{j=1}^{S} d_{ij} p_i p_j$$

Table 2
Invertebrate traits and categories considered in this study. Codes were used as labels in Fig. 5.

Trait	Categories	Code
	<0.25	A.1
	0.25–0.5	A.2
	0.5–1	A.3
(A) Maximal size	1–2	A.4
	2–4	A.5
	4–8	A.6
	>8	A.7
	<1 year	*B.1*
(B) Life cycle duration	>1 year	*B.2*
	<1	C.1
	1	C.2
(C) Potential generations per year	>1	C.3
	Aquatic passive	D.1
	Aquatic active	D.2
(D) Form of dissemination	Aerial passive	D.3
	Aerial active	D.4
	Egg	E.1
	Larva	E.2
	Nymph	E.3
(E) Aquatic stage	Adult	E.4
	Ovoviviparity	F.1
	Isolated eggs, free	F.2
	Isolated eggs, cemented	F.3
	Clutches, fixed	F.4
	Clutches, free	F.5
	Clutches, in vegetation	F.6
(F) Reproduction	Clutches, terrestrial	F.7
	Asexual reproduction	F.8
	Eggs, statoblasts	G.1
	Coccons	G.2
	Housing	G.3
	Diapause	G.4
(G) Resistance forms	No resistance forms	G.5
	Microorganisms	H.1
	Detritus <1 mm	H.2
	Plant detritus >1 mm	H.3
	Living macrophytes	H.4
	Living microphytes	H.5
	Dead animal <1 mm	H.6
	Living microinv.	H.7
(H) Food	Living macroinv.	H.8
	Vertebrates	H.9
	Absorber	I.1
	Deposit feeder	I.2
	Shredder	I.3
	Scraper	I.4
	Filter feeder	I.5
	Piercer	I.6
(I) Feeding habits	Predator	I.7
	Prasite	I.8
	Tegument	J.1
	Gill	J.2
	Plastron	J.3
	Spiracle (aerial)	J.4
(J) Respiration	Hydrostatic vesicle (aerial)	J.5
	Flier	K.1
	Surface swimmer	K.2
	Full water swimmer	K.3
	Crawler	K.4
	Burrower	K.5
	Interstitial	K.6
(K) Locomotion	Temporarily attached	K.7
	Permanently attached	K.8
	Flag/boulders/cobbles/pebbles	L.1
	Gravel	L.2
	Sand	L.3

Table 2 (*Continued*)

Trait	Categories	Code
(L) Substrate	Silt	L.4
	Macrophytes	L.5
	Microphytes	L.6
	Branches rots	L.7
	Organic detritus littler	L.8

with ρ_i being the proportion of the ith species, d_{ij} the dissimilarity of species i and j, and S the total number of species or taxa in the assemblage. Rao's diversity coefficient for each sample was determined using the package FD, which uses the species-by-species Euclidean distance matrix for the computation of the index (Laliberté and Legendre, 2010).

Following the approach used by Gallardo et al. (2011), the relations between metric values and fine sediment were assessed using Generalised Additive Models (GAMs). This method uses smoothing curves to model the relationship between response and explanatory variables and is suitable for modelling non-linear relationships. The model for all response variables (i.e. metrics) had the general form:

$$Y_i = \alpha + f(\text{sediment}_i) + \varepsilon_i$$

where Y_i is the value of the metric in sample i, sediment$_i$ is the value of deposited fine sediment in sample i, ε_I is the residual for sample I and α the population intercept. The function $f()$ is the population smoothing function, with cubic regression splines used in all models to estimate the smoothed relationship between response variable and predictors. For count data (EPT richness and taxon richness) a Poisson distribution was selected to model the residual variation, while a binomial distribution was used for proportion data (%EPT) and a Gaussian distribution for continuous variables (invertebrate density, EPT density, Shannon, evenness and functional diversity). The percentage of deviance explained by the fitted model was used to assess its goodness-of-fit; inspection of residual plots confirmed modelling assumptions. Analyses were performed using the package mgcv (Wood, 2011) within the R environment (*R Development Core Team 2009*), which allows the application of Generalised Cross Validation (GCV) to automatically select the degrees of freedom of the smoothers (i.e. amount of smoothing).

3. Results

3.1. Fine sediment

Mean values of fine sediment stored in the channel of the Isábena ranged from 90 g/m^2 at those sites without the influence of badlands to 400 g/m^2 in the mainstem and up to 1000 g/m^2 in those tributaries where badlands comprised a relatively large proportion of their catchment area (Fig. 2). Maximum individual values were found in Villacarli, where fine sediment reached 1800 g/m^2. ANOVA and Tukey's HSD indicated that fine sediment values differed significantly between the three groups of sites shown in Fig. 2 ($p < 0.001$), but that there were no differences between sites within each group (mainstem p-value = 0.09; tributaries p-value = 0.1).

