Journal of Freshwater Ecology
Publication details, including instructions for authors and subscription information:
http://www.tandfonline.com/loi/tjfe20

Using Stream Macroinvertebrates to Compare Riparian Land Use Practices on Cattle Farms in Southwestern Wisconsin

Brian M. Weigel [a], John Lyons [b], Laura K. Paine [c], Stanley I. Dodson [d] & Daniel J. Undersander [e]

[a] Department of Zoology, University of Wisconsin, Madison, WI, 53706, USA

[b] Bureau of Integrated Science Services Wisconsin Department of Natural Resources, Monona, WI, 53716, USA

[c] Columbia County Agriculture Center, Portage, WI, 53901, USA

[d] Department of Zoology, University of Wisconsin, Madison, WI, 53706, USA

[e] Department of Agronomy, University of Wisconsin, Madison, WI, 53706, USA

Published online: 06 Jan 2011.

Using Stream Macroinvertebrates to Compare Riparian Land Use Practices on Cattle Farms in Southwestern Wisconsin

Brian M. Weigel
Department of Zoology, University of Wisconsin
Madison, WI 53706 USA
John Lyons
Bureau of Integrated Science Services
Wisconsin Department of Natural Resources
Monona, WI 53716 USA
Laura K. Paine
Columbia County Agriculture Center
Portage, WI 53901 USA
Stanley I. Dodson
Department of Zoology, University of Wisconsin
Madison, WI 53706 USA
and
Daniel J. Undersander
Department of Agronomy, University of Wisconsin
Madison, WI 53706 USA

Abstract

Vegetative riparian buffer strips are typically used to curb stream degradation due to cattle grazing, but intensive rotational grazing has shown promise as an alternative best management practice. We compared aquatic macroinvertebrate assemblages among stream segments within continuously grazed pastures, intensive rotationally grazed pastures, undisturbed grassy vegetative buffer strips, and undisturbed woody vegetative buffer strips. We collected macroinvertebrate and stream sedimentation data from four streams in each land use category in two consecutive years. In an attempt to account for inherent watershed variability among streams, we represented watershed condition with a sample collected upstream of each treatment reach. Watershed condition tended to have greater influence on macroinvertebrate measures than local riparian land use. However, local riparian land use influences were apparent if watershed condition was statistically accounted for with analysis of covariance. Stream reaches with intensive rotational grazing tended to have macroinvertebrate assemblage characteristics intermediate of the buffer and continuously grazed reaches. Although we detected some differences in macroinvertebrate assemblages that apparently reflected very local land use, our results suggest the macroinvertebrates were mostly responding to large-scale watershed influences.

INTRODUCTION

Intensive agriculture on or near riparian land can degrade stream water quality (Karr and Schlosser 1978, Osborne and Wiley 1988) and instream habitat (Wohl and Carline 1996, Wang et al. 1997). Clearing the steep hillsides in the Driftless Area Ecoregion (Omernik and Gallant 1988) of southwestern Wisconsin for agriculture has increased surface runoff, sediment yield (Graczyk and Sonzogni 1991), and flood magnitude and frequency (Knox 1977, Potter 1991). Non-point source pollution from agriculture is a serious problem that affects a greater length of streams in Wisconsin than any other type of environmental degradation (WDNR 1994). Streams in southwestern Wisconsin and in adjacent areas of Illinois, Iowa, and Minnesota have suffered tremendous environmental damage from livestock, yet they have great potential for biotic integrity and ecosystem health with improved land use (Thorn et al. 1997, Trimble 1997).

Continuous grazing, the conventional method of cattle grazing in the Midwest (Fortenbery and Saupe 1995), causes serious stream and riparian degradation (Kauffman and Krueger 1984, Wohl and Carline 1996). Trimble and Mendel (1995) found that heavy grazing compacts soil, reduces infiltration, and increases runoff, bank erosion, and sediment yield to streams. Cattle that graze in riparian areas erode stream banks directly by trampling the banks and indirectly by removing bank vegetation. Overgrazing causes devegetated reaches to become wider and shallower (Trimble 1994).

Vegetative buffer strips are prescribed for riparian areas as a best management practice (BMP) to curb degradation caused by continuous grazing (Karr and Schlosser 1978, Osborne and Kovacic 1993, Barling and Moore 1994). Some buffer strips are installed just by fencing an area to exclude livestock along the stream corridor. These buffer strips usually originate in grassy vegetation, but over time, change to include woody shrubs and eventually trees. Vegetative buffer strips effectively trap sediment and nutrients before they enter the stream (Barling and Moore 1994, Owens et al. 1996), but filtering qualities differ between grassy and woody vegetative buffers (Osborne and Kovacic 1993), and these differences may be reflected in the stream biota.

Intensive rotational grazing (IRG) has been proposed as an alternative BMP to protect streams and riparian areas (Undersander et al. 1993). When IRG is practiced, the farmer converts row-crops into pasture and divides the total pasture area into smaller pastures called "paddocks". Some paddocks allow cattle access to the stream for watering. The entire herd grazes the stream paddock for a short period (12 hours to three days) before being rotated to the next paddock. Paddocks and stream banks are undisturbed for three - five weeks between grazing episodes and therefore have a chance to recuperate before being grazed again.

