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Effects of Local Land Use on Physical Habitat, Benthic Macroinvertebrates, and Fish in the Whitewater River, Minnesota, USA

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ABSTRACT / Best management practices (BMPs) have been developed to address soil loss and the resulting sedimentation of streams, but information is lacking regarding their benefits to stream biota. We compared instream physical habitat and invertebrate and fish assemblages from farms with BMP to those from farms with conventional agricultural practices within the Whitewater River watershed of southeastern Minnesota, USA, in 1996 and 1997. Invertebrate assemblages were assessed using the US EPA's rapid bioassessment protocol (RBP), and fish assemblages were assessed with two indices of biotic integrity (IBIs). Sites were classified by upland land use (BMP or conventional practices) and riparian management (grass, grazed, or wooded buffer). Physical habitat characteristics differed across buffer types, but not upland land use, using an analysis of covariance, with buffer width and stream as covariates. Percent fines and embeddedness were negatively correlated with buffer width. Stream sites along grass buffers generally had significantly lower percent fines, embeddedness, and exposed streambank soil, but higher percent cover and overhanging vegetation when compared with sites that had grazed or wooded buffers. RBP and IBI scores were not significantly different across upland land use or riparian buffer type but did show several correlations with instream physical habitat variables. RBP and IBI scores were both negatively correlated with percent fines and embeddedness and positively correlated with width-to-depth ratio. The lack of difference in RBP or IBI scores across buffer types suggests that biotic indicators may not respond to local changes, that other factors not measured may be important, or that greater improvements in watershed condition are necessary for changes in biota to be apparent. Grass buffers may be a viable alternative for riparian management, especially if sedimentation and streambank stability are primary concerns.

Water pollution in the United States is a common and widespread problem. Currently, most water pollution is attributed to nonpoint sources (US Environmental Protection Agency 1996). As the name implies, nonpoint-source pollution arises across wide areas, such as agricultural fields throughout a watershed. Agriculture is currently the leading source of water pollution and a contributing factor to impairment of 70% of streams considered impaired in the 1996 National Water Quality Inventory (US Environmental Protection Agency 1996).

For streams included in the 1996 US Environmental Protection Agency (USEPA) survey, the most common agricultural pollutant was sediment, which was a contributing factor for 50% of impaired streams (US Environmental Protection Agency 1996). Agricultural practices can contribute sediment to streams in several ways. In row-crop agriculture, bare soil between rows is easily eroded and can be transported to streams via runoff (Waters 1995). Grazing in riparian areas can reduce vegetation on streambanks, making them more susceptible to erosion (Kauffman and Krueger 1984, Trimble and Mendel 1995, Fitch and Adams 1998, Strand and Merritt 1999). Sediment delivered to streams can either be suspended and create turbid water, or settle to the stream bottom. Although some sediment input to streams is natural, excessive sediment is a pollutant and can have negative effects on stream biota (Waters 1995).

Due to the negative impacts of sedimentation on fish and invertebrates, and the impact of soil loss on agricultural productivity, reduction of soil erosion and sediment delivery to streams has been a major goal of conservation agencies. Several strategies have been employed to reduce soil erosion. One is to remove marginal lands from production, as in the Conservation

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Reserve Program (CRP). Other solutions fall under the umbrella of techniques known as best management practices (BMPs). BMPs are intended to reduce agricultural soil loss while at the same time allowing land to remain in production (Amemiya 1970).

BMPs for row-crop agriculture employ alternative methods of tillage (e.g., chisel plow, ridge till, contour plowing, strip cropping, etc.). BMP tillage does not till residue under completely, as does conventional tillage with a moldboard plow, but leaves some on the surface to hold soil in place and retard the movement of soil suspended in runoff (Amemiya 1970). Riparian buffers act as a BMP by filtering sediment from agricultural runoff (Wilkin and Hebel 1982, Daniels and Gilliam 1996) and by stabilizing streambanks. Riparian buffers also benefit streams by conserving riparian vegetation and preventing streambank trampling by livestock (Johnson and Moldenhauer 1970, Marcuson 1977, Barling and Moore 1994, Trimble and Mendel 1995). Wider buffers have shown increased capacity to filter sediment (Erman et al. 1977, Neibling and Alberis 1979, Dillaha et al. 1989). Comparisons of grass, trees, and grazed buffers have yielded conflicting results, with some studies finding grass buffers to be superior at filtering sediment (Rabeni and Smale 1995, Daniels and Gilliam 1996), whereas others have found trees to be best (Erman et al. 1977, Beschta and Platts 1986). Although there have been studies of the relative soil loss from different types of tillage (Blevins et al. 1990, Razavian 1990, Gaynor and Findlay 1995), and comparisons of sedimentation rates from different riparian buffer types (Erman et al. 1977, Beschta and Platts 1986, Daniels and Gilliam 1996), assessment of their benefits to stream biota is lacking.

