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Effects of fine sediment inputs from a logging road on stream insect communities: a large-scale experimental approach in a Canadian headwater stream

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Abstract

A forest headwater stream was manipulated (logging road-crossing amended) to induce fine sediment inputs. Benthic inorganic sediment concentrations (particles 1.5–250 μm) increased from a 2-year pre-disturbance average of about 800 g m^{-2} to over 5000 g m^{-2} that persisted for 3 years. Aquatic insect communities were examined over the 5-year study period in the manipulated and nearby reference streams. Overall, the effects of the fine sediment increases on aquatic insect communities were minimal. There were no significant effects of sedimentation on total aquatic insect abundance or biomass. An index of multivariate dispersion gave no evidence of community stress at the manipulated site. Multivariate ordination plots and time trends among univariate community metrics indicated only subtle changes in community structure. Among the univariate metrics (16 time series analyses in total), six gave evidence of a sediment impact on aquatic insect communities. Of those, the clearest indications of an effect were small reductions in diversity and richness of spring communities. These resulted from a significant decline in the proportion of spring shredders, accompanied by a significant increase in the percent Chironomidae. This large-scale experimental approach integrated the realism of a whole-stream study with the control of a manipulative study by including pre-manipulation measurements and excluding other confounding catchment disturbances. In this regard, it may provide a more realistic measure of benthic community-level responses to sedimentation in streams at a magnitude associated with logging activity than many previous studies.

Introduction

Catchment disturbances that result in fine sediment inputs to streams have been identified as major contributors to the degradation of freshwater habitats and to impacts on aquatic biota (Waters 1995). In particular, agricultural tillage, urbanization, mining, road construction and logging activity can result in fine sediment inputs and deposition in stream beds. The biological consequences can include declines in benthic invertebrate abundance and/or changes in community structure (Tebo 1957; Somer and Hassler 1992; Wood and Armitage 1997; Fossati et al. 2001). However, most studies that have examined the effects of fine sediments in streams on benthos have relied on small-scale experiments in artificial stream channels or flumes (Rosenberg and Wiens 1978; McClelland and Brusven 1980), or on comparative studies between reference and sediment-impacted sites after the disturbance or along a gradient of sedimentation (Lenat et al. 1981; Ermin and Ligon 1988; Hogg and Norris 1991; Barton and Farmer 1997; Zweig and Rabeni 2001). These provide valuable information on the potential effects of sedimentation on stream insects, but may not be representative of real-world responses by benthic insects to fine sediment inputs because of limitations in scale and the lack of baseline or pre-disturbance community measures.

Logging activities, particularly in recent years under best management practices, often produce only moderate amounts of fine sediment deposition on stream beds in comparison to more destructive land use activities (Wood and Armitage 1997). Reported effects of fine sediments on stream habitat and benthos from logging (Campbell and Doeg 1989; Webster et al. 1992; Binkley and Brown 1993; Keenan and Kimmins 1993) may have been confounded by other catchment disturbances such as canopy modification or changes in nutrient flux (Murphy et al. 1981; Growns and Davis 1994). While some studies have demonstrated adverse effects on benthos under conditions of heavy sediment loading, others have detected little or no response to more moderate fine sediment increases (Culp et al. 1985; Fairchild et al. 1987). Thus it is not clear to what extent stream benthos is adversely affected by sedimentation at magnitudes associated with logging activity, especially logging road construction or maintenance. Models have been developed to estimate sediment yield from logging activities (Burns and Hewlett 1983) from which sediment deposition in streams could be predicted, but few studies have linked actual benthic sediment concentrations to responses by benthic invertebrates under realistic conditions.

To examine benthic insect responses to stream sedimentation under realistic conditions, we measured fine sediment concentrations and aquatic insect communities in a headwater stream below a logging road crossing. No actual logging or any other disturbance occurred in the stream catchment. Sampling was initiated two years before the road crossing was disturbed, and continued for three years after. In this sense, the study was a stream reach-scale manipulation providing pre-disturbance baseline measurements and comparisons to a nearby reference stream. This approach integrated the realism of a natural stream reach study with the experimental control of a manipulative study by including pre-sedimentation measurements and excluding other catchment disturbances. The study differed from most previous sediment impact studies in that it 1) focused on time course analyses over five years, examining trends before and after the sedimentation at the impacted and reference sites, 2) was not confounded by other catchment disturbances such as canopy modification or altered nutrient fluxes, and 3) quantified and characterized stream bed sedimentation.

Materials and methods

Study site

The study was conducted in the Turkey Lakes Watershed, a 10.5-km^2 area located approximately 60 km north of Sault Ste. Marie, Ontario, Canada, and 13 km inland from Lake Superior. The watershed is dominated by old-growth sugar maple (*Acer saccharum* Marsh.) with scattered yellow birch (*Betula alleghaniensis* Britton). The two streams sampled for this study were first-order streams draining hardwood forest catchments with channel slopes of about 10%. Average bankfull width of the reference stream was 1.9 m, while the sediment impacted stream had an average bankfull width of 1.6 m. Study reaches were 120 m in both streams. Bottom substrates were characterized by visual estimates of percent composition of sand, gravel, rubble and boulder in 10-m sections and averaged over the study reaches. The reference stream was 13% boulder, 34% rubble, 48% gravel and 5% sand, and the sediment impacted stream was 8% boulder, 25% rubble, 58% gravel and 9% sand.

