



Current questions in water management

Extension

**Book of abstracts to the WISER final conference
Tallinn, Estonia, 25-26 January 2012**



Publisher: Eesti Maaülikool, Estonia

Edited by Astrid Schmidt-Kloiber, Anne Hartmann, Jörg Strackbein, Christian K. Feld & Daniel Hering

WISER – Water bodies in Europe

Integrative Systems to assess Ecological status and Recovery

Funded by the European Union under the 7th Framework Programme,
Theme 6 (Environment including Climate Change), contract No. 226273

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ISBN 978-9949-484-19-5



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WISER – Water bodies in Europe:
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www.wiser.eu

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Theme 6 (Environment including Climate Change)
Contract No. 226273

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Layout and compilation:
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Printed in Estonia by Ecoprint, www.ecoprint.ee

Cross-taxon responses to stress gradients in streams and lakes

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Key words: *biomonitoring, impairment, multiple taxonomic groups, regression*

Introduction

Anthropogenic activity increasingly alters the factors that affect biodiversity at local, regional and global scales, ultimately influencing the structure and function of aquatic ecosystems. Anthropogenic alteration of habitat structure and natural disturbance regimes often homogenize communities, which potentially jeopardises the provision of ecosystem goods and services (Hooper et al. 2005). Thus, different approaches have been developed to assess the ecological effects of human stressors on ecosystems (e.g. Dale & Beyeler 2001). However, surprisingly few studies have compared the discriminatory power of different approaches to detect change, and fewer still have compared the precision and sensitivity of different taxonomic groups commonly used in bioassessment (e.g. Johnson & Hering 2009).

Stream and lake systems are particularly well suited for this type of study as several taxonomic groups, such as fish, invertebrates, macrophytes and algae are routinely used in bioassessment. Here we evaluate the precision and sensitivity of multiple taxonomic groups to detect putative human-induced change in streams and lakes. Nutrient enrichment from agricultural land use is a common form of anthropogenic perturbation. It can therefore be hypothesized that primary producers (benthic diatoms, phytoplankton and macrophytes), due to their first principle relation with nutrients, would respond strongly to low to moderate changes in nutrients. This conjecture is based on the assumption that at high concentrations, nutrients are no longer limiting growth and production. Both primary (many of the invertebrate taxa) and secondary (many of the fish taxa) consumers were expected to respond to nutrient enrichment, albeit less strongly than the primary producers (i.e. benthic diatom and phytoplankton response \approx macrophyte > invertebrate > fish). Because streams are physically more disturbed, their biota are more strongly exposed to hydrological disturbance compared to lakes.

We therefore expect that taxon responses in streams to nutrient enrichment are confounded by physical disturbance (e.g. scouring from spates) and therefore weaker compared to lakes.

Specifically, we test the following three hypotheses: (i) taxonomic assemblages differ in their response to environmental stressors (ii) response signatures differ between lakes and streams and (iii) response signatures can be used to select complementary indicators.

Material and Methods

Lake and stream data used in this study come from data bases of previous EU projects and more regional data bases which have been standardised and collated in the WISER data base. To allow for cross-taxon and between-habitat comparisons, data were stratified to ecosystems where multiple taxonomic groups have been measured. We obtained 67 stream sites in Germany, where water quality, phyto-benthos, macrophytes, benthic invertebrates and fish have been measured. For the lakes, we obtained 59 sites extending over a broader spatial gradient from the UK to the Baltic countries, where water quality, phytoplankton, macrophytes and fish have been measured.

Nonmetric multidimensional scaling (NMDS) was carried out to determine the biological turnover, or gradient length, of the species data sets as well as to condense the species by site matrix into three indices of community composition; namely NMDS scores of the first, second and third ordination axes. Environmental gradients in streams and lakes were determined using Principal Component Analysis (PCA) on centred and standardized variables. The multivariate variables of community composition were related to environmental gradients, determined through PCA, by regression. Linear and parabola (second-order polynomial) regressions were estimated using the least-squares principle.

Three metrics were used to compare the response of the taxonomic groups to stress. Precision was calculated as the squared correlation (coefficient of determination, adjusted R^2) between the observed and fitted values as a measure of variance accounted for by the model. Sensitivity of a taxonomic assemblage to stress was estimated as the magnitude of change (slope) of a predicted stress–response relationship. If quadratic models were used, the slope was calculated as the absolute value of the sum of the slope (β) coefficients. The error of the predictive model expressed as the root mean square error was used to compare the robustness of the response relationships (Johnson & Hering 2009). Across taxonomic-group comparisons were done only if the fitted model was significant ($P < 0.05$).