3.2. Assemblage structure and trait composition

NMDS produced a 2-axis ordination (Fig. 3) with a final stress value of 0.18, which indicates a satisfactory solution (stress values >2 indicate unreliable and poor solutions; Clarke, 1993). The direction of the vector for fine sediment shows the sediment gradient (increasing from left to right), with the length of the arrow being proportional to the correlation between the variable and the ordination. Fine sediment proved to be highly correlated with the

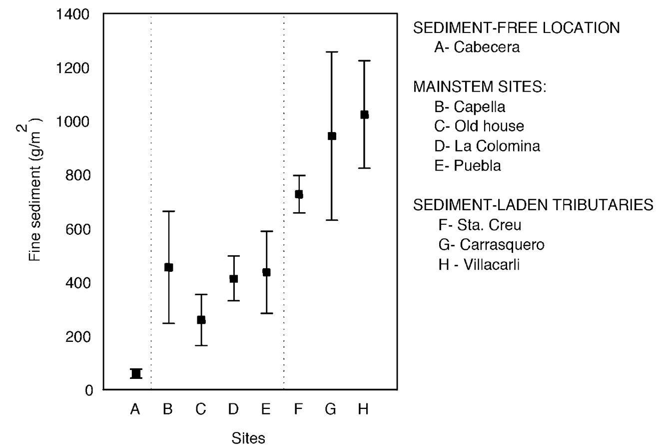

Fig. 2. Mean values of fine sediment content on the river bed. Bars represent the 95% confidence intervals.

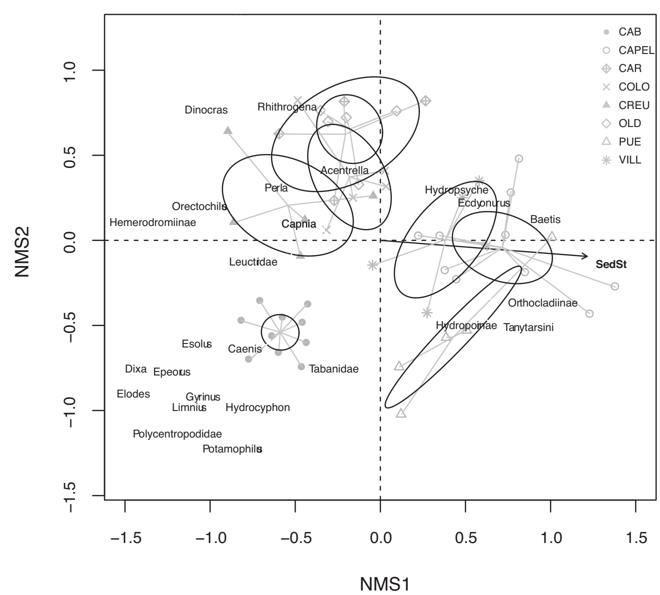

Fig. 3. NMDS ordination plot for the invertebrate data. Lines were drawn connecting each sample and the centroid of the group they belong to. 95% confidence ellipses around class centroids were also drawn. The overlapping of these ellipses indicate that classes are not significantly different at level $p < 0.05$. For clarity of the representation, those species with low NMDS1 scores (<0.3) were removed from the plot.

first axis of the ordination ($r^2 = 0.45$, p-value <0.001). Three major groups are evident, each formed by a cluster of ellipses. Chance-corrected within group agreement (A) from the MRPP was 0.2, indicating a lower heterogeneity within the three groups identified than expected by chance (McCune and Grace, 2002). The groups were clearly spread along axis 1. Cabecera (sediment-free site, located in the right hand side of the plot) was separated from the rest, indicating marked differences between its invertebrate assemblages and those at locations downstream. Samples from the sediment-laden tributaries (Carrasquero, Villacarli and Sta. Creu) formed another group located at the positive end of the ordination axis. This group sat very close to the group formed by the samples from the mainstem, indicating some similarities between their faunas. No clear spatial pattern was evident within the group formed by sites along the mainstem, suggesting no major longitudinal changes in assemblage composition.

Ordination of taxa highlighted invertebrates typical of sites with high sedimentation, as well as those most sensitive to fines. EPT

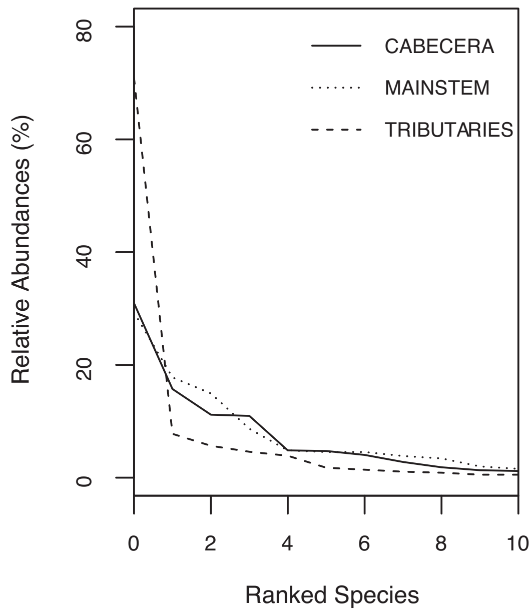

Fig. 4. Ranked-abundance plots for the three main groups defined from the NMDS ordination. Change in species relative abundances is more gradual in Cabecera and the mainstem, with a more evenly distribution of individuals between the taxa than in the tributaries.