Sampling sites in the manipulated catchment were downstream from an old road crossing. The road crossing (gravel road over culvert) was considered stable in that there were no indications of erosion or sediment inputs from the road at the beginning of the sampling period, and pre-disturbance benthic sediment concentrations were similar to sites without road crossings. For example, in the fall of 1996 (one year before the disturbance) average fine sediment concentration at the manipulated site was 680 g m^{-2}, and at the reference site was 280 g m^{-2}, while in three other nearby streams without road crossings average concentrations were 214, 536, and 1380 g m^{-2}. In preparation for logging in the watershed, the road crossing at the manipulated site was upgraded to facilitate logging truck access. The road was widened, boulders removed, and ditches cleared on either side to increase drainage. No sediment control measures were taken during the road amendment.

Benthic sediment concentrations

Procedures for collection, processing, and analyses of fine sediment samples are given in Kreutzweiser and

Capell (2001). Briefly, fine sediments (< 250 µm) were collected from stream beds with a 17.5-cm diameter core sampler in which the substrates were stirred vigorously with a steel rod to suspend the fine particles. A manual bilge pump with a 2-mm screened intake was used to pump 3.8 l of water with suspended sediments through a 250-µm sieve into a collection bucket. The water and sediment that passed through the sieve were stirred vigorously and a 25-mL subsample was drawn with a syringe and placed in a glass vial. The sediments were suction filtered on to 1.5-µm glass fibre filters, dried at 60 ºC for 48 h, weighed, then placed in a muffle furnace at 500 ºC for 2 h, and weighed again to determine the organic (ash-free dry weight) and inorganic fractions. This provided a sediment size fraction of 1.5–250 µm and the sediment concentration on stream beds was expressed as g m^{-2}. Ten sediment bed load samples were collected from each stream at each sample time. The samples were collected in a stratified random manner from each section; immovable substrates, plunge pools, and distinct depositional areas were avoided to ensure that sediment bed load measures were taken from similar habitats in which stream benthos was measured. Benthic sediment samples were collected in late spring (end of May) and in early fall (early October) of each year from 1995 to 1999. The road amendment occurred in the late summer of 1997 so the sampling regime provided 5 pre-disturbance and 5 post-disturbance measurements. Differences in trends over time of sediment loading on stream beds between reference and disturbed sites were examined following the BACI approach of Stewart-Oaten et al. (1986), but with randomization tests to detect nonrandom patterns in differences between reference and the sediment impacted site before and after the disturbance.

The average surface cover of deposited sediment was visually estimated in both streams under baseflow conditions at the end of the study period. Fifteen 1-m wide transects were placed randomly within each study reach. The percent of stream bottom covered by fine sediment, in 5% increments, was visually estimated at each transect by a single observer. This approach was used by Zweig and Rabeni (2001) and found to be as effective as calculated embeddedness measures in representing the extent of deposited sediments as a proportion of stream bottom area. Differences between the two sites in sediment surface cover were compared by ANOVA following an arcsine/square root transformation for percentages.

Aquatic insect communities

Aquatic insect communities were quantitatively sampled with modified Surber-type samplers. The collection net was 0.32 m wide with a 363 µm mesh, and the side walls were 1 m long. The side walls were held in place by rods driven into the stream bed at the upstream ends of the samplers, and the total area sampled was 0.32 m^2. Ten replicate samples were collected by random allocation at each study reach each time the sediment core samples were taken. Plunge pool or boulder locations were excluded. The contents of the collection net were preserved in 10% formaldehyde and dyed with phloxine B to facilitate sorting under a magnifier. Insects were identified to genus or species, except for Chironomidae. Seven of the 10 sorted and identified samples from each site and time were randomly selected, dried at 60 ºC for 72 h, weighed and recorded as total insect biomass per sample. On every sampling occasion, insects comprised more than 99% of the total macroinvertebrates collected, and other invertebrates occasionally encountered (e.g., gastropods, oligochaetes, hirudinids) were excluded from the analyses.

Insect community data analyses

This was an un-replicated, whole-stream study design with emphasis on comparisons over time among response variables between the manipulated and reference streams. Because the insect communities were sampled twice each year (in late May and early October), the data were separated into spring communities and fall communities and analyzed separately to reduce seasonal variability among annual comparisons. Aquatic insect community structure was examined by a multivariate ordination procedure, nonmetric multidimensional scaling (NMDS) (Clarke 1993). For these analyses, data were restricted to those taxa that contributed at least 1% of the total abundance on at least two sample dates. This adjustment reduced the confounding effects of highly uncommon species in the data. Data were log transformed ($\log x + 1$) to down-weigh the influence of abundant taxa and take into account rarer taxa as well. Average abundance for each taxon at both sites at each sample time was used in the NMDS ordination. Two-dimensional scatter plots were constructed from the first two ordination axes in NMDS with each point representing the average community structure of a given site at a specified sample time. The NMDS points for both sites were tracked over time through ordination space to determine if a divergence of trends occurred after the disturbance. Differences in community structure between sites were determined by a multivariate analysis of similarity (ANOSIM), which is based on a non-parametric permutation procedure applied to the ranked Bray-Curtis similarity matrix underlying the NMDS ordination (Clarke 1993). The similarity matrices were further analyzed to provide multivariate measures of community stress. An index of multivariate dispersion (IMD) was computed which contrasted the ranked similarities between sites averaged over time. The IMD values range from -1 to $+1$. Values near zero imply no differences between groups, whereas values near the extremes infer that variability in multivariate structure differs between groups. Relative dispersion values were also computed for both sites, with larger values corresponding to greater within-group variability (Clarke and Warwick 1994).