Results

Taxonomic group response to the two environmental gradients revealed by PCA differed both among groups and between habitat types (streams vs lakes) (Table 1). In streams, invertebrates (0.407) and fish (0.47) had the highest precision (adj. R^2 values) for the nutrient gradient, followed by macrophytes (0.233) and phytobenthos (0.110). By comparison, relationships with the secondary gradient were less robust, with highest precision found for invertebrates (0.200), followed by macrophytes (0.17) and fish (0.168); phytobenthos was not

significantly related to the secondary gradient (Table 1). For lakes, macrophytes were more precisely related to the nutrient gradient (max. adj. R^2 value = 0.47) than phytoplankton (max. 0.334) and fish (max 0.189). Contrary to streams, precision was higher for the secondary gradient compared to the nutrient gradient; precision for macrophytes was 0.561, and amounted to 0.411 and 0.353 for phytoplankton and fish, respectively.

Changes in assemblage structure (NMDS scores) showed strong responses, indicated by slope values, to both environmental gradients (PC1 and PC2) in streams and lakes (Figs. 1 and 2). For streams, the best relationship (i.e., highest sensitivity) between the primary gradient (PC1) and assemblage structure was found for fish (NMDS 1, slope = 0.500), followed by invertebrates (NMDS 1, slope = 0.414), macrophytes (NMDS 1, slope = -0.345) and phytobenthos (NMDS 2, slope = -0.195) (Fig. 1). Changes in the composition of fish assemblages, as described by NMDS 2, was also significantly related to the primary gradient, but with lower precision and sensitivity (adj. R^2 = 0.08, slope = -0.129). None of the other NMDS dimensions of the other taxonomic groups were significantly related to PC 1 ($P > 0.05$). When assemblages were regressed only against total P, the responses were generally stronger compared with the PC 1 gradient (Fig. 1). Changes in assemblage composition for fish (NMDS 1), invertebrates (NMDS

	PCA 1			PCA 2		
	Adj. R^2	Slope	RMSE	Adj. R^2	Slope	RMSE
Streams						
Phytobenthos (n=1/0)	0.110	0.195	0.727	---	---	---
Macrophytes (n= 1/1)	0.233	0.345	0.727	0.17	0.052	0.802
Invertebrates (n=1/1)	0.407	0.414	0.706	0.200	0.463	0.746
Fish (n=2/1)	Min: 0.08 Max: 0.47	0.129 0.500	0.409 0.718	0.168	0.609	0.863
Lakes						
Phytoplankton (n=3/1)	Min: 0.079 Max: 0.334	0.736 1.173	0.404 0.412	0.411	0.856	0.587
Macrophytes (n=1)	0.460	0.963	0.527	0.561	1.325	0.381
Fish (n=3/2)	Min: 0.118 Max: 0.189	0.430 1.361	0.804 0.300	0.083 0.353	0.763 0.702	0.397 0.688

Table 1: Summary statistics of regression analysis of three variables of community structure (NMDS axes scores) of taxonomic groups in streams (n=67) and lakes (n=59) against two environmental gradients extracted using principal component analysis (PCA). The primary axis of both streams and lakes was interpreted as showing a gradient in nutrient enrichment and land use and the second PCA axis showing variability in land use alone. Adj. R^2 , coefficient of determination; RMSE, root mean square error of the prediction. Only absolute values of slopes, and the minima and maxima from significant regressions are shown; n indicates the number of significant regressions of each taxonomic group variable (i.e., NMDS dimensions 1 – 3) with PCA1/PCA2.