Reductions in sediment input from agriculture are important to stream biota because many invertebrates and fishes require a streambed relatively free of fine sediment (Minshall 1984, Waters 1995). Excess amounts of sediment deposited in the stream can reduce food resources and habitat for invertebrates by covering hard substrates and filling interstitial spaces. Invertebrates that scrape algae from hard substrates for food are expected to decline. In contrast, invertebrates that filter food from the water column will increase. Herbivorous fish similarly decline with increasing sedimentation and are replaced by generalist and omnivorous feeders (Berkman and Rabeni 1987). Invertebrates that require interstitial spaces tend to decline with increases in sedimentation and are usually replaced by burrowing taxa that prefer silt habitats (Lenat et al. 1979, 1981, Lemly 1982). The density of invertebrates may decrease with the loss of invertebrates sensitive to sedimentation (Lemly 1982), but densities often remain the same due to an increase in the abundance of sediment-tolerant taxa (Lenat et al. 1979). Fish reproductive success can also decrease due to sedimentation that can smother eggs (Cordone and Kelly 1961, Platts and Megahan 1975, Berkman and Rabeni 1987, Chapman 1988).

Due to their ease of interpretation, water quality or physical habitat sampling, such as the protocols suggested by Platts et al. (1987), were historically used more often than biotic indicators to determine impacts of pollution (Cairns and Pratt 1993, Davis 1995). As understanding of the responses of fish and invertebrates to pollution has increased, use of biotic indicators of environmental conditions has grown, either alone or in conjunction with physical habitat monitoring (Karr 1981, Cairns and Pratt 1993). Biotic indicators are advantageous because a host of environmental factors are integrated by the community (Karr 1981, Karr et al. 1986). Biota also incorporate a longer time period than a measure of physical habitat at a fixed point (Karr 1981, Karr et al. 1986). Therefore, sampling only physical habitat parameters may miss critical impacts that biotic indicators may reflect.

Based upon knowledge of how nonpoint sources affect fish and invertebrates, metrics to quantify differences in aquatic communities have been developed. Responses by invertebrates and fish to nonpoint-source degradation have been detected by examining changes in trophic, reproductive and functional feeding guilds (Berkman and Rabeni 1987, Plafkin et al. 1989); species richness (Hendicks et al. 1980, Lemly 1982, Cooper 1987, Lenat and Crawford 1994); and tolerance to pollution (Hilsenhoff 1982 and 1987). Impacts can more easily be detected when using an approach such as the index of biological integrity (IBI) or the rapid bioassessment protocol (RBP), which integrates several metrics rather than simply using a single metric (Karr 1987, Plafkin et al. 1989).

The goal of this study was to compare the effects of conventional and BMP agricultural practices, including riparian buffers, on stream ecosystems. Our specific objectives were to compare upland practices and riparian management types using physical habitat variables (Platts and others 1983), upland practices and riparian management types using multimetric indices for invertebrates (Plafkin and others 1989) and fish (Karr and others 1986), and to relate multimetric indices to physical habitat variables.

Methods

Study Site

The Whitewater basin drains approximately 100,000 ha in the southeastern Minnesota counties of Olmsted,