Insect communities were further examined by a number of univariate metrics taken from Davis et al. (2001). These included mean abundance, biomass, taxon richness, percent shredders (functional feeding group classifications following Merritt and Cummins 1996), percent scrapers, percent filterers, percent EPT (Ephemeroptera, Plecoptera, Trichoptera), and percent Chironomidae. We also included Shannon-Wiener diversity and Margalef's richness indices. Trends over time in these univariate metrics were compared between sites by a repeated measures orthogonal polynomial coefficient analysis (Meredith and Stehman 1991). This approach accounts for the correlation among samples drawn from the same experimental unit over time. It also focuses on the response curve, i.e., a comparison of changes over time among groups, and does not require the stringent assumptions of a split-plot analysis. Trends were examined by linear response models, and the treatment X time interaction in the analysis was the primary term of interest for detecting significant treatment effects. Significant treatment X time interaction ($p < 0.05$) indicated that trends over time were not parallel between the two sites. A significant divergence over time was taken as a treatment (sedimentation) effect. Abundance data were log transformed, and percent data were arcsine/square root transformed to improve normality and increase homogeneity of variances. The Shannon-Wiener diversity and Margalef's richness indices were based on average abundance data at each time, precluding statistical comparisons between sites, and trends over time for these indices were examined visually.

Results

The road amendment resulted in a significant increase (BACI $p=0.0032$) in fine sediment bed load (from an average 2-yr pre-disturbance level of 800 g m^{-2} to about 5000 g m^{-2} after the disturbance) that was sustained for the duration of the study period. Most of the increased sediment (about 80%) consisted of particles in the coarse silt size range of 40–63 μm (Kreutzweiser and Capell 2001). By the end of the study period, the average visible surface cover of deposited sediment at the manipulated site was 46.7% and at the reference site was 14.5% (ANOVA $p=0.0006$). The inorganic fraction of fine sediments increased from a pre-disturbance average of 85.9% to 97.1% (ANOVA $p=0.0299$) at the manipulated site, while the average inorganic fraction of fine sediments at the reference site was 82.1% before the disturbance and 81.2% after.

There were no measurable effects of sedimentation on total insect abundance in spring samples. Mean abundance of spring samples was consistently higher in the reference than in the manipulated site, but this occurred before as well as after the sediment inputs (Figure 1A), and trends over time were essentially parallel (time X treatment interaction $p=0.6737$). There was also little indication of a sedimentation effect on abundance of aquatic insects in fall samples. By the fall of 1999, total abundance at the reference site showed a sharp increase (Figure 1B), and this resulted in a marginally significant divergence in trends (time X treatment interaction $p=0.0750$). The large error associated with the mean estimate in fall 1999 indicates a highly patchy distribution within the study reach, and most of this increased abundance (90%) was comprised of Chironomidae. When the fall 1999 samples were removed from the repeated measures analysis, trends over time between sites were similar (time X treatment interaction $p=0.5269$).

There was also no indication that the increased sediment affected total insect biomass. Biomass in spring samples from the manipulated site tended to be lower than in the reference, but this trend was apparent before the sediment inputs (Figure 1C) and there were no detectable differences in time trends overall (time X treatment interaction $p=0.8581$). Total biomass in fall samples was more variable, and while there tended to be a post-disturbance decline at the manipulated site, followed by an increase the following year (Figure 1D), no significant difference in time