Changes in overland runoff characteristics and a reduction in fertilizer inputs should follow conversion from continuous grazing to IRG since this conversion requires transforming row-crop fields to pastures. Runoff and fertilizer changes may be reflected in dissolved oxygen, nutrient values, and other water quality parameters in streams. We expected water quality differences among the land use treatment reaches. However, accurate and precise measurement of such differences is extremely difficult for the number of sites we considered because most differences are only detectable during runoff events, which require continuous and expensive monitoring equipment (Graczyk and Sonzogni 1991).

The goal of this study was to compare the effects of four riparian land use treatments - continuous grazing, IRG, grassy buffers, and woody buffers - on macroinvertebrate assemblages. We used stream macroinvertebrates as indicators of water quality changes instead of trying to measure water quality parameters directly. We chose to investigate aquatic macroinvertebrate assemblages, which primarily consist of aquatic insects, to indicate water quality changes because they are integrators of aquatic and terrestrial influences (Rosenberg and Resh 1993, Karr and Chu 1997). Macroinvertebrate assemblages can be characterized in a number of ways (e.g., Lenat 1988, Plafkin et al. 1989). Attributes and indexes calculated from the macroinvertebrate assemblage will be termed "metrics" throughout this paper.

We calculated 27 macroinvertebrate metrics in three categories: organic pollution tolerance, taxa richness, and feeding ecology. High total taxa richness and Ephemeroptera, Plecoptera, Trichoptera (EPT) richness are generally indicative of optimal water and environmental quality (Lenat 1984, Lenat and Penrose 1996). We expected undisturbed stream reaches (i.e., grassy and woody buffer sites) to rank best in all metrics. Intensive rotational grazing has intermittent cattle disturbance in riparian areas. Therefore, IRG stream reaches should have habitat and metric rankings higher than reaches with continuously grazed pastures but perhaps not as high as scores from undisturbed buffer reaches.

Based on previous studies, we predicted that continuous grazing in the riparian area would be associated with the most erodable stream banks and coarse substrate embeddedness. Bank erodability estimates suggest where the sediment is coming from, and the embeddedness estimates tell us if the sediments are being held in the coarse substrate of the treatment reach. In turn, we predicted a negative response of the macroinvertebrate assemblage to the sedimentation in continuously grazed reaches.

METHODS AND MATERIALS

Study area. We selected streams in one ecoregion, the Driftless Area (Omernik and Gallant 1988), in an attempt to reduce natural variability among streams. We expected some inherent stream variability to remain. Vertical geographic relief is as much as 110 m/km in this area. Sandstone and limestone bluffs with forested ridges are characteristic. Valley bottom streams receive important spring flow from the bluffs and the valley floor is usually dominated by agriculture. All streams selected for study were wadeable (<1 m deep), 2nd - 4th order, and averaged 3.7 m wide. These coldwater streams (<22°C maximum summer daily mean; Lyons et al. 1996) had an average gradient of 4.0m/km and discharge of 0.128 m^3/s.

We selected streams that had been managed according to one of the four land use categories (IRG, continuous grazing, grassy buffer strip, and woody buffer strip) for at least three years prior to sampling to avoid problems with sites being in a transitional state from one land use to another. Grassy buffer strips mostly consisted of a thick turf of reed canarygrass (*Phalaris arundinacea*) with an occasional shrub or small tree. Woody buffer strips consisted of mostly medium-sized box elders (*Acer negundo*) and various popples (*Populus* spp.) and willows (*Salix* spp.). The understory was sparsely vegetated with shade-tolerant flora. Minimum buffer width on each side of the stream was 10 m measured perpendicular from the top of the first bank.

Macroinvertebrate and habitat sampling. We designated two sequential sampling reaches per stream. The treatment reach was downstream and was within one of the four land use treatment categories. Upstream was a second reach, which was used as the covariate in our data analysis to account for inherent watershed differences among streams. We sampled four streams in each land use treatment. Samples were collected in 1996 and 1997 from the same reaches between 21 March and 24 April. We designated the sampling reach lengths according to stream channel morphology (reach length range: 101 - 272 m) based on the Simonson et al. (1994)

transect method that we used to assess bank erodability and coarse substrate embeddedness. We estimated bank erodability by measuring the length of bare bank within 1 m of the wetted portion of the stream. We visually estimated the percent embeddedness of gravel and cobble substrates at four points within each transect.

We collected macroinvertebrate samples from a riffle at the downstream end of each station. The Hilsenhoff (1987) riffle kick-net sampling and grid-pan subsorting procedures were followed with slight modifications. We collected five kick-net (25.4 X 45.7 cm frame and 800 X 600 micron mesh) samples per riffle station. Each sample was preserved separately in 75% ethanol. A minimum of 125 individuals were subsorted to increase the likelihood of having at least 100 identifiable animals needed for pollution tolerance metric calculations. Individuals in the subsample were identified to species level if possible. Chironomid identification necessitated slide mounting for most species.