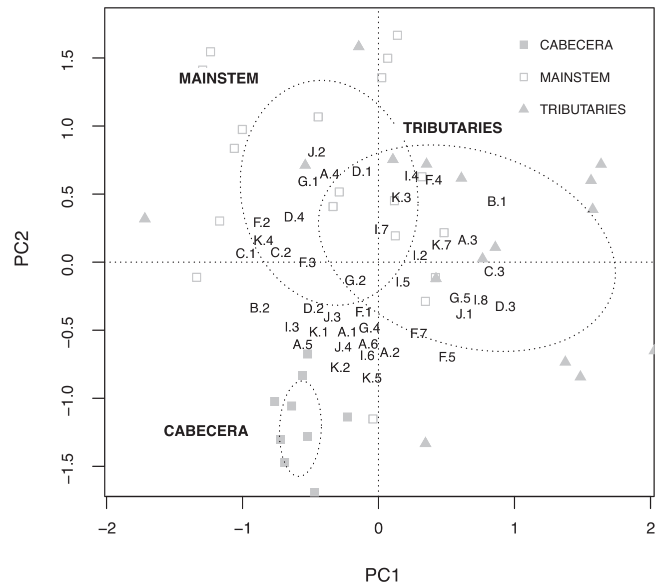

Fig. 5. PCA ordination showing the position of sites described in terms of their trait (i.e. functional) composition. 95% confidence ellipses around centroids were plotted around each group. Codes for the trait data are described in Table 2. The first two axes explained 47% of the variance in the trait data.

taxa showed a variety of responses to the sediment gradient. Some mayfly and stonefly genera such as *Baetis*, *Ecdyonurus* and the caddis *Hydropsyche* were aligned with the group formed by the tributaries, indicating their predominance in these locations. Other EPT genera were inversely correlated to fine sediment values, such as the mayfly *Epeorus*, the plecopterans *Dinocras* and to a lesser extent *Perla*, *Capnia* and *Leuctridae*. In general, coleoptera genera such as *Gyrinus*, *Dryops* and *Elodes*, as well as the dragonfly *Onychogomphus*, proved to be sensitive to fines as they were consistently weighted towards the negative end of NMDS axis 1.

Density and taxon richness differed significantly between the three groups of sites [ANOVA: $F(8, 47) = 27.86$ and 45.40 respectively; $p < 0.001$]. Cabecera showed the highest mean values (2700 ind/m^2 and 42 taxa), while the lowest means were found in the tributaries, particularly in Sta. Creu river (100 ind/m^2 and 4 taxa). Sites on the mainstem showed intermediate values of richness (5–11 taxa) and density (370–500 ind/m^2). In general, assemblages at all sites were dominated by EPT taxa, with ephemeropterans being the most abundant. The most common taxon at all sites was *Baetis*, comprising approximately 30% of the total abundance in Cabecera and the mainstem and 70% in the tributaries. Fig. 4 shows the ranked-abundance diagrams for the groups defined in the NMDS ordination. Cabecera and the mainstem showed a more equitable distribution, with a larger proportion of taxa with intermediate abundances (i.e. *Esolus*, *Hydropsyche*, *Ecdyonurus*). Conversely, one taxon (i.e. *Baetis*) was clearly dominant in the tributaries, with the remainder being uncommon (e.g. *Cheumatopsyche*, *Acentrella*, *Rhyacophila*).

PCA (Fig. 5) identified major patterns in biological traits present among sites. Both axes were correlated with fine sediment ($r^2 = 0.3$, p-value = 0.001), implying that overall trait distribution patterns were influenced by sedimentary conditions. Cabecera and the group formed by the tributaries draining badland areas corresponded to the extremes in the fine sediment gradient and were clearly separated in the ordination space. Samples from sites along the mainstem were widely dispersed on the PCA rather than forming distinct site clusters. The same applied to samples from sites in the tributaries. The proximity of the mainstem and tributary sites (specifically the overlap of their 95% confidence ellipses around group centroids) suggested a similar set of traits in their invertebrate assemblages.

Traits responsible for the major differences between sites were mainly related to the reproductive potential of organisms (i.e. number of generations per year) and the duration of their life cycle. Results indicated that multivoltine taxa (>1 generation per year) with short life-cycle durations (<1 year) were favoured in those locations with high volumes of fine sediment. Forms of locomotion seemed to respond differently to sedimentation. Unlike swimmers and temporarily attached taxa, crawlers and burrowers were particularly sensitive and were absent from the sediment laden tributaries. Assemblages in sediment-free locations were characterised by a more diverse set of maximum size categories, with animals ranging from very small (<0.25 mm) to large (4–8 mm) body sizes, while only small sizes (0.5–1 mm) were frequent in the tributaries. No resistance forms seemed to be selected by fine sediment. To a lesser extent, sedimentation also appeared to favour certain feeding groups (deposit feeders and scrapers), as well as aerial forms of dissemination and tegumental respiration. There were no major patterns related to general substrate affinities or aquatic stages (i.e. categories of these traits showing low component loadings in the PCA analysis).