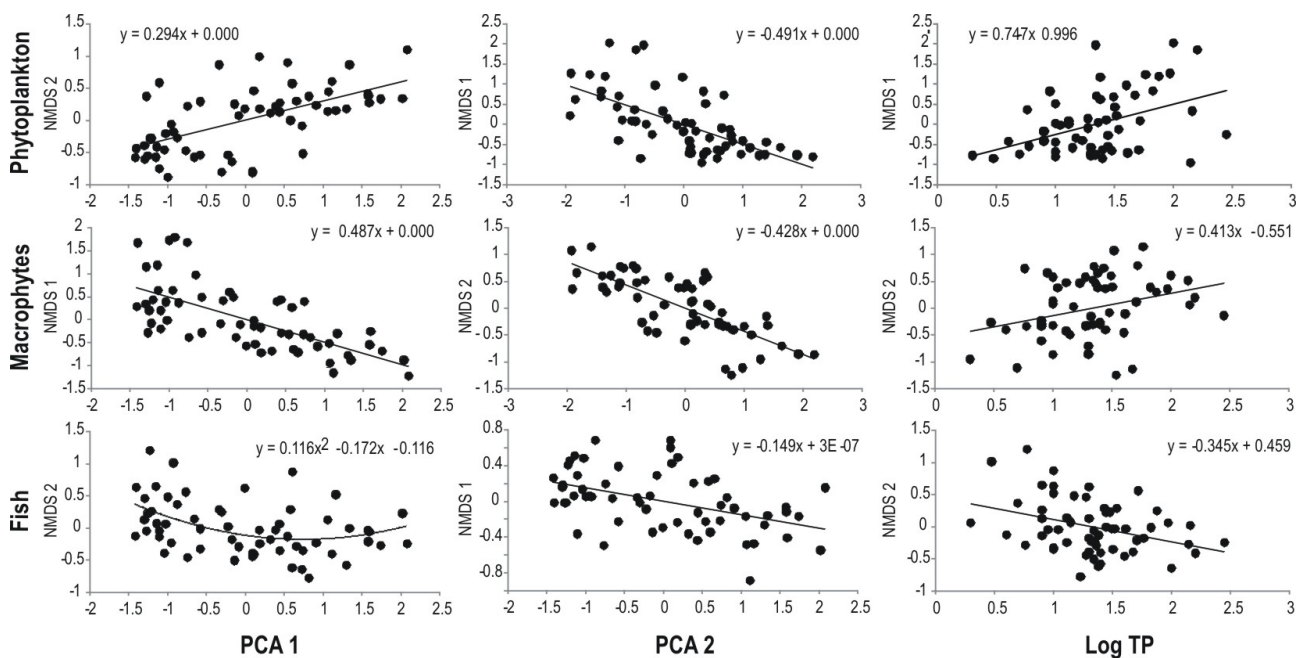
2), and macrophytes (NMDS 1) were also related to the secondary gradient (PC2); however, this relationship was weak and linear for macrophytes (slope = 0.052) but stronger and non-linear for invertebrates (absolute value of sum of slope coefficient = 0.463) and fish (absolute value of sum of slope coefficient = 0.609) (Fig. 1). Regarding measures of robustness of the regression relationships, expressed as RMSE, the stream assemblages usually showed values >0.7 when related to the primary and secondary environmental gradient (Tab. 1).

For lakes, the strongest response was noted for changes in macrophyte assemblages related to the nutrient gradient (NMDS 1, slope = -0.487), followed by phytoplankton (NMDS 2, slope 0.294), and fish (NMDS 2, absolute value of sum of slope coefficient = 0.288). Similar to streams, changes in assemblage composition were generally higher when assemblage variables were regressed against total P (Fig. 2). Contrary to the streams, responses were stronger for the secondary gradient; NMDS 1 for phytoplankton had a slope of -0.491, followed by macrophytes (slope = -0.428) and fish (slope = -0.149). For lakes, the measures of robustness (RMSE) were weaker than in the streams, and <0.527 and <0.688 for most assemblages when regressed against PC1 and PC2, respectively (Tab. 1).

Discussion

Anthropogenic alterations of many landscapes are leaving clear footprints on the structure and function of aquatic ecosystems. Lakes and stream habitats, in particular, have been directly and indirectly affected by many forms of anthropogenic stress. Our results support the importance of elevated nutrient levels and alterations in land use as the two main environmental gradients explaining variability in stream and lake condition; the first PC axis, interpreted as representing elevated nutrient concentrations, explained from 30% (streams) to 28% (lakes) of the variation, while the second PC axis, corresponding to land-use characteristics, explained another 17% and 21% of the variability among streams and lakes, respectively.

Overall, our results differ from those of other studies where, for example, variation in primary producer assemblages were better correlated to nutrient gradients than the primary (invertebrate) and secondary (fish) consumers (e.g., Johnson & Hering 2009). The different outcomes between studies highlights limitations when using a space-for-time approach because ecological responses to stress gradients may be context dependent and vary as a function of local and regional environmental settings, thereby challenging the design of monitoring programmes.



[Figure 1: Regression plots of assemblage composition of stream assemblages against PC 1, PC 2, and log total P. Shown are plots from regressions of NMDS dimensions with the highest model fit. Trend lines and equation details are also shown (n= 67).]