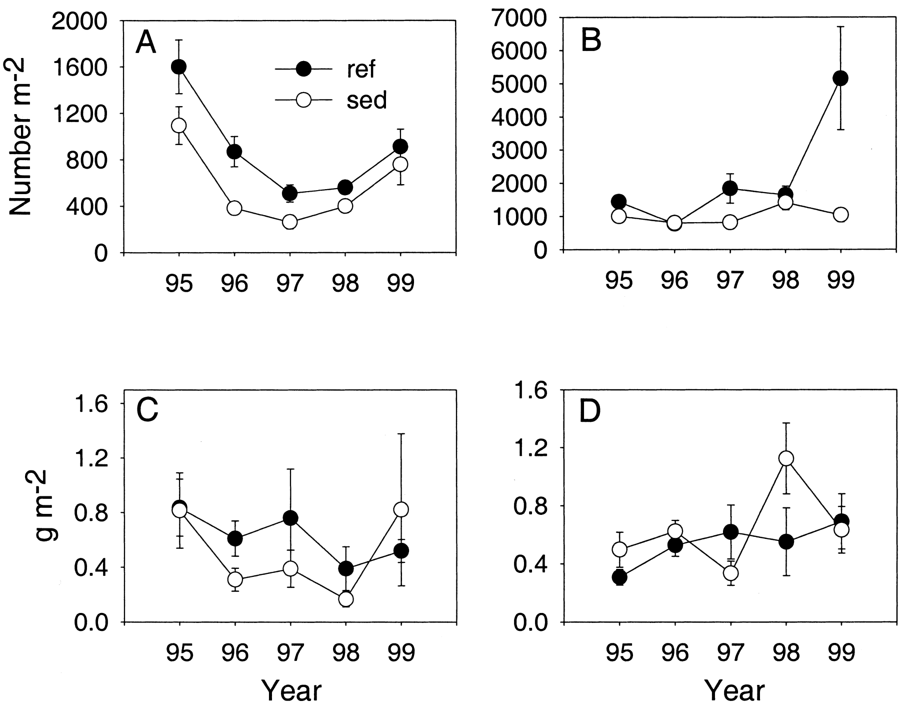

Figure 1. Mean (± 1 SE, n=10) abundance of aquatic insects in spring (A) and fall (B), and mean total biomass of aquatic insects in spring (C) and fall (D) samples from the reference (ref) and sediment (sed) sites. Disturbance occurred in late summer 1997.

trends could be detected (time X treatment interaction p=0.9800).

The NMDS ordination showed that among both spring and fall samples, there were distinct differences in community structure between the two sites both before and after the sediment inputs (Figure 2). The ANOSIM procedure detected significant differences between sites ($p < 0.01$) at all paired site comparisons at each sample time both before and after the disturbance. The distinct grouping of samples by site indicated that there were greater differences between sites than between years, although there were also considerable shifts in community structure over time.

In spring samples, community composition changed in roughly the same direction at both the reference and disturbed site before the manipulation, but to a greater extent at the manipulated site (Figure 2). There was little change in community structure at the reference site between the third and fourth sample times (last pre- and first post-disturbance) while a slightly greater shift occurred at the manipulated site. Between the fourth and fifth sample time, the community structure at both sites shifted in ordination space but the distance was slightly greater and in a different direction at the manipulated site than at the reference. Changes in community structure of fall samples between the first two (pre-disturbance) sample times were of similar magnitude and direction (Figure 2). Post-disturbance trends among fall communities were more divergent. Between sample times 2 and 3, there was a comparatively large shift in community structure at the reference site, while the shift in community structure at the manipulated site was much smaller and in the opposite direction. Over the next two post-disturbance sample times, the shifts in community structure at the manipulated site were slightly greater and in different directions than at the reference site. Overall, the ordination indicated that there were distinct differences in community structure between the manipulated and reference sites, before and after the sediment inputs. However, the slightly divergent trends through ordination space over the post-disturbance period suggested some changes in community structure in response to the sedimentation.

The index of multivariate dispersion (IMD) value for a comparison of variability in multivariate structure between sites averaged over time was -0.16. This low value indicated that there was little difference in multivariate structure between sites overall, and the negative sign indicated that there was greater

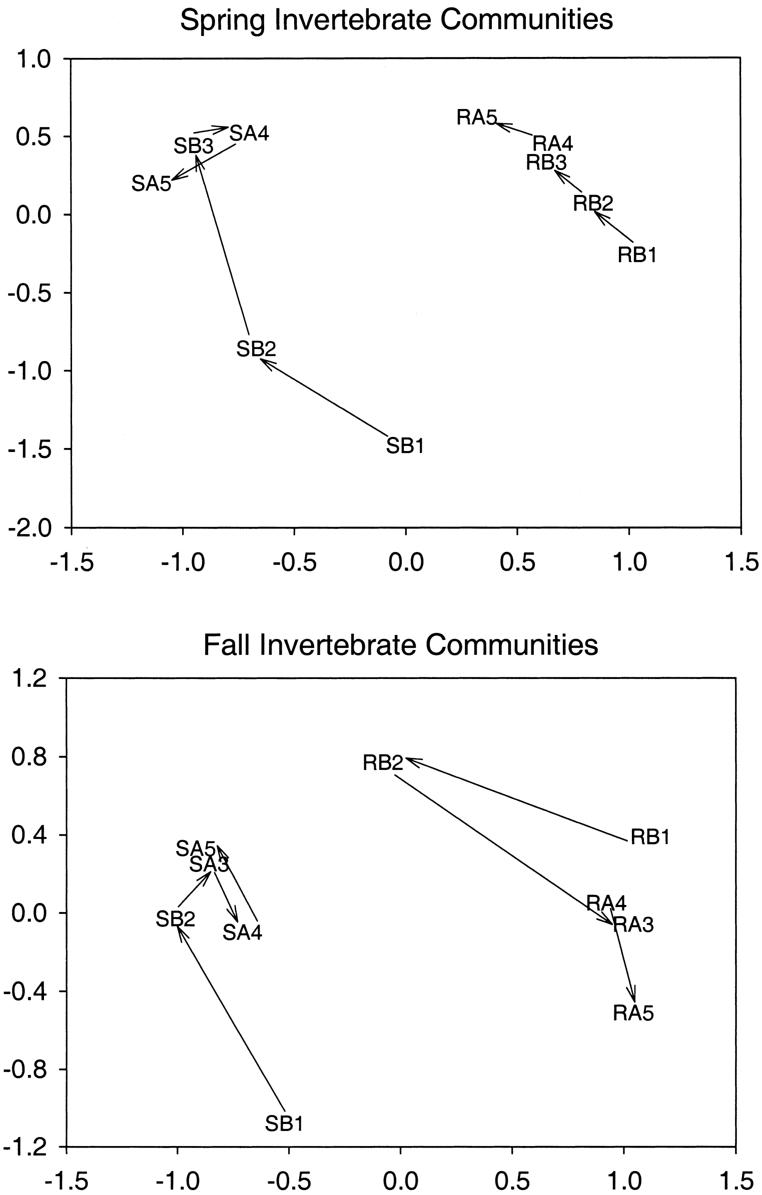

Figure 2. Two-dimensional plots from NMDS ordination of spring and fall invertebrate communities. At each point, the first letter indicates site (S = sediment site, R= reference), second letter indicates before (B) or after (A) the sediment inputs, and the numbers indicate sequential sample times.