The temperature of the maximally packed matrix indicated a significantly nested pattern in taxon distribution across the Isábena (Fig. 6, $T = 10.47$; $p < 0.01$). The nested rank order of taxa was significantly correlated with fine sediment content on the river bed ($r_s = 0.35$, $p = 0.008$). Therefore, those found in taxon-poor locations were subsets of those found in the taxon-rich, sediment-free ones. This suggests that the sequential removal of taxa across the catchment was driven by sedimentary conditions, with only common species persisting in fine sediment-rich locations. Results from Spearman's rank correlations between trait categories and the site ranking in the matrix indicated that certain traits appeared to be driving the nested patterns (see Table 3). The least nested locations, which had less fine sediment content, were mainly characterised by long-lived taxa with longer life cycles, larger sizes, a higher representation of shredders and deposit feeders and taxa with abdominal gill placement. Conversely, locations at the nested end of the matrix

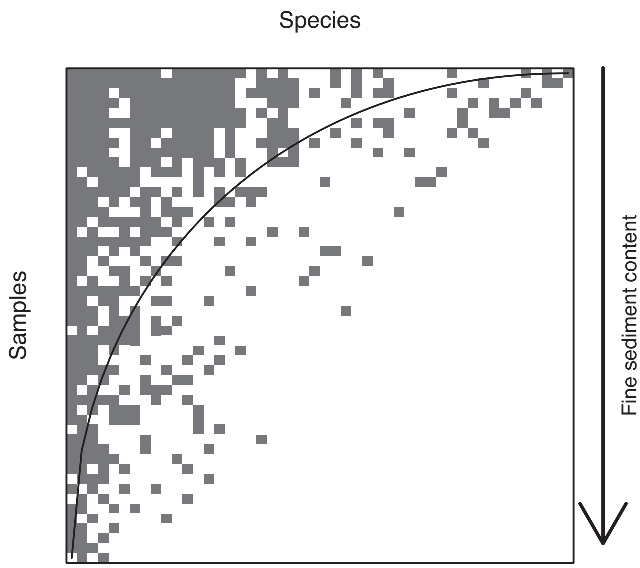

Fig. 6. Maximally packed matrix from the nestedness analysis. Filled squares indicate the presence of a particular species in a specific patch. The isocline reflects the curvature of maximum packing or perfect order.

Table 3
Spearman correlation coefficients (r_s) between trait categories and the ranking of sites from the maximally packed matrix. Note that only those traits with categories showing significant values are shown.

Trait	Category	Spearman, r_s	p-Value
Maximal size	0.25–0.5	−0.65	<0.0001
	0.5–1	0.34	0.014
	1–2	0.41	0.003
	2–4	−0.46	0.0006
	4–8	−0.44	0.001
Life-cycle duration	<1 year	0.69	<0.0001
	>1 year	−0.69	<0.0001
Potential generations per year	<1	−0.51	0.0035
	1	−0.29	0.0001
	>1	0.48	0.0004
Feeding habits	Shredder	−0.53	<0.0001
	Scraper	0.43	<0.0001
	Filter feeder	−0.55	<0.0001
	Deposit feeder	0.23	<0.0001
Respiration	Gills	0.45	0.0008
Locomotion	Swimmer	−0.58	<0.0001
	Crawler	−0.35	0.01
	Burrower	−0.49	<0.0001

held short-lived taxa with smaller body sizes, a larger number of potential generations per year and a higher representation of scrapers, deposit feeders and burrowers.

3.3. Responsiveness of metrics to fine sediment

Fig. 7 shows the values of the metrics for the study sites. All metrics showed lower values in the mainstem and particularly the sediment-laden tributaries, suggesting an inverse response of these metrics to fine sediment content across the study catchment. %EPT was the exception, with a higher EPT to all taxa ratio in locations with larger values of fines. The deviance in metric values explained by the fitted GAMs (goodness-of-fit) varied from 22 to 91% (Table 4). The density metrics (invertebrate density and EPT density) were those best explained by fine sediment content (deviance values above 80%). Conversely, %EPT and evenness had poorer model fits (deviance values of 43% and 22% respectively). Metrics based on biological traits also performed well, particularly functional diversity, with almost 70% of it deviance explained by variation in fine sediment.

Table 4
Deviance explained and p-value (significance of the smoother) for each GAM fitted to metric and fine sediment values.