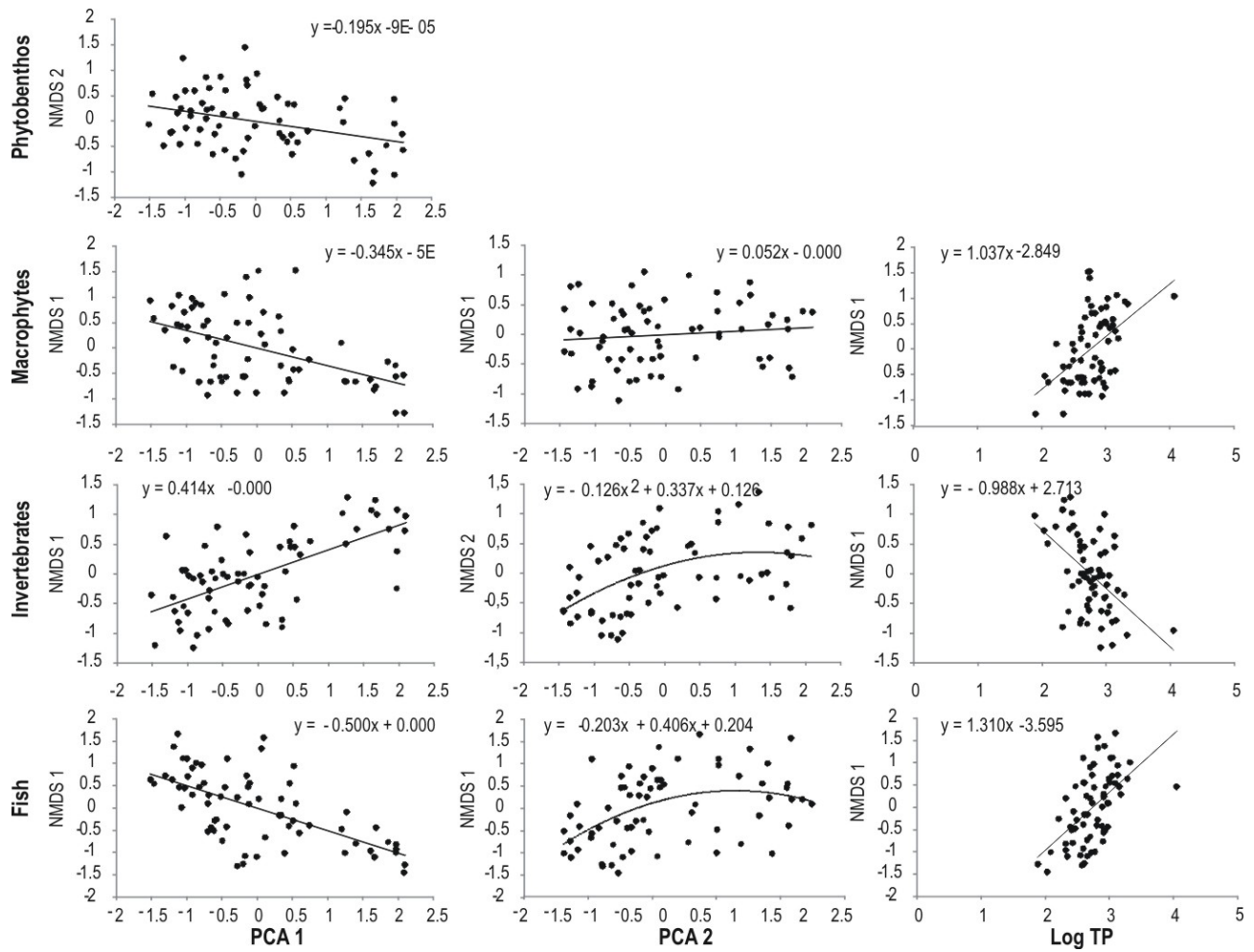


Figure 2: Regression plots of assemblage composition of lake assemblages against PC 1, PC 2, and log total P. Shown are plots from regressions of NMDS dimensions with the highest model fit. Trend lines and equation details are also shown ($n = 59$).

The different taxonomic groups also showed varied response to the two environmental gradients. For example, macrophytes showed linear responses to both stressor gradients in lakes and streams. Fish showed a linear response to the nutrient gradient and a non-linear relationship with the secondary gradient in streams, while the opposite relationships were found in lakes. A similar non-linear relationship was found for invertebrates to the secondary gradients in streams. These findings illustrate how selection of taxonomic group can be crucial for detecting change, and support earlier work focussed on single taxonomic groups like fish, macrophytes, invertebrates and benthic diatoms. These findings of asymmetric response clearly show how selection of different taxonomic groups can be used to strengthen inference in a multiple lines of evidence approach and to select complementary indicators.