Metric calculations. We entered the taxonomic data in the Wisconsin Department of Natural Resources (WDNR) "Bugprogram" (Szczytko 1996) for 19 metric calculations, and we calculated eight additional taxa richness metrics (Table 1). We analyzed each of the five samples per riffle site separately, calculating a score for each metric from each sample. We then averaged the five scores from each riffle to represent the site score.

The three organic pollution biotic indexes calculated were Hilsenhoff's Biotic Index (HBI; Hilsenhoff 1987), family-level biotic index (FBI; Hilsenhoff 1988), and mean pollution tolerance value (MPTV; Lillie and Schlesser 1994). These organic pollution metrics are based on empirical evidence showing certain macroinvertebrate species tolerate organic enrichment better than others. Taxa are assigned a pollution tolerance value from 0 - 10, with 10 being the most pollution tolerant and 0 being the most sensitive.

We calculated 14 taxa richness and abundance metrics. Among these metrics were species richness, generic richness, and four variations of the Ephemeroptera-Plecoptera-Trichoptera (EPT) metric, including EPT abundance (number of individuals), percent EPT abundance, EPT generic richness, and percent EPT generic richness. Plecopteran and dipteran metrics were calculated as abundances, percent abundances, generic richness, and percent generic richness.

Feeding ecology metrics categorize taxa based on morphological feeding mechanisms, and include shredders, scrapers, and collectors. The collector category is subdivided into filterers and gatherers (Merritt and Cummins 1984). We calculated percent abundance and percent generic richness for each of these five categories.

Statistical analysis. We employed an analysis we believe is novel to stream biomonitoring, analysis of covariance (ANCOVA; Draper and Smith 1981), in an attempt to statistically account for inherent watershed variability among sites. Macroinvertebrates sampled from the treatment riffle was our estimate of the total influence of local riparian land use treatment and the upstream watershed influence on the macroinvertebrates. Macroinvertebrates sampled from the upstream riffle were used as an estimate of the influence of only the upstream watershed condition on the macroinvertebrates. We ran an ANCOVA model for each metric (e.g., species richness) from the treatment station as the dependent variable, riparian land use treatment as the main effect variable, year (1996 or 1997) as the blocking variable, and the matching metric (e.g., species richness) at the upstream station as the covariate. If the land use treatment term was significant in the ANCOVA model, we interpreted changes between the upstream and downstream site as a response to the riparian land use treatment.

Metric values were transformed to approximate normality or to improve variance homogeneity accordingly before analysis. We began with the most complex ANCOVA model:

$$y = C + B + L + (C*B) + (C*L) + (Y*L) + (C*B*L)$$

where y = metric at downstream site, C = covariate or metric at upstream site, B = year blocking variable, L = main effect or land use treatment, followed by all possible interactions. To ease the interpretation, we simplified the models by pooling non-significant ($p > 0.05$) terms into the error term. We began simplification by removing the three-way interaction term if it was not significant and refit the model. Then we removed the non-significant two-way interaction terms and refit the model. Non-significant blocking and covariate terms were removed unless they were part of a significant two-way or three-way interaction term. The ANCOVA model generated separate parameter estimates (i.e., y-intercept and slope) for each land use treatment category. Pairwise comparisons were made, corrected for multiple comparisons with the Tukey - Kramer adjustment (SAS 1990), to check for significant differences among land use treatments for these parameter estimates. Differences were considered significant if $p \leq 0.05$. The analysis became analysis of variance (ANOVA) if the covariate or covariate interaction terms were not significant, and subsequent pairwise comparisons were also adjusted with Tukey - Kramer procedures.

Once we had developed our final model for each metric, we used the least squares mean (lsmeans) statement (SAS 1990) to calculate a standardized mean metric score per land use treatment. When the covariate term was significant, the least squares mean statement determined the average covariate score for a particular metric across all land use treatments and then, using the final model, it estimated a mean metric value for each of the four land use categories. This approach allowed us to compare the relative influence of the land use treatments when watershed condition (i.e., the covariate) was held constant among all reaches. Habitat variables were analyzed with the same procedures used for the macroinvertebrate metrics. All statistical analyses were performed using SAS software (version 6, SAS Institute Inc., Cary, North Carolina).

RESULTS

Taxonomic composition was similar among sites within a land use treatment category, but the number of individuals per taxon often differed greatly among these sites. We identified 114 total taxa from all samples inclusive over both years. Among these were 7 coleopteran, 9 ephemeropteran, 8 plecopteran, 21 trichopteran, and 63 dipteran taxa including 38 chironimid taxa. Five taxa were ubiquitous: a hydropsychid caddis fly (*Ceratopsyche slossonae*), riffle beetle (*Optioservus fastiditus*), sideswimmer (*Gammarus pseudolimnaeus*), swimming mayfly (*Baetis* *tricaudatus*), and midge (*Diamesa* sp.). Any one of the above taxa dominated the macroinvertebrate assemblage at certain sites.