Metric	Deviance explained (%)	p-Value
Invertebrate density (ind/m^2)	91.7	<0.001
Taxon richness	75.3	<0.001
EPT density (ind/m^2)	81.9	<0.001
EPT richness	71.9	<0.001
%EPT	43.4	<0.001
Evenness	22	0.01
Shannon	57.4	<0.001
Trait richness	53.7	<0.001
Functional diversity	68.8	<0.001

4. Discussion

In-channel fine sediment constitutes an important component of sediment transfer through river systems, and thus plays an important role in the sediment budget of drainage basins (Wilson et al., 2004). López-Tarazón et al. (2011a,b) studied in-channel surface fine sediment storage in the Isábena and reported a mean value of 990 g/m^2 for their study sections, with a maximum of 8400 g/m^2. Larsen et al. (2009) reported values for in-channel fine sediment in the river Usk (where human activities have negligible impact) ranging from 2 to 147 g/m^2. Comparison of these values helps emphasise the extremely high fine sediment content of the riverbed in parts of the Isábena. Nonetheless, some sections of the Isábena are relatively sediment free, due to an absence of badland areas in their catchments. Such contrasts within this one, otherwise undisturbed catchment provided a basis for assessing differences in assemblages related to fine sediment. The consistency of the results of the NMDS and trait-based analyses provide strong evidence of sediment related patterns in invertebrate assemblages across Isábena. Sites with high values of deposited sediment were taxonomically depauperate and functionally homogeneous, with spatial patterns reflecting sample site position relative to the badland areas (i.e. sediment sources). An important point that should be borne in mind when interpreting these results is that our study assessed the overall effects of sedimentation on assemblages. Thus, it is not possible from our data to disentangle effects of the direct physical impacts of fine sediment on invertebrates (e.g. clogging of gills, reduction of interstitial space) from physico-chemical changes (e.g. low oxygen concentrations, changes in pH) which may be occurring within the bed as a consequence of sedimentation (Ryan, 1991).

4.1. Structural and functional effects of fine sediment

Small increases in deposited sediment can cause a decrease in invertebrate densities due to a reduction in habitat availability and quality, even though there may be little or no change in assemblage structure (Rabeni et al., 2005). With greater amounts of deposited sediment, densities of sediment-tolerant species may increase and alterations in assemblage structure may occur. In our study, sediment accumulation was associated with reduced invertebrate density and taxon richness, with the most impoverished sites located in the sediment laden tributaries draining badlands (some sites supported as little as 4 taxa and 100 individuals/m^2). Invertebrate assemblages were generally homogeneous along the mainstem of the river. This homogeneity indicates that there was no clear downstream ecological recovery gradient related to distance from headwater sediment sources. Despite the distance from these