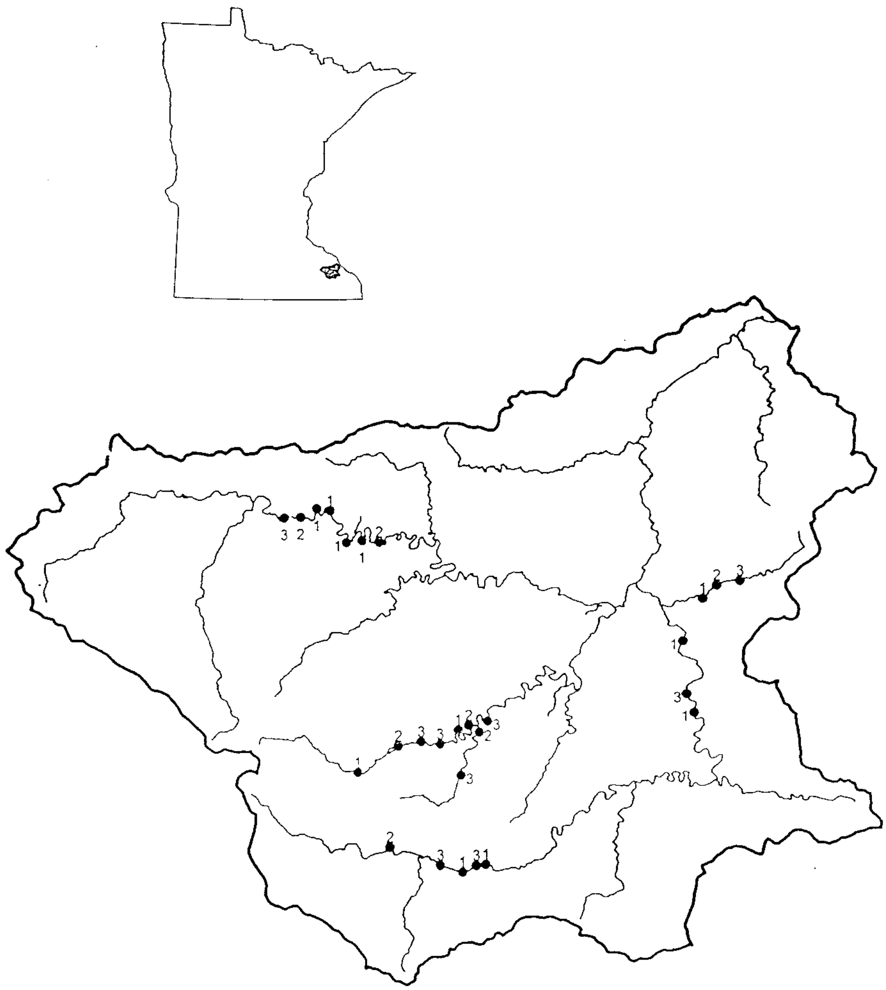

Figure 1. Location of study sites within the Whitewater River watershed in southeastern Minnesota. Numbers associated with each site indicate buffer condition; 1 = wooded, 2 = grass, 3 = grazed; associated upland conditions are listed in Table 1.

Winona, and Wabasha. The Whitewater River has three main branches, the North, Middle, and South, which meet to form the main stem before emptying into the Mississippi River (Figure 1).

The topography of the watershed has been shaped by glacial history (Broussard et al. 1975). The basin lies within a relatively unglaciated region that Omernik and Gallant (1988) termed the Driftless Area Ecoregion. However, the main branches originate in a glaciated, relatively flat area, the Western Corn Belt Plains (Omernik and Gallant 1988). The streams meander and flow slowly through rolling plains before entering deep, unglaciated valleys in the middle portion of the watershed. The unglaciated valleys receive significant inputs of groundwater from underlying limestone and sandstone bedrock. The upper portions of all three branches, which receive little groundwater input, harbor a warmwater fish assemblage including darters, minnows, and suckers. The middle and lower portions contain a coldwater assemblage dominated by salmonids and sculpin.

Following European settlement in the second half of the nineteenth century, most of the watershed was converted to agriculture. Severe erosion occurred during the early 1900s due to the lack of soil conservation practices (Waters 1977, Trimble and Lund 1982, Trimble 1999). As better soil conservation practices were implemented in upland areas under cultivation, the range of coldwater fishes increased (Thorn et al. 1997). Soil loss from upland areas has mostly been abated (Trimble and Lund 1982). Most sediment entering streams is currently from streambank erosion that takes place as streams cut into banks of alluvial sediment (Trimble 1993, 1997, 1999) deposited from the uplands prior to 1940 (Trimble 1999, Trimble and Lund 1982). Current land use in the watershed is 46% in cropland, 25% pasture, 24% woodland, and 5% in other land uses (Minnesota Pollution Control Agency

Table 1. Land classification, riparian type and stream type of Whitewater River sampling sites for 1996 and 1997[a]

Site	Branch	Upland use	Upland class	Buffer type	Stream type	County
South 1*	Upper South	Row crop	BMP	Grass	Warm	Olmsted
South 2*	Upper South	Grazed	Conv	Grazed	Warm	Olmsted
South 3	Upper South	Row crop	Conv	Wooded	Warm	Olmsted
South 4	Upper South	Row crop	BMP	Grazed	Warm	Olmsted
South 5	Upper South	Row crop	Conv	Wooded	Warm	Olmsted
South 6	Lower South	Wooded	BMP	Wooded	Cold	Winona
South 7	Lower South	Grazed	Conv	Grazed	Cold	Winona
South 8	Lower South	Wooded	BMP	Wooded	Cold	Winona
South 9	South Trib.	Grazed	BMP	Wooded	Cold	Winona
South 10	South Trib.	Grazed	BMP	Wooded	Cold	Winona
South 11	South rib.	Wooded	BMP	Wooded	Cold	Winona
Middle 1	Upper Middle	Row crop	Conv	Wooded	Warm	Olmsted
Middle 2*	Upper Middle	Row crop	Conv	Grass	Warm	Olmsted
Middle 3	Upper Middle	Row crop	Conv	Grazed	Warm	Olmsted
Middle 4*	Upper Middle	Row crop	Conv	Grazed	Warm	Olmsted
Middle 5	Upper Middle	Row crop/ grazed	BMP	Wooded	Warm	Olmsted
Middle 6	Upper Middle	Row crop	BMP	Grass	Warm	Olmsted
Middle 7	Middle Trib.	Grazed	Conv	Grazed	Cold	Olmsted
Middle 8	Middle Trib.	Grazed	Conv	Grass	Cold	Olmsted
Middle 9	Upper Middle	Grazed	Conv	Grazed	Cold	Olmsted
North 1*	Upper North	Row crop	BMP	Grazed	Warm	Wabasha
North 2*	Upper North	Row crop	BMP	Grass	Warm	Wabasha
North 3	Upper North	Row crop	BMP	Wooded	Warm	Wabasha
North 4*	Upper North	Row crop	Conv	Wooded	Warm	Wabasha
North 5	Upper North	Row crop	BMP	Wooded	Warm	Wabasha
North 6	Upper North	Wooded	BMp	Wooded	Cold	Wabasha
North 7	Upper North	Grazed	Conv	Grass	Cold	Wabasha