variability among reference samples than among manipulated site samples (see Clarke and Warwick (1994) for interpretation of IMD). This was further indicated by the relative dispersion values based on the ranked Bray-Curtis similarity matrix. The value for the reference site was 1.076 and for the sediment site was 0.924, indicating that there was slightly higher within-group variability in multivariate structure at the reference site than at the manipulated site.

The subtle changes in community structure at the sediment site in comparison to the trends at the reference site were examined further by the community-level, univariate metrics. The Shannon-Wiener diversity and the Margalefs' richness indices both showed a declining post-disturbance trend among spring communities at the manipulated site in comparison to the reference site (Figure 3A, Figure 3C). Trends in diversity and richness were less apparent among fall

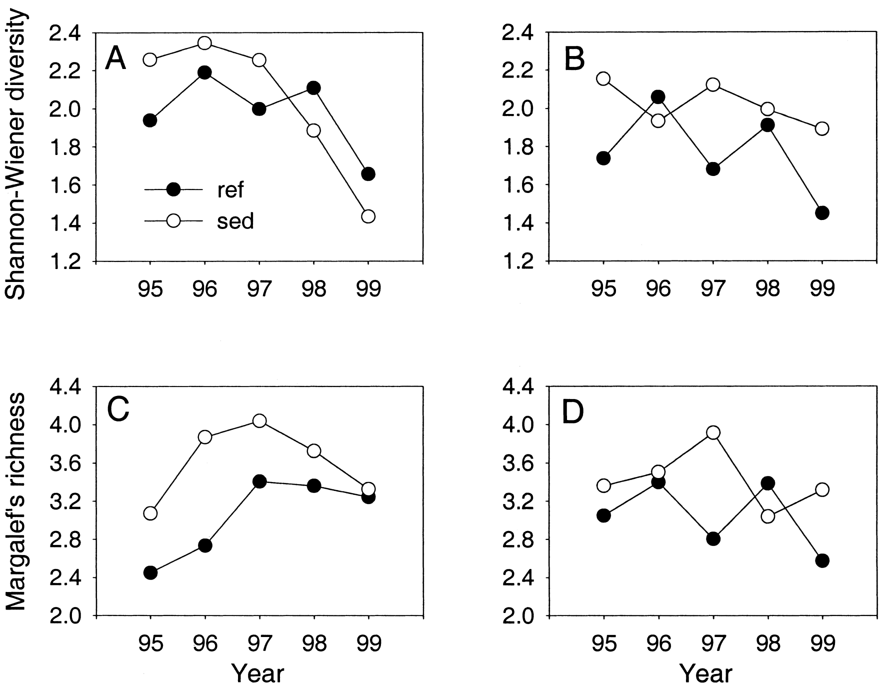

Figure 3. Shannon-Wiener diversity (A and B) and Margalef's richness indices (C and D) based on mean relative abundances from reference (ref) and sediment site (sed) samples in spring (A and C) and fall (B and D) of each year. Disturbance occurred in late summer 1997.

communities, and year to year differences were as great or greater than site differences with no clear indication of a sediment effect on fall communities (Figure 3B, Figure 3D).

Among the six community metrics that were examined by the repeated measures analysis, differences between sites in time trends overall were small but tended to be more pronounced in spring than in fall. Three metrics showed significant divergence over time in spring (Figure 4). The percent shredders in spring communities clearly declined at the manipulated site after the disturbance (time X treatment interaction $p < 0.0001$). The spring EPT component also declined at the manipulated site, but reflected a similar decline at the reference site over the same time. The greatest difference in percent EPT between sites occurred before the disturbance, and this contributed to the significant difference (time X treatment interaction $p < 0.0001$) in EPT trends overall. The percent Chironomidae increased at the manipulated site after the disturbance, and while there tended to be an increase at the reference site as well by the end of the sampling period, a significant difference in time trends between sites was detected (time X treatment interaction $p = 0.0006$). Taxa richness declined at the impacted site (Figure 4), but most of this decline occurred before the disturbance and the time trends overall were only marginally different between sites (time X treatment interaction $p = 0.0897$). There were no clear treatment-related patterns or significant differences in time trends between sites among spring scraper and filterer community metrics.

Only one of six community metrics among fall samples showed a significant divergence in trends (Figure 5). The filterer component of fall communities at the sediment site declined slightly after the disturbance, and as a result of increases in the percent filterers over the same period at the reference site, the time trends were significantly different between sites (time X treatment interaction $p = 0.0246$). The significant decline in percent shredders that occurred in spring samples was less apparent in fall samples. There was a tendency for lower percent shredders in fall samples on two occasions after the disturbance, but no significant difference between sites could be detected (time X treatment interaction $p = 0.6887$). There was no evidence of sediment impact among the four remaining metrics for fall communities (Figure 5).