Monitoring changes in biodiversity is costly and therefore much focus has been on finding surrogates for biodiversity (e.g. McGeoch 1998). Our study showed that many of the taxonomic groups were significantly correlated with one another in both ecosystem types, although the strength of correlation coefficients varied markedly between the taxonomic assemblages. These findings imply that several of the taxonomic groups may be used as surrogates or as proxies of wholesale aquatic biodiversity; that is, indicating change in diversity for other groups, and resulting in considerable economic savings. Judged from the strength of correlation between taxonomic groups in this study, we conclude that fish and macrophytes are the best candidates of biodiversity surrogates in streams and lakes, respectively.

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Assessing the impacts of land use on stream invertebrate assemblages: the importance of habitat and season.

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Abstract

Previous studies in streams have well established the connection between catchment land use and riparian alterations to in-stream physicochemical degradation and to resulting changes in the composition of organisms such as benthic macroinvertebrates. However, most have focused on a single habitat type (e.g. riffles) and it is still largely unknown how organisms, such as benthic macroinvertebrates, at the mesohabitat scale within a reach are differently influenced by land use within the catchment. This study focused on pool and riffle habitats as they are expected to differ in physicochemical characteristics, basal food availability, and the associated benthic macroinvertebrate communities. More specifically this study addressed whether sampling from riffle habitats alone, or in conjunction with pool habitats, improve our ability to detect and distinguish the effects of catchment land use, reach-wide riparian vegetation, and elevated nutrient concentrations on benthic macroinvertebrate assemblages. Combinations of taxa metrics/indices were used to test differences in pool and riffle sensitivity to detection of anthropogenic disturbance.

Seasonal samples of benthic macroinvertebrates and environmental variables were collected from nine small to medium (orders 2-4), lowland (altitude 11 to 191 m.a.s.l.) boreal streams in south-central Sweden. The study streams were chosen to represent a primary agricultural land use gradient, including alterations in riparian habitat, stream habitat and canopy cover and elevated nutrient concentrations. The streams were circumneutral (pH 6.3 to 8.3) and ranged from nutrient poor (18 µg NO₂-NO₃/L and 9 µg TP/L) to nutrient rich (1824 µg NO₂-NO₃/L and 198 µg TP/L). Catchment area of the study streams varied from 9 to 156 km². Catchments of four of the streams were predominantly forested (73-98% forested), with very little influence of agriculture (0-0.7% arable land/pastures), while the other five consisted of landscapes more influenced by agricultural activity (8.4-43% arable land/pastures),

with relatively less forested area (36-61%). Forest type also differed between the two stream groups; the four forest-dominated streams had catchments consisting almost entirely of coniferous trees, while forests of the five agricultural streams consisted of mixed coniferous/deciduous forests. Width of undisturbed riparian vegetation along the sampling reach differed between the two groups; the four forest-dominated streams had at least 100 m of undisturbed vegetation, while average undisturbed widths of the five agricultural streams ranged from 6 to 55 m.

Results

Environmental gradients

Principal component analysis (PCA) was used to define the major environmental gradients. The first PC axis was related to variables indicative of nutrient enrichment and land use, the second PC axis was related to habitat variables (Fig. 1).

Streams with forested and agriculture catchments clearly separated along the first PC axis (nutrient pollution-land use gradient). Placement along the second PC axis showed differences in stream size and in-stream habitat, e.g. positive orientation of larger streams with a higher percentage of open canopy and larger dimensions of in-stream habitats.

Comparison of within stream habitats revealed much larger and more predictable differences in agricultural streams, while habitats of forested streams were less variable. Relative orientation of pool and riffle habitats along the second PCA axis from agriculture streams were reversed in three forested streams. Pool and riffle habitats in forested streams were more similar in substrate size, percentage of mosses and liverworts, and woody debris. Thus, habitat depth and width may be the stronger influential characteristics of habitat differences in forested streams.