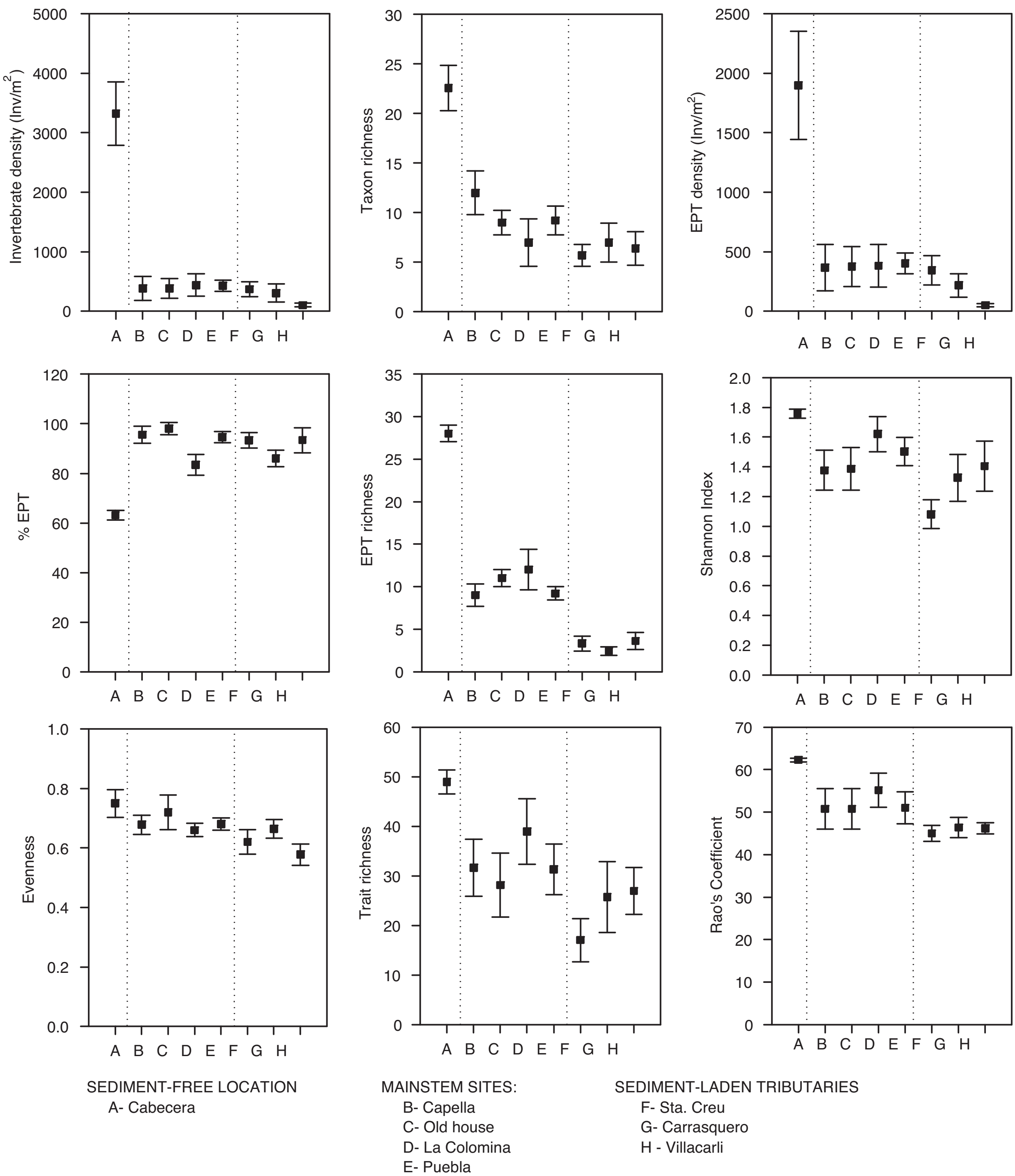

Fig. 7. Mean values and 95% confidence intervals of the metrics computed for each sampling location.

sources, large accumulations of fines were found along the main channel at the most downstream study sites, providing a plausible explanation for observed patterns of assemblage composition. The presence of such large accumulations of fine materials here indicates that fine sediment input exceeds the transport capacity of the river mainstem. This situation is typical of rivers draining highly erodible materials where sediment production is high (Walling and Amos, 1999; López-Tarazón et al., 2010).

Previous studies of the implications of fine sediment inputs to streams indicate that, in the longer term, assemblage structure changes from one comprising a range of Ephemeroptera, Plecoptera and Tricoptera to one dominated by animals such as Oligochaeta, Chironomidae and Bivalva, taxa adapted to burrowing (Wood and Armitage, 1997). However, this shift was not observed in the Isábena, where EPT taxa were dominant and other genera (e.g. *Gyrinus*, *Potamophilus*, *Polycentropodidae*) were common only in those locations with low volumes of fines. The Ephemeroptera seems to be an important invertebrate Order to consider in fine sediment impact studies as it contains taxa responding in markedly different ways to sedimentation. For example, *Acentrella*, *Epeorus* and *Rhithrogena* have been reported to be fine sediment intolerant or moderately intolerant while *Baetis*, *Ephemerella* and *Paraleptophlebia* appeared to be tolerant taxa (Relyea et al., 2000; Wallace and Gurtz, 1986). In the Isábena, assemblages in sediment impacted sites were clearly dominated by *Baetis* (which made up more than 50% of the assemblages) and *Hydropsyche*.

As predicted by the habitat template model (Southwood, 1977), changes in habitat in the Isábena appeared to select for certain biological and ecological traits. Changes in trait profiles reflect changes in the ability of invertebrates to cope with disturbances, as only those traits conferring resistance and resilience are selected (Statzner and Bêche, 2010). Results from the Isábena are in accordance with other studies which have indicated that modified habitats may show shifts in trait structure as those taxa with sensitive traits are filtered out (Peru and Dolédec, 2010; Feio and Doledec, 2012). Across the Isábena, the representation of short life cycles, small sizes, deposit feeders and tegumental respiration increased as fine sediment storage increased. Life history was clearly the trait most affected by sedimentation, with polivoltinism being favoured in sediment-rich locations. Fine sediment, once deposited on the river bed, can easily be re-suspended even during relatively low flow conditions (López-Tarazón et al., 2009). The selection of polivoltinism and short life cycles to the detriment of merovoltine or bivoltine organisms and long-lived taxa, reflects a selection of organisms capable of rapid colonisation and adaptation to such low stability habitats, where fine sediment is frequently mobilised (Larsen et al., 2011). In addition, deposited sediment fills interstices and reduces porosity, affecting the larger organisms which require larger interstitial space (Larsen and Ormerod, 2010).

The habit trait group, which indicates invertebrate locomotion and attachment, also proved to be a responsive measure to sedimentation. In the Isábena only crawlers and swimmers responded markedly to fines: while crawlers rapidly disappeared with increasing sedimentation, swimmers appeared to be the most tolerant group. This positive correlation of swimmers with fine sediment may result from the ability of these organisms to move out of the most impaired areas; crawlers move slowly and thus may not be able to escape from areas experiencing sedimentation.