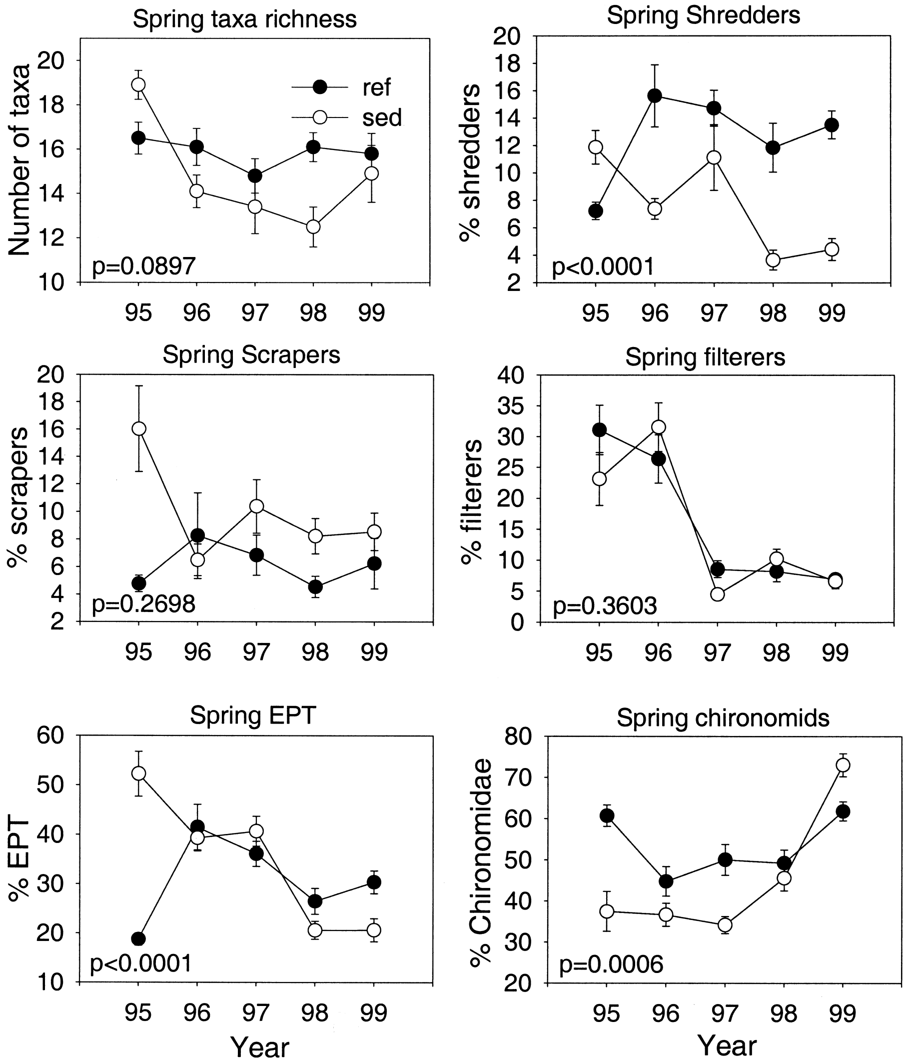

Figure 4. Community-level metrics (mean ± 1SE, n=10) for reference (ref) and sediment site (sed) samples in spring of each year. Probability values are from time X treatment interaction terms in repeated measures orthogonal polynomial coefficient analyses. Disturbance occurred in late summer 1997.

Discussion

Despite large and sustained increases in fine sediment deposition in a forest stream (about a 7-fold increase for 3 yr), only small changes in aquatic insect community structure could be detected. The NMDS ordination demonstrated that over the 5-yr study period there were generally greater differences in community structure between sites than among years, but that these differences were as large or larger before the sediment inputs. There was only a subtle divergence in ordination time trends between sites after the disturbance. The multivariate measures of community stress (indices of multivariate dispersion) indicated little difference between sites overall, with a tendency for higher variability in multivariate structure among reference samples. Of the univariate metrics examining spring and fall communities at the manipulated and reference sites, the clearest indications of an effect on community structure were small reductions in diversity and richness of spring communities. These were primarily the result of a decline in the proportion of spring shredders after the disturbance, accompanied by a significant increase in the percent Chironomidae. None of these metrics gave evidence of a sediment impact on fall communities.