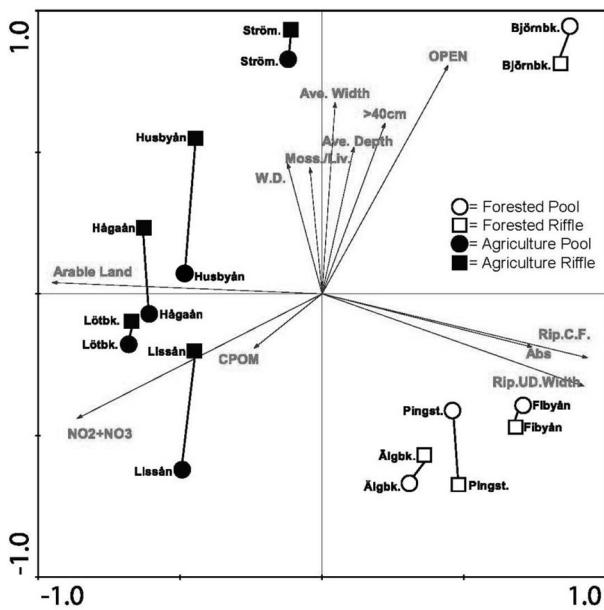


Figure 1. Principal component analysis (PCA) of the 12 environmental variables and pool and riffle habitats of the nine study streams. Percent arable land (Arable Land), nitrite and nitrate (NO_2+NO_3), undisturbed riparian width (Rip. UD. Width), riparian coniferous forest (Rip. C.F.), water color (Abs), percent coarse particulate organic matter (CPOM), open stream canopy (OPEN), average habitat width (Ave. Width), percent substrate greater than 40 cm (>40cm), average habitat depth (Ave. Depth), mosses and liverworts (Moss/Liv.), percent woody debris (W.D.).

Macroinvertebrate response

ANOSIM showed significant differences in assemblage composition between agriculture and forested streams when samples were combined or when riffle and pool samples were analyzed separately. Likewise, comparisons of riffle and pool habitats across all streams also showed significant differences in assemblage composition.

SIMPER analysis identified taxa associated with pool and riffle habitats (Table 1). In general, agricultural streams had higher overall mean abundances compared to forested streams for most of the top ranked 12 taxa for both pool and riffles habitats. Although several taxa occurred in both pool and riffle habitats, four taxa were unique to pools and another four taxa were unique to riffle habitats. Of the taxa found in both habitats, mean abundances were not indicative of habitat type, with the exception of Simuliidae, where mean abundance in riffles were higher than pools for both agriculture and forested streams.

Pool habitats showed large differences between forested and agricultural streams in mean abundances of four taxa; *Pisidium* in agriculture stream pools was more than 3x the mean abundance in forested streams, and mean abundance of three midges (Orthoclaadiinae, Chironomini, Ceratopogonidae) were 2x as high in agricultural streams compared to forested streams. Differences were even more pronounced in riffle habitats where two taxa (*Pisidium*, *Hydropsyche siltalai*) were more than 6x greater, and two species (*Elmisaenea*, *Gammarus pulex*) were more than 2x greater in agricultural streams compared to forested streams. Riffles also contained one species (*Nemoura cinerea*) that showed the opposite trend, with mean abundances greater than four times in forested streams compared to agricultural streams.

Redundancy analysis (RDA) was used to determine what environmental variables characterized macroinvertebrate assemblages of riffle and pool habitats (Table 1, Fig. 2). Five environmental variables were retained in the RDA model using forward selection (Fig. 2). The first axis explained 23.6% of the total variation of macroinvertebrate community structure and was characterized by mosses/liverworts and large substrates. The second axis accounted for an additional 5% of the

Ave. diss. = 57.25	Pools (mean abundance)		Ave. diss. = 55.05	Riffles (mean abundance)	
TAXON	Agriculture	Forested	TAXON	Agriculture	Forested
Orthoclaadiinae	40.6	21.6	Simuliidae	65.9	42.6
<i>Pisidium</i> sp.	33.9	9.48	Orthoclaadiinae	65.3	42.1
Tanytarsini	28.7	21.1	Oligochaeta	54.4	35
Oligochaeta	33.2	19.4	<i>Pisidium</i> sp.	32.5	3.94
<i>Tanypodinae</i>	28	23	Chironomini	29.8	20.1
Chironomini	21.7	9.32	<i>Elmisaenea</i> *	33.9	14.8
<i>Asellus aquaticus</i> *	14.7	14	<i>Gammarus pulex</i>	28.9	10
Ceratopogonidae*	22.3	11.1	Tanytarsini	30.1	26.6
<i>Gammarus pulex</i>	12.1	6.32	<i>Limnius volckmari</i> *	19.4	15.4
Simuliidae	8.13	2.47	<i>Nemoura cinerea</i> *	3.42	15.1
Limnephilidae*	4.4	5.01	<i>Hydropsyche siltalai</i> *	17.5	2.76
<i>Ephemera vulgate</i> *	3.58	3.16	<i>Tanypodinae</i>	20.6	17.4

Table 1. Summary of top 12 SIMPER taxa between habitat (pools and riffles) and stream type (agriculture and forest). *Indicates taxa which occur only in pool or riffle habitats.

variation. Taxa were positively related to mosses and liverworts and large substrates in riffle habitats. Taxa associated with pool habitats were related to low water color and reduced riparian vegetation.