There were no significant year-to-year fluctuations in the final models for metrics in which the treatment term was significant. Only two metrics had a significant year term - percent filterers by abundance and percent shredders by genera.

Upstream watershed influence. Upstream watershed condition usually influenced the macroinvertebrate metrics much more than the local riparian land use treatment. The covariate term was highly significant in 22 of 27 metric models (Table 1), and accounted for 61 - 98% of the total variance explained (R^2) in models of which the covariate term was significant (Figure 1). Percent plecopteran genera, and percent abundance of scrapers and collectors were metrics in which the treatment term was significant but the covariate was not significant. The models for these metrics were reduced to ANOVA, and therefore the covariate did not account for any of the explained variation.

Table 1. Macroinvertebrate metrics with *p*-values for the covariate and treatment terms in their final models. MPTV = mean pollution tolerance value, HBI = Hilsenhoff's biotic index, FBI = family-level biotic, EPT = Ephemeroptera - Plecoptera - Trichoptera orders.

	p-values			*p*-values	
Metric	Covariate	Treatment	Metric	Covariate	Treatment
Organic pollution			Taxa richness		
MPTV*	0.0001	0.0561	Species richness*	0.0001	0.0272
HBI	0.0001	0.2257	Generic richness*	0.0001	0.0443
FBI	0.0001	0.5709	EPT genera[d]	0.0029	0.0162
			EPT abundance	0.0001	0.9243
Feeding ecology (by abundance)			EPT % genera	0.0001	0.1052
% shredder*	0.0001	0.0330	EPT % abundance	0.0001	0.9030
% scraper*	ns	0.0395	Plecoptera genera	0.0193	0.1307
% filterer[a]	0.0001	0.2089	Plecoptera abundance	0.0061	0.7046
% gatherer	0.0001	0.3335	Plecoptera % genera*	ns	0.0005
% collector*	ns	0.0129	Plecoptera % abundance	ns	0.0894
			Diptera genera*	0.0001	0.0066
Feeding ecology (by genera)			Diptera abundance	0.0001	0.2919
% shredder[b]	0.0010	0.7942	Diptera % genera[c]	0.0001	0.0183
% scraper	ns	0.1192	Diptera % abundance	0.0001	0.2793
% filterer[c]	0.0001	0.0120			
% gatherer	0.0014	0.4983			
% collector*	0.0002	0.0309			

* estimated mean metric scores were calculated and included in Table 2
[a] significant year effect ($p = 0.0042$)
[b] significant year effect ($p = 0.0046$) and covariate X year interaction ($p = 0.0048$)
[c] treatments not significantly different after adjustments for multiple comparisons
[d] significant covariate X year X treatment interaction ($p = 0.0285$)
[e] significant covariate X treatment interaction ($p = 0.0045$)

Physical stream habitat. The land use treatment term was the only significant factor ($p = 0.0137$) in the percent embeddedness of coarse substrate final model. Comparisons among treatments (Figure 2) showed continuously grazed reaches had significantly greater embeddedness of coarse substrate than IRG reaches ($p = 0.0131$) and marginally more than grassy buffer reaches ($p = 0.0597$).

The land use treatment term was the only significant factor ($p = 0.0108$) in the bank erodability final model. Continuously grazed reaches had significantly higher bank erodability than grassy buffer reaches ($p = 0.0060$), whereas IRG and woody buffer reaches were intermediate and not statistically different (Figure 2).

Organic pollution. Once upstream watershed conditions had been accounted for, some differences in macroinvertebrate assemblages were evident among land use treatments (Table 2). The covariate term was highly significant and the land use treatment term was marginally significant in the mean pollution tolerance value (MPTV) final model. Woody buffer reaches had a macroinvertebrate assemblage significantly ($p = 0.0369$) less tolerant of organic pollution than reaches with continuously grazed reaches, whereas grassy buffer and IRG treatment assemblages were intermediate and not significantly different from other treatments. Scores for all land use treatments were within the "very good" (3.51 - 4.50) qualitative rating with continuous grazing approaching only "good" (4.51 - 5.50) (Hilsenhoff 1987). The land use term was not significant in either the HBI or FBI models.