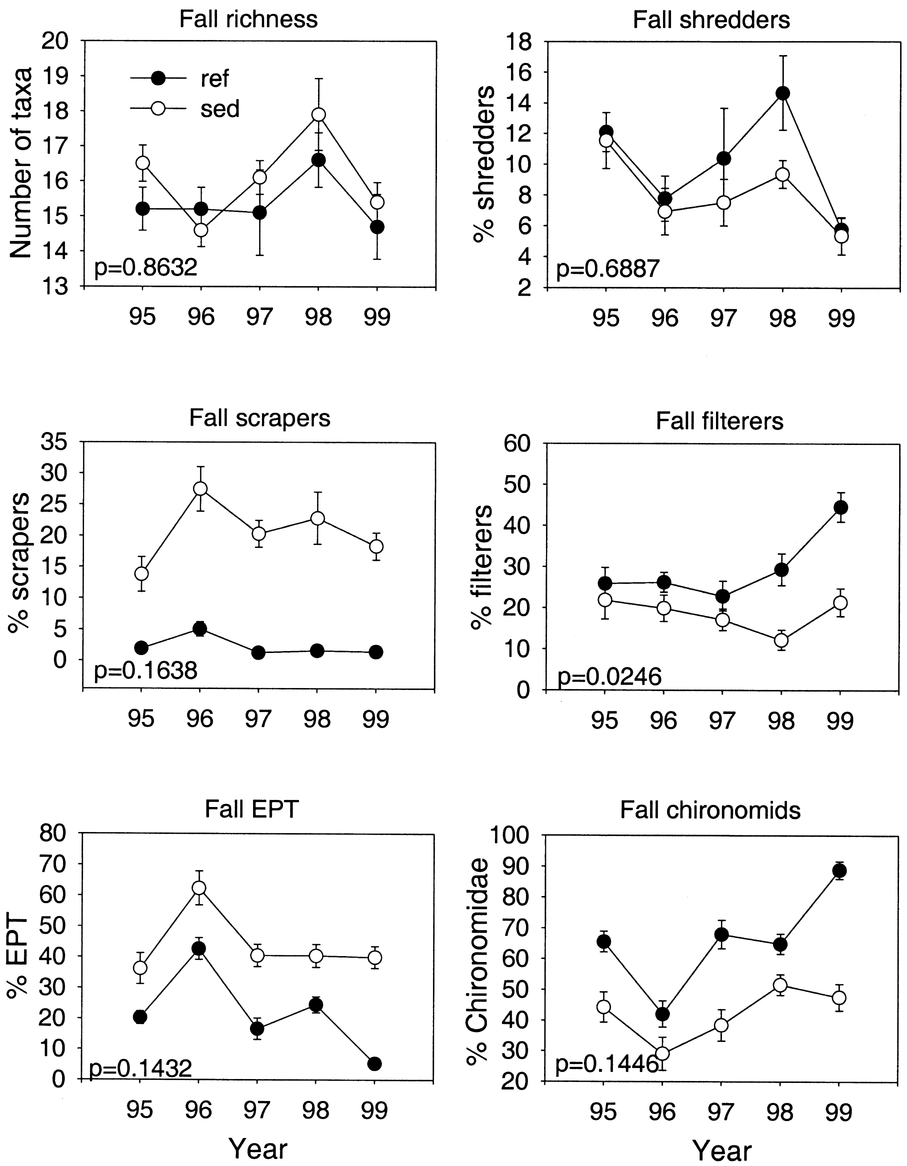

Figure 5. Community-level metrics (mean ± 1SE, n=10) for reference (ref) and sediment site (sed) samples in fall of each year. Probability values are from time X treatment interaction terms in repeated measures orthogonal polynomial coefficient analyses. Disturbance occurred in late summer 1997.

The decline in Shannon-Wiener diversity and Margalef's richness indices of spring communities at the manipulated site reflected shifts in relative abundance among taxa, not loss of species. Richness tended to be lower at the manipulated site, but the greatest reduction in mean number of taxa occurred before the sediment inputs. The increase in the proportion of chironomids in spring communities at the manipulated site is consistent with their general tolerance of sediment-degraded habitats (Davis et al. 2001), although Angradi (1999) showed differential responses among Chironomidae subfamilies to sediment additions. However, changes in the proportion of chironomids also occurred in fall communities at the reference site suggesting that natural environmental factors are as likely to cause shifts of this magnitude as the sediment inputs from the road crossing. The significant decline in the proportion of spring shredders suggests that the sedimentation affected the habitat or resource quality for this functional group.

Since there were no catchment disturbances to alter organic matter inputs at the sediment site, the effects on shredders presumably occurred from siltation of the coarse particulate matter on which this functional group feeds. This could result from physical interference by sediment particles on feeding behaviour, or from reduced palatability to shredders through a reduction in microbial colonization and conditioning of particulate organic matter surfaces. There was no indication of these effects on fall shredders, and this may have resulted from shredder utilization of fresh litter inputs from leaf fall before impingement of organic matter surfaces by fine sediments from high-water events.

It is noteworthy that the multivariate ordination, the analysis of similarity, and many of the univariate community metrics showed a clear distinction in community structure between the two sites, before the sediment inputs. The reference site was selected because of its apparent similarity to the manipulated site. Catchment characteristics, bankfull widths, study reach lengths and orientation, forest composition, canopy structure and density, environmental influences (given the proximity of sites), and substrate qualities were similar at the two streams. The differences in aquatic insect community structure were consistent over the study period, with greater differences between sites than among years within a site. The distinct natural differences in community structure of these paired streams underscores the need for establishing baseline or pre-treatment conditions to which after-impact conditions can be compared in unreplicated, whole-stream experiments such as this one (Stewart-Oaten et al. 1986). A paired-stream comparison after the disturbance would have indicated distinct differences in aquatic insect community structure suggesting a marked effect of sedimentation on benthic insects, but it is clear that distinct differences in community structure in these streams were present before the disturbance.