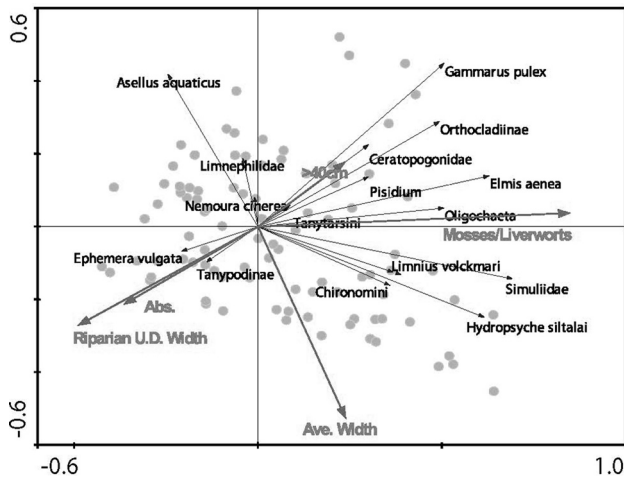


Figure 2. Redundancy analysis (RDA) of riffle and pool assemblages in nine streams. Only 12 indicator taxa (SIMPER) are shown for clarity. Significant environmental variables ($p < 0.05$) were mosses/liverworts, riparian undisturbed width (Riparian U.D. Width), average width of habitat (Ave. Width), large substrate ($>40\text{cm}$), water color (Abs.).

Macroinvertebrate-environment relationships

Analysis of five metrics of diversity showed that only number of individuals differed between riffle and pool assemblages (Table 2). Mean number of individuals was significantly higher in riffle compared to pool habitats in both autumn ($p = 0.038$) and spring ($p = 0.0003$). Riffle habitats in spring had higher proportion of % EPT taxa (0.327) compared to pools (0.204), although difference were not statistically significant ($p = 0.058$).

Four measures of assemblage composition turnover (CA axis 1-2 scores for presence/absence (P/A) and abundance (ABS)) and the five diversity metrics for riffle and pool habitats in spring and autumn were regressed against the nutrient gradient (PC1). Two metrics were used to compare the response of the macroinvertebrate assemblages to the PC nutrient enrichment-land use gradient: precision, calculated as the squared correlation (coefficient of determination, adjusted R^2) and sensitivity, estimated as the magnitude of change (slope) (Table 3, Fig. 3).

Neither precision (R^2) nor sensitivity (slope) of the eight metrics differed between riffle and pool habitats (Fig. 3). However, a few strong relationships were found when comparing single metrics (Table 3).

	Autumn			Spring		
	Pool	Riffle	(p)	Pool	Riffle	(p)
Number of individuals	341±161	544±215	0.038	218±36	496±126	0.0003
Taxon richness	25.7±9.1	29.1±7.14	0.4001	22.1±5.86	26.3±6.87	0.1307
Shannon diversity	2.79±0.389	2.97±0.384	0.3099	2.66±0.28	2.83±0.37	0.1451
Evenness	0.685±0.096	0.692±0.065	0.4529	0.670±0.040	0.669±0.667	0.9648
EPT (%)	0.263±0.159	0.258±0.114	0.1451	0.204±0.105	0.327±0.125	0.0576

Table 2. Mean \pm SD of diversity metrics of macroinvertebrate assemblages from riffle and pool habitats in nine streams sampled in autumn and spring. Mann-Whitney test and associated significance level (p).

Metric	Autumn Pool			Autumn Riffle			Spring Pool			Spring Riffle		
	R^2	Slope	(p)	R^2	Slope	(p)	R^2	Slope	(p)	R^2	Slope	(p)
CA1 (P/A)	0.149	0.144	NS	0.138	0.025	NS	0.521	0.222	0.017	0.458	0.210	0.026
CA2 (P/A)	0.074	0.113	NS	0.435	0.238	0.031	0.141	0.008	NS	0.139	0.014	NS
CA1 (ABS)	0.018	0.084	NS	0.070	0.086	NS	0.265	0.144	NS	0.054	0.081	NS
CA2 (ABS)	0.203	0.144	NS	0.320	0.183	NS	0.140	0.010	NS	0.231	0.146	NS
Number of individuals	0.289	43.10	NS	0.203	58.70	NS	0.071	3.879	NS	0.359	41.33	NS
Taxon richness	0.028	1.250	NS	0.142	0.022	NS	0.071	0.634	NS	0.106	0.608	NS
Shannon diversity	0.092	0.035	NS	0.116	0.028	NS	0.010	0.044	NS	0.087	0.040	NS
Evenness	0.219	0.023	NS	0.096	0.006	NS	0.076	0.004	NS	0.140	0.001	NS
% EPT	0.870	0.015	NS	0.100	0.026	NS	0.003	0.016	NS	0.446	0.044	0.029

Table 3. Summary statistics of regression analysis of diversity and community metrics (axis 1-2 CA scores) of macroinvertebrate assemblages from riffle and pool habitats in nine streams sampled in autumn and spring regressed against the first axis of a PCA (nutrient enrichment-land use gradient).