Feeding activities have been reported to be affected by deposited sediment in a variety of ways. For example, Rabeni et al. (2005) studied changes in functional feeding groups associated with sedimentation and found a greater proportion of gatherers and a lower proportion of filterers and scrapers. They also concluded that trophic or feeding groups were more sensitive to deposited sediment than the habit group. Contrary to these results, trophic composition in the Isábena did not show such a strong relationship with sedimentation. Certain feeding groups, such as deposit feeders, were apparently favoured by deposited sediment. Filterers appeared to be the most sensitive group, while the other groups did not show marked changes in their frequencies across the fine sediment gradient. Filterers have been considered the most intolerant group, as respiration structures are clogged by fine particles (Lemly, 1982; Wood and Armitage, 1997). Scrapers, which feed on periphyton attached to the substrate and need clean surface on which to graze, have usually been found to be intolerant (Wasserman et al., 1984; Rabeni et al., 2005; Oliver et al., 2012). A decrease in this group was not seen in the Isábena, and in fact there was a slight increase in their representation in sediment-rich locations. Published work on trophic composition is rather contradictory, with several studies reporting that functional feeding groups are not useful in detecting ecological impairment. For example, Culp and Davis (1983) reported that feeding groups did not change significantly throughout the year between sediment impacted and unimpacted sites. Duncan and Brusven (1985) also found similar proportions of feeding groups in logged and unlogged streams. However, these results may be attributable to the dominant influence of other abiotic controls, such as bed scour and flow hydraulics, leaving changes in food resources of secondary importance (Relyea et al., 2000). Overall, the results for feeding were ambiguous and less clear cut in the Isábena than for other functional traits.

The reduction of the availability and quality of the habitat in the Isábena led to a sequence of non-random species loss which strongly depended on ecological and biological traits. The systematic drop-out of species over the sediment gradient promoted nested patterns across the catchment, with rare taxa limited to areas with low volumes of fines. In particular, trait categories conferring resilience (i.e. small size, short generation time and polivoltinism) increased in frequency in sediment-rich locations. The same trait selection and nested patterns were found in the experimental study conducted by Larsen et al. (2011) and in the sediment-rich sites over larger areas studied by Larsen and Ormerod (2010). Results from the Isábena provide further evidence that nestedness is a consequence of a non-random colonisation–extinction pattern driven by the selection of pre-adapted generalist species.

Benthic invertebrates play an important role in ecosystem functioning, as they serve as a food source and convert allochthonous and autochthonous plant material into energy for higher order organisms. In the Isábena, high sedimentation levels appeared to promote changes in invertebrate trait proportions. Such changes have potential implications for secondary production, biomass and metabolism and litter decomposition (Duncan and Brusven, 1985). Assessment of such implications would help develop more fully our understanding of the effects of fines on river ecosystem function and, in turn, provide the basis for more holistic and effective stream integrity assessment tools (Feio and Doledec, 2012).

4.2. Performance of the diversity metrics

Invertebrate diversity metrics calculated for the Isábena samples differed in their sensitivity to fine sediment content. Taxon richness and invertebrate density decreased markedly with increasing fine sediment content of the river bed. Previous studies have reported declines in abundance of certain invertebrate groups following an increased input of fine sediment, most notably plecopterans (Culp and Davis, 1983). Even though invertebrate density and taxon richness may be useful in distinguishing between unaffected sites and highly impacted ones (i.e. sites showing abundant assemblages or very poor ones respectively), they may not be useful in distinguishing intermediate levels of disturbance. This may be due to the fact that fine sediment favours some taxa at the expense of others, and, since these indices do not take into account any measure of differing relative abundances, they may fail to detect more subtle ecological changes.

Metrics that incorporate measures of equitability or the relative abundance of taxa have long been used to detect differences between impacted and unimpacted streams (Letterman and Mitsch, 1970). Shannon's index values in the Isábena decreased as fine sediment content increased, a result consistent with that of Sarver (2003). This index can also be used as a heterogeneity measure as it takes into account species' relative abundance. Shannon index values indicated a homogenising effect of fine sediment on invertebrate assemblages in the Isábena, with sites where the largest accumulations occurred being dominated by a small number of taxa (Fig. 7). However, values of evenness did not show a marked response to fine sediment accumulation. Kilgour et al. (2004), who assessed the sensitivity of several diversity metrics to different stressors, also found evenness to be less sensitive than taxon richness and Shannon's index. Larsen et al. (2011) concluded from their experimental studies that deposited sediments did not affect diversity index values. They attributed this to the loss of rare taxa at greater sediment cover, which led to fewer taxa that had an even distribution of individuals between them. Additionally, as pointed out by Peru and Dolédec (2010), natural assemblages rarely have a perfectly even distribution of taxa; this adds a degree of uncertainty to the use of metrics which are based on the assumption of the evenness of undisturbed systems.