Many other studies have shown significant effects of fine sediment deposition on stream benthic invertebrates (Tebo 1957; Barton 1977; Lenat et al. 1981; Erman and Ligon 1988; Barton and Farmer 1997; Fossati et al. 2001). These, and others, have indicated that benthic insect communities are generally intolerant of fine sediment deposition, and that catchment disturbances resulting in sediment inputs to streams are likely to cause reductions in abundance and/or changes in community structure (Campbell and Doeg 1989; Waters 1995; Wood and Armitage 1997). However, across a range of sedimentation studies, results have been variable. In extreme cases where sediment inputs have been extensive, aquatic insects have clearly been adversely affected (Somer and Hassler 1992; Fossati et al. 2001). In other studies, conclusions are drawn from small-scale, controlled experiments and short-term responses (Rosenberg and Wiens 1978; McClelland and Brusven 1980). These are useful for examining response mechanisms or processes but may not adequately represent community-level responses to fine sediment deposition because they do not incorporate all the biophysical properties and processes or temporal dynamics inherent in natural systems. Many studies have made comparisons of benthic invertebrate communities among disturbed and reference sites, or across a gradient of sediment deposition or manipulation, after the sediment input disturbance. Among these, some have detected significant adverse effects (Lenat et al. 1981; Hogg and Norris 1991; Barton and Farmer 1997; Zweig and Rabeni 2001), while others have reported subtle or no effects on stream benthos (Murphey et al. 1981; Culp et al. 1985; Fairchild et al. 1987; Richards and Bacon 1994; Angradi 1999).

Some of this discrepancy among studies may arise from differences in deposited sediment characteristics and amounts. Many of the previous studies have described sediments ranging from silt- to sand-size particles up to 2mm, have not reported organic or inorganic fractions, and have quantified in terms of a measure of surface cover or concentrations of suspended sediment additions rather than actual benthic sediment concentrations. Differential invertebrate responses may reflect differences in fine sediment composition in terms of particle size and structure. While an association between substrate particle size and benthic invertebrate community structure has been demonstrated across a broad range of sizes (Minshall 1984; Bourassa and Morin 1995), it is less clear how differences in particle sizes or composition across fine sediment size groups ($<$ 2mm) may affect benthic communities. In the present study, we showed that sediment inputs from the road crossing were mostly comprised of small particles in the 40-63 μm size range, consisted almost entirely (97%) of inorganic materials, and resulted in an average benthic concentration of over 5000 g m^{-2}, roughly a 7-fold increase. Given this magnitude of sediment increase, and that the small particle sizes of this sediment were in the range of particle sizes that have been shown to interfere with hydrological exchange processes and stream bed quality (Schälchli 1992; Brunke and Gonser 1997), we expected to find marked effects on the benthic insect community.

The fact that the subtle changes in insect community structure of this study were less than in many previous studies of sediment inputs at similar magnitudes may also reflect differences in sampling frequency and study duration. Most previous studies focused on short-term (several weeks to months) measurements, or one-time comparisons, which may have detected short-term impacts, while sampling in the present study was integrated over a longer time period. More frequent sampling immediately after the sediment disturbance might have detected significant, short-term changes in insect abundance and diversity. However, our sampling frequency (5 times over 2 ½ yr after the disturbance) integrated responses over the longer term taking into account the natural stochastic variability over time and may have provided a more realistic indication of ecologically-significant effects of the disturbance than measures of short-term changes.

The minimal effects of sedimentation on aquatic insects communities of this study may have occurred because these stream substrates were largely comprised of finer materials (about 55% gravel, 7% sand) before the sediment inputs and the insect communities may have already been naturally adapted to finer substrates. A fine sediment load of this magnitude and duration in a stream with more coarse materials may have had a greater effect on benthic insects. On the other hand, stream beds of more coarse materials are less prone to embeddedness (the proportion of substrate particles covered by fine materials), and therefore may not be more susceptible to sedimentation effects on benthos.

Conclusions

Fine sediment inputs from logging activities, particularly road crossings, have been implicated in stream habitat degradation (Campbell and Doeg 1989; Webster et al. 1992; Binkley and Brown 1993; Keenan and Kimmins 1993). However, except under extreme cases, effects of sedimentation on benthic communities in large-scale field studies have been equivocal. In some cases, responses by benthic invertebrate communities to sedimentation from logging operations may have been confounded by concomitant influences of activities such as canopy modifications or other catchment disturbances. The large-scale experimental approach of the present study integrated the realism of a whole-stream study with the control of a manipulative study by including pre-sedimentation measurements and excluding other confounding catchment disturbances. The road crossing at the manipulated site was upgraded in an actual logging operation to accommodate haul trucks and other equipment, so the resulting sediment inputs to the stream should be typical of road-crossing disturbances in similar hardwood forest terrain. In this regard, the present study may provide a more realistic measure of community-level effects on benthic insects of sedimentation at a magnitude associated with logging activity, than many previous studies. That only subtle effects on aquatic insects were detected indicates that benthic insect communities of northern hardwood forest streams can exhibit some degree of resistance to fine sediment inputs at this magnitude. Additional experimental studies at this or larger scales in other hardwood forest watersheds would be useful to further evaluate the degree to which stream insect communities can tolerate fine sediment inputs from forestry operations. The implementation of sediment control strategies under best management guidelines for stream crossings would further reduce the likelihood of adverse effects on stream habitat quality and benthic communities.

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