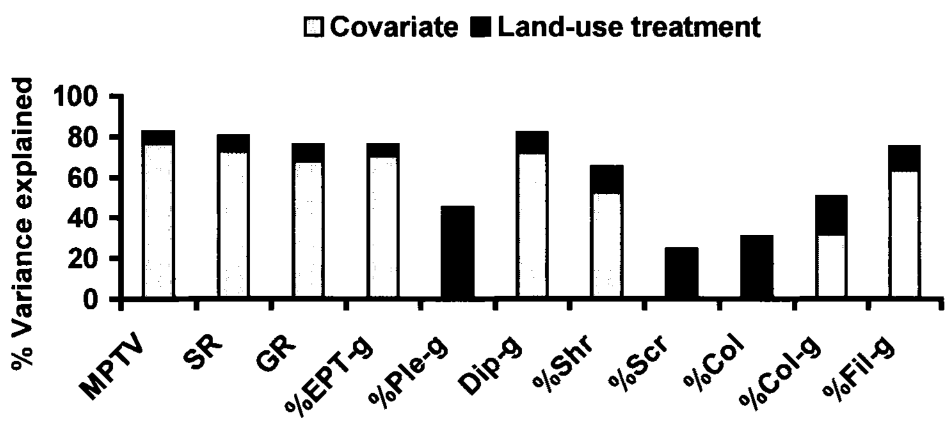

Figure 1. Percent variance explained (R^2) by the covariate and by the land use treatment terms in the final model calculated for 11 selected macroinvertebrate metrics. The covariate is represented if it explains a significant amount of the variation in the model ($p \leq 0.05$). The p-value for the land use treatment follows each metric acronym. Mean pollution tolerance value (MPTV; $p = 0.0561$), species richness (SR; $p = 0.0272$), generic richness (GR; $p = 0.0443$), percent EPT (Ephemeroptera-Plecoptera-Trichoptera) genera (%EPT-g; $p = 0.1052$), percent plecopteran genera (%Ple-g; $p = 0.0005$), dipteran genera (Dip-g; $p = 0.0066$), percent shredder richness (%Shr; $p = 0.0330$), percent scraper richness (%Scr; $p = 0.0395$), percent collector richness (%Col; $p = 0.0129$), percent collector genera (%Col-g; $p = 0.0309$), and percent filterer genera (%Fil-g; $p = 0.0120$).

Taxa richness. Both species richness and generic richness metrics had significant land use terms ($p = 0.0272$ and $p = 0.0443$, respectively) and highly significant covariate terms. Grassy buffer reaches had significantly lower species richness ($p = 0.0239$) and generic richness ($p = 0.0441$) than continuously grazed reaches, whereas IRG and woody buffer reaches were not significantly different from other treatments.

The covariate term was not significant in the percent of plecopteran genera final model, whereas the land use term was significant ($p = 0.0005$). The IRG reaches averaged a significantly higher percent plecopteran genera than continuously grazed ($p = 0.0042$) and grassy buffer ($p = 0.0076$) reaches. Woody buffer reaches also averaged a higher percent plecopteran genera than continuously grazed ($p = 0.0105$) and grassy buffer ($p = 0.0186$) reaches. The same trends held for percent plecopteran abundance but with less statistical significance of the land use term ($p = 0.0894$) in the final model.

The covariate term was highly significant in the final model for number of dipteran genera, and the land use term was significant ($p = 0.0066$). Grassy buffer reaches had significantly fewer dipteran genera than continuously grazed ($p = 0.0097$) and IRG ($p = 0.0220$) reaches and marginally fewer than woody buffer ($p = 0.0933$) reaches.

Some evidence ($p = 0.1052$) suggested the EPT genera responded differently to the land use treatments, and the covariate term was highly significant. Grassy buffer reaches averaged a higher percentage of EPT genera (mean = 33.1) than continuously grazed reaches (mean = 26.1) with some statistical evidence ($p = 0.0934$), whereas IRG (mean = 32.7) and woody reaches (mean = 31.9) had no statistical evidence of differences after adjustments for multiple comparisons.

Feeding ecology. Feeding ecology metrics by percent individual abundance and generic abundance suggested woody and grassy buffers had very different assemblages. The covariate term was highly significant in all feeding ecology

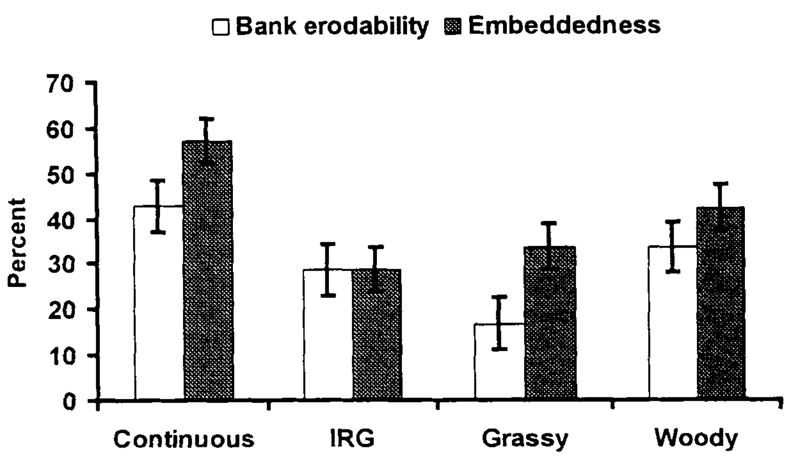

Figure 2. Predicted mean values (± 1 s.d.) for percent bank erodability and embeddedness of coarse substrates in the four riparian land use treatments. Continuous = continuous grazing, IRG = intensive rotational grazing, Grassy = grassy buffer strip, and Woody = woody buffer strip.