EPT metrics are frequently used in bioassessment. In the Isábena, EPT density and EPT richness decreased as the fine sediment content of the sampled patches of bed increased, as found by Gray and Ward (1982) and Waters (1995). These taxa are expected to respond to changes in substrate composition and to show lower values of abundance and richness in streams with large volumes of deposited sediment (Pollard and Yuan, 2010). Nevertheless, in our study the %EPT metric did not respond in this way, but instead showed a weak positive relationship with fine sediment. This finding is in line with Relyea et al. (2000), who could not discriminate among streams with varying levels of fine sediment using %EPT. The pattern in %EPT values in the Isábena was due to the fact that while invertebrate assemblages at sites with high values of fine sediment comprised only a small number of individuals, these were often dominated by EPT taxa (notably *Baetis* and *Hydropsyche*). This led to the increased values of the %EPT metric with increasing sedimentation. Previous studies have indicated that some EPT taxa are relatively tolerant of fine sediment. For example, Angradi (1999) and Kaller and Hartman (2004) reported a positive correlation between the metric % Baetidae of Ephemeroptera and fine sediment. Ulfstrand (1975) reported that several groups of mayflies (i.e. Baetidae and Heptagenidae) have certain properties that confer resilience (i.e. less specialised feeding habits, high fecundities and short generation times). Thus, the specific ecologies of these animals need to be borne in mind when drawing conclusions from EPT metrics, particularly %EPT.

Taxon-based metrics, because they only take into account the presence/absence and relative abundances of species, may fail to detect functional changes occurring in response to habitat modification (Peru and Dolédec, 2010). The use of trait-based metrics should help overcome such problems. Some studies have pointed to functional diversity as being the most versatile metric as it provides an indication not only of species number and dominance but their functional role in the assemblage (Gallardo et al., 2011; Mouillot et al., 2006). A high proportion of the variability in functional metrics values computed for the Isábena could be accounted for by variability on fine sediment (Table 4). Rao's coefficient was indicated by Lêps et al. (2006) to be a good candidate for an efficient functional diversity index and our results support this assertion.

4.3. Fine sediment and flow regimes

The Mediterranean climate is relatively predictable, with marked floods occurring mainly during spring and autumn. Streams in this region are physically, chemically and biologically shaped by sequential, predictable, seasonal patterns of flooding and drying (Gasith and Resh, 1999). Floods will affect channel morphology of the stream, and depending on their magnitude, may impact reach scale morphology and local grain-size distributions. The erosion of fine sediments from badlands and their transport to river channels is not just controlled by rainfall intensity, but many other factors (e.g. vegetation cover, slope, connectivity). In the case of the Isábena, event-based observations in an experimental badland show that freeze-thawing processes in winter eventually loosen the materials and increase the sediment supply in subsequent rainfall events. Therefore, theses processes control variability in fine sediment stored in the channel over the annual scale and consequently bed grain size distributions within stream reaches (López-Tarazón et al., 2011a).

As well the transport and deposition of fine material, changes to flood magnitude and frequency may facilitate or interfere with colonisation by invertebrates (Kochersberger et al., 2012). In the case of the Isábena, colonisation and recovery following episodes of fine sediment deposition may be impeded by the baking of exposed sediment which occurs during the hot summer months. Fine sediment present across the channel is exposed as flows drop and, once dried, the bed is likely to become impenetrable, preventing exchange between surface and groundwaters, inhibiting the use of interstitial refugia by invertebrates and affecting the dissolved oxygen levels in the surface and shallow hyporheic zones. Even once rewetted by subsequent flow events, baked (cohesive) sediment may not be immediately removed because of its relatively high entrainment threshold. Such cohesion will likely perpetuate the ecological effects of episodes of sedimentation, although this has yet to be studied. Similarly, the effects of longer term (seasonal, annual) changes in patterns of sedimentation in the Isábena remain unknown.

5. Conclusions

The aim of this study was to assess the effects of naturally occurring fine sediment on an otherwise relatively undisturbed catchment. We examined invertebrate assemblage taxonomic structure and functional traits and assessed the ability of a set of commonly used biodiversity metrics to detect fine sediment impacts. Overall, results were in line with those of studies of sedimentation effects in anthropogenically modified catchments, with a decline in species richness, total density and trait diversity with increasing fine sediment content on the river bed.

Functional traits proved to be useful to understand the mechanisms responsible for observed patterns in the abundance and distribution of macroinvertebrates in the Isábena. Life history traits that conferred resilience to populations in sediment-rich locations were influencing the nested patterns observed across the catchment. Results help support the argument that nested patterns can result from fragmentation of the habitat due to changes in its quality, and not only as a consequence of larger-scale biogeographical processes (Hylander et al., 2005; Larsen and Ormerod, 2010).

Climate change, habitat modification, fragmentation and loss are leading to unprecedented rates of diversity loss. Even though efforts are being made to detect and identify the causes of impairment and the organisms at risk, there is still limited understanding of which the types of organisms should be targeted for protection. The study of trait responses and nested patterns could have considerable significance for the identification of sensitive species and hence their conservation (Larsen and Ormerod, 2010). The present study provides further support for the view that habitat change and fragmentation associated with excessive fine sediment loads are resulting in the elimination of species with sensitive traits, and that this is promoting nested patterns.

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