The strongest relationship was found for CA1-P/A in spring for both pool ($p = 0.017$, $R^2 = 0.521$, slope = 0.222) and riffle ($p = 0.027$, $R^2 = 0.459$, slope = 0.210) habitats. There were no significant relationships between any of the four autumn pool CA's and the nutrient gradient, however CA2-Abundance had the strongest relationship ($R^2 = 0.204$, slope = 0.144), followed closely by CA1-P/A with only precision being lower ($R^2 = 0.150$, slope = 0.144). Autumn riffle CA2-P/A had a significant relationship ($p = 0.032$, $R^2 = 0.436$, slope = 0.238), while CA2-Abundance had the next strongest response ($R^2 = 0.321$, slope = 0.183), followed by CA1-P/A ($R^2 = 0.138$, slope = 0.0252). CA1-Abundance had the weakest response in all habitats/seasons except for spring pool habitats, where both CA2's had the weakest response.

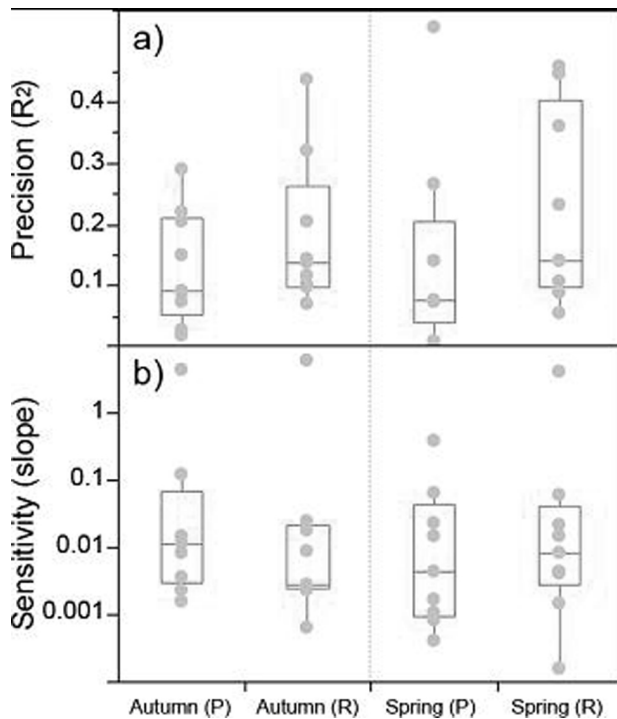


Figure 3. Box plots (median, 25th, 75th percentile and range) of precision and sensitivity of eight community metrics of macroinvertebrate assemblages from nine streams sampled spring and autumn pool (P) and riffle (R) habitats.

The remaining community metrics resulted in only one significant relationship; spring riffle habitat %EPT Taxa ($p = 0.029$, $R^2 = 0.446$, slope = 0.0446). Number of individuals had the next strongest response in spring riffle habitats ($R^2 = 0.360$, slope = 41.34), and had the strongest response across all other seasons and habitats; autumn pool ($R^2 = 0.289$, slope = 43.1), autumn riffle ($R^2 = 0.204$, slope = 58.7), spring pool ($R^2 = 0.072$, slope = 3.88). Conversely, evenness had the least response across all seasons and habitats except for percent EPT taxa during autumn from pool habitats.

Conclusion

Benthic macroinvertebrate assemblages of pool and riffle habitats differed between streams with agricultural and forested catchments. Macroinvertebrate assemblages from riffle habitats showed overall greater response to the nutrient gradient, with spring sampling showing the strongest relationship, followed by pool assemblages in autumn, while the poorest relationships were found for spring pool habitats. Both pools and riffles in spring and autumn showed high sensitivity of number of individuals to the nutrient gradient, however precision was relatively low for spring pools. These results may be a reflection of a higher amount and quality of basal food resources in riffle habitats compared to pools, as well as increased sensitivity to stress associated with pools, especially within the agricultural streams (e.g., sediment deposition, lower oxygen).

If sampling is restricted to a single habitat, riffles may be better for detecting to the effects of elevated nutrient concentrations and land use, especially when sampling is confined to a single season. However, mesohabitat sampling could be warranted when measuring the amount of difference between habitats within streams in an ecological context (e.g., differences in basal food resources), or when comparisons need to be made between streams that do not share all habitat types (e.g., pool-riffle, run-riffle, run-pool).