Table 2. Macroinvertebrate metrics with a land use term ($p \leq 0.10$) indicating a difference among treatments after adjusting for multiple comparisons. Metrics with a significant interaction term were excluded. Treatment means were estimated as the least squares mean. If the covariate term was significant, the least squares mean factored the covariate term into the estimate. Treatments with the same letter following the estimated means are not significantly different at the $p \leq 0.05$.

	Estimated mean for each land use treatment			
Metric	IRG	Continuous	Grass	Wood
Organic pollution				
MPTV[a]	4.139 ab	4.417 a	4.098 ab	3.968 b
Taxa richness				
Species richness	17.1 ab	18.0 a	15.0 b	17.2 ab
Generic richness	15.8 ab	16.6 a	13.8 b	16.1 ab
Plecoptera % genera	2.9 a	0.01 b	0.07 b	2.4 a
Diptera genera	8.95 a	9.26 a	6.53 b	8.44 ab
Feeding ecology by abundance				
% shredder	2.5 ab	4.1 ab	0.9 a	4.6 b
% scraper	17.1 ab	12.0 a	14.6 ab	28.5 b
% collector	73.8 ab	72.3 ab	82.7 a	58.8 b
Feeding ecology by genera				
% collector	66.8 ab	67.7 ab	73.1 a	65.2 b

[a] treatment term for MPTV was only marginally significant at $p = 0.0561$

metrics except percent scraper and collector individuals, and percent scraper genera. Percent abundance of shredder ($p = 0.0330$), scraper ($p = 0.0395$), and collector individuals ($p = 0.0129$), and percent collector ($p = 0.0309$) and filterer genera ($p = 0.0120$) were significantly different among land uses. A riffle beetle (*Optioservus fastiditus)* was the most abundant scraper that especially dominated the total abundance at some woody buffer sites. The percent of scraper abundance was higher in woody buffer treatment reaches than in continuously grazed reaches ($p = 0.0389$) and marginally higher than grassy buffer reaches ($p = 0.0868$), whereas IRG treatment reaches were intermediate and not significantly different from other treatments. The percent of shredder abundance was higher in woody than grassy buffer reaches ($p = 0.0410$), whereas IRG and continuous grazing treatment reaches were intermediate and not significantly different. The percent of collector abundance was higher in woody than grassy buffer treatment reaches ($p = 0.0068$), whereas IRG and continuous grazing treatment reaches were intermediate and not significantly different. The percent of collector genera was also higher in woody than grassy buffer treatment reaches ($p = 0.0281$), whereas IRG and continuous grazing treatment reaches were intermediate and not significantly different. Marginal evidence suggests that woody buffer ($p = 0.0527$) and IRG ($p = 0.0624$) treatments had a lower percent of filterer genera than grassy buffer reaches.

DISCUSSION

Macroinvertebrate response to watershed condition. Accounting for inherent stream variability in comparative analyses of multiple streams is a challenge for scientists. Categorizing streams within a narrow range of physicochemical and geographic constraints may account for much of the natural variability among the studied reaches. Ideally, we would have a population of streams to study that only differed in the riparian land use treatment. Since this was impossible to find, we expected some inherent variability among streams to exist, even though we used physicochemical and geographic guidelines for site selection.

Watershed condition explained most macroinvertebrate assemblage characteristics, indicating the importance of statistically accounting for watershed variation among streams in comparative analyses. Our results demonstrated that substantial, inherent differences among streams are likely to persist even with careful selection of similar streams from one ecoregion. Differences that may be difficult to account for include variation in watershed land cover, stream size, habitat dynamics, thermal regime, and various water chemistry characteristics. We noticed very different results in comparing ANOVA models to ANCOVA models in which the covariate term was significant, even differences in the order the treatment groups were ranked by their means. Analysis of covariance essentially allowed us to statistically account for inherent variability among streams and compare the treatment reaches as if the only difference among reaches was the land use treatment.

Macroinvertebrate response to local riparian land use treatments. We found evidence that macroinvertebrate assemblages change within about 100 - 300 m of stream in response to riparian land use treatment after we statistically accounted for inherent watershed differences among streams. Although the direct mechanisms that influenced the macroinvertebrates are not easily identifiable in a comparative analysis, we suggest the physical in-stream habitat associated with the land use treatments also helps explain the macroinvertebrate – land use associations.

Organic pollution. The mean pollution tolerance value (MPTV) indicated a marginally significant difference between continuously grazed and woody buffer reaches. In other studies of agricultural effects on stream macroinvertebrates, both Richards et al. (1993) and Lenat (1984) found that water quality (i.e., nutrients/organic pollution) has less impact on macroinvertebrates than physical variables like sedimentation. In this study, we found the macroinvertebrate assemblage responded in a way that suggests higher organic pollution in the continuously grazed reaches than the woody buffer reaches.

Taxa richness. We found that continuously grazed reaches, the reaches with the most erodable banks and embeddedness of coarse substrates, have the highest species and generic richness and lowest representation of EPT taxa. Taxa have differential responses to increasing fine sediment, and there tends to be a shift in the macroinvertebrate community toward more tolerant taxa (Richards et al. 1993). It is possible that low levels of sedimentation may create a new habitat type and increase taxa richness. Furthermore, high taxa richness in the continuously grazed reaches may be in response to slightly increased organic inputs, as suggested by MPTV. Taxa richness may increase as energy sources increase in a system originally with relatively low nutrients, but as the inputs become elevated to the level of stressors, there is a decline in taxa richness.

Feeding ecology. The feeding ecology metrics did not show consistent differences between IRG and continuous grazing nor between the grazing regimes and the buffer treatments, but feeding ecology measures differentiated the buffers. Woody buffer reaches had the most specialists (i.e., shredders and scrapers) and the fewest generalists (i.e., filterers and gatherers, or collectors), whereas grassy buffer reaches had the fewest specialists and the most generalists. Continuously grazed reaches had the fewest scrapers by abundance, but otherwise, continuous grazing and IRG reaches were intermediate between woody and grassy buffer reaches for feeding ecology metrics.

Short retention time of coarse particulate organic matter in riffles (Sweeney 1993) and the fact that we sampled only riffles may have contributed to the overall low shredder abundances. Seasonal allochthonous input of leaves (Sweeney 1993, Wiley et al. 1990) and wider channels in wooded reaches probably increased available niches (Sweeney 1993), which led to more specialized taxa in woody reaches. Continuously grazed and IRG reaches did not have macroinvertebrate feeding ecology percentages consistently more similar to grassy buffers than to woody buffers as expected. One IRG site was slightly influential in that it had some trees in the stream-side paddocks. Both grazing treatments had allochthonous input in the form of manure since cattle had direct stream access. Undigested or partially digested material in manure at the grazing treatment reaches may have served as a surrogate carbon source for leaves and woody debris at the woody buffer treatment reaches.

Conclusions. Our results demonstrate the importance of accounting for inherent stream variability in comparative analyses of multiple streams. Without sampling two sites from each stream and using the upstream site as a covariate in our ANCOVA models, we believe we would have been misled by our results. Our results suggest macroinvertebrate assemblages have a greater response to watershed conditions than to local riparian conditions.

Within 100 - 300 m of stream we had some macroinvertebrate and habitat responses to local riparian land use, and we expect these differences would be more apparent at greater stream-reach distances. We conclude from the MPTV and taxonomic richness metrics that continuous grazing in riparian areas has negative impacts on macroinvertebrates assemblages. Intolerant macroinvertebrates from specialized feeding guilds composed the macroinvertebrate assemblage at woody buffer stream reaches, suggesting woody buffer reaches are least impacted. Physical in-stream habitat, and EPT and percent plecopteran metrics suggest grassy buffer and IRG reaches are also less impacted than continuously grazed reaches. Even though grassy buffers and IRG reaches had less embeddedness of coarse substrate, the macroinvertebrates did not consistently respond to the improved habitat, supporting our conclusion that the macroinvertebrate assemblages are primarily responding to large-scale watershed influences such as land cover.

ACKNOWLEDGEMENTS

This research was supported in part by Federal Aid in Sport Fish Restoration, Agriculture in Concert with the Environment, and Sustainable Agriculture Research and Education programs, North Central Region, U.S. Department of Agriculture. B. Weigel is grateful for the Bill Beck Memorial Scholarship awarded by the Central Wisconsin Chapter of Trout Unlimited and additional scholarship support through the Oshkosh Foundation. We thank Paul Rasmussen (WDNR) for help in designing the statistical analysis, Jeff Dimick and Richard Lillie for quality assurance of taxonomy, and the following individuals for valuable field or laboratory assistance: Heath Benike, Steve Clark, Jason Freund, Scott Harpold, Paul Kanehl, Matt Keefer, Jeff Knetter, Chris Natoli, and Lizhu Wang. We thank Joseph Kawatski and anonymous reviewers for comments on the manuscript. We are grateful for the cooperation and input from the farmers upon whose land we worked.

LITERATURE CITED

Barling, R. D., and I. D. Moore. 1994. Role of buffer strips in management of waterway pollution: a review. Environmental Management 18:543-558.

Draper, N. R., and H. Smith. 1981. Applied regression analysis. Second edition. John Wiley and Sons, New York.

Fortenbery, T. R., and W. E. Saupe, editors. 1995. Status of Wisconsin farming, 1995. Publication of the Department of Agricultural Economics, University of Wisconsin-Madison.

Graczyk, D. J., and W. C. Sonzogni. 1991. Reduction of dissolved oxygen concentration in Wisconsin streams during summer runoff. Journal of Environmental Quality 20:445-451.

Hilsenhoff, W. L. 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist 20:31-39.

Hilsenhoff, W. L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society 7:65-68.

Karr, J. R., and I. J. Schlosser. 1978. Water resources and the land-water interface. Science 201:229-234.

Karr, J. R., and E. W. Chu. 1997. Biological monitoring and assessment: using multimetric indexes effectively. EPA 235-R97-001. University of Washington, Seattle.

Kauffman, J. B., and W. C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. Journal of Range Management 37:430-437.

Knox, J. C. 1977. Human impacts on Wisconsin stream channels. Annals of the Association of American Geographers 67:323-342.

Lenat, D. R. 1984. Agriculture and stream water quality: a biological evaluation of erosion control practices. Environmental Management 8:333-344.

Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. Journal of the North American Benthological Society 7:222-233.

Lenat, D. R., and D. L. Penrose. 1996. History of the EPT taxa richness metric. North American Benthological Society Bulletin 13:305-307.
Lillie, R. A., and R. A. Schlesser. 1994. Extracting additional information from biotic index samples. Great Lakes Entomologist 27:129-136.
Lyons, J., L. Wang, and T. D. Simonson. 1996. Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. North American Journal of Fisheries Management 16:241-256.
Merritt, R. W., and K. W. Cummins, editors. 1984. An introduction to the aquatic insects of North America, 2nd edition. Kendall/Hunt Publishing Company, Dubuque, Iowa.
Omernik, J. M., and A. L. Gallant. 1988. Ecoregions of the upper Midwest states. U.S. Environmental Protection Agency, EPA/600/3-88/037, Corvallis, Oregon.
Osborne, L. L., and M. J. Wiley. 1988. Empirical relationships between land use/cover and stream water quality in an agricultural watershed. Journal of Environmental Management 26:9-27.
Osborne, L. L., and D. A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29:243-258.
Owens, L. B., W. M Edwards, and R. W. Van Keuren. 1996. Sediment losses from a pastured watershed before and after stream fencing. Journal of Soil and Water Conservation 51:90-94.
Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross, and R. M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. EPA/440/4-89-001. Office of Water, US Environmental Protection Agency, Washington, DC.
Platts, W. S., and F. J. Wagstaff. 1984. Fencing to control livestock grazing on riparian habitats along streams: is it a viable alternative? North American Journal of Fisheries Management 4:266-272.
Potter, K. W. 1991. Hydrological impacts of changing land management practices in a moderate-sized agricultural catchment. Water Resources Research 27:845-855.
Richards, C., G. E. Host, and J. W. Arthur. 1993. Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment. Freshwater Biology 29:285-294.
Rosenberg, D. M., and V. H. Resh, eds. 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.
SAS 1990. SAS/STAT user's guide. Version 6. Fourth Edition. SAS Institute, Inc., Cary, North Carolina.
Simonson, T. D., J. Lyons, and P. D. Kanehl. 1994. Guidelines for evaluating fish habitat in Wisconsin streams. U.S. Forest Service General Technical Report NC-164.
Sweeney, B. W. 1993. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in eastern North America. Proceedings of The Academy of Natural Sciences of Philadelphia 144:291-340.
Szczytko, S. W. 1996. Bugprogram documentation for rapid bioassessment monitoring. College of Natural Resources, University of Wisconsin, Stevens Point.

Thorn, W. C., C. S. Anderson, W. E. Lorenzen, D. L. Hendrickson, and J. W. Wagner. 1997. A review of trout management in southeast Minnesota streams. North American Journal of Fisheries Management 17:860-872.
Trimble, S. W. 1994. Erosional effects of cattle on streambanks in Tennessee, U.S.A. Earth Surface Processes and Landforms 19:451-464.
Trimble, S. W. 1997. Stream channel erosion and change resulting from riparian forests. Geology 25:467-469.
Trimble, S. W., and A. C. Mendel. 1995. The cow as a geomorphic agent - a critical review. Geomorphology 13:233-253.
Undersander, D. J., B. Albert, P. Porter, A. Crossley, and N. Martin. 1993. Pastures for profit: a guide to rotational grazing. University of Wisconsin-Extention, Madison, Wisconsin. Publication A3529.
Wang, L., J. Lyons, P. Kanehl, R. Gatti. 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. Fisheries 22:6-12.
Wiley, M. J., L. L. Osborne, and R. W. Larimore. 1990. Longitudinal structure of an agricultural prairie river system and its relationship to current ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences 47:373-384.
WDNR (Wisconsin Department of Natural Resources). 1980. Wisconsin trout streams. Wisconsin Department of Natural Resources, Madison. Publication 6-3600(80).
WDNR (Wisconsin Department of Natural Resources). 1994. Wisconsin water quality assessment report to Congress. Wisconsin Department of Natural Resources, Madison. Publication WR254-94-REV.
WDNR (Wisconsin Department of Natural Resources). 1995. Field procedures manual. Part B: collection procedures, benthic invertebrate surveys – benthic samples, 702.1. Wisconsin Department of Natural Resources, Madison.
Wohl, N. E., and R. F. Carline. 1996. Relations among riparian grazing, sediment loads, macroinvertebrates, and fishes in three central Pennsylvania streams. Canadian Journal of Fisheries and Aquatic Sciences 53:260-266.

Received: 2 August 1999 Accepted: 21 December 1999