[a]Asterisks indicate sites sampled only in 1997. BMP = best management practice, Conv = conventional practice.

1996). However, cropland is disproportionately (>70%) represented along the warmwater reaches in the glaciated, relatively flat area where the streams originate. When the main branches enter coldwater reaches in the Driftless Area, cropland is replaced by broad forested areas, primarily wildlife management areas administered by the Minnesota Department of Natural Resources. Thus, cropland is the dominate land use in the catchment areas upstream of most sites.

Study sites were selected and classified based on upland and riparian land use. Upland land use at a site was classified as a BMP if no-till, reduced tillage, or contouring was in place; otherwise, the site was classified as conventional. Uplands were designated as the local farmstead on either side of the stream adjacent to the riparian zone. Riparian buffers were classified by their dominant vegetation type: ungrazed grass, grazed grass, or wooded within 150 m of the stream. In 1996, 20 sites were sampled; in 1997, seven additional sites were sampled, for a total of 27 sites (Table 1, Figure 1). Sites were added in 1997 so that the number of sites in each combination of upland and riparian land use was more balanced. Buffer width varied significantly with buffer type ($P = 0.075$). In 1997, wooded buffers (94 ± 14 m) were wider than grass buffers (32 ± 16 m), with grazed buffers (74 ± 15 m) intermediate and not significantly different than wooded areas.

Eight sites were located in the coldwater portion of the watershed, and all others were in warmwater portions of the watershed (Table 1). Sites were closely grouped along six stream segments to minimize variation in large-scale influences (i.e., soil type, topography, or surficial materials of the watershed and stream order) within each group. Sites were located in third- and fourth-order reaches in all main branches of the Whitewater, as well as along two second-order tributaries to minimize hydraulic differences.

Physical Habitat

Sampling was conducted in August and September each year using a transect method based on mean stream width (MSW), following the protocol of Simonson et al. (1994). Sampling sites were chosen at the downstream end of the appropriate buffer type. MSW for a site was determined by averaging 10 width measurements. Sites that were 5-m or less in width were sampled with 13 transects spaced 3 MSW apart. Sites wider than 5-m had 20 transects spaced 2 MSW apart.

A portion of the instream measures was taken at four evenly spaced points along each transect. At each point, depth was measured using a wading rod, velocity was measured at 0.6 of total depth using a Marsh-McBirney Model 2000 flowmeter, substrate composition was estimated visually based on a modified Wentworth scale (Bovee and Cochnauer 1977), and embeddedness was estimated visually to the nearest 20% (Platts et al. 1983). The mean of four variables was estimated visually to the nearest 5% within a section of the stream 1 MSW in length, centered on the transect: the percent of stream shaded by the canopy at noon, percent riffle, pool, or run; percent of exposed soil along the streambank; and percent cover for fish 200 mm or larger. Habitats included as cover were overhanging bank vegetation, woody debris, instream vegetation, and boulders.

Riparian measures were taken on only one streambank per transect, alternating the side measured. The width of the riparian buffer was measured to the nearest meter. The percentage of the vegetation as grass, forb, tree, and shrub was estimated visually to the nearest 5% for each category along the transect. The length of overhanging vegetation where the transect intersected the streambank was measured to the nearest 0.1 m. Data were aggregated across all transects for each site to calculate mean values.

Invertebrates

Invertebrate sampling was conducted during the first week of June each year based on the US EPA Rapid Bioassessment Protocol III (RBP) (Plafkin et al. 1989). Separate invertebrate samples were collected at each of the first three riffles encountered above the downstream end of the study reach. Invertebrates were sampled using a kick net (30 sec/sample) in a high velocity and a low velocity portion of each riffle, and combined into a composite sample for a total of three samples per study reach.

Three coarse particulate organic matter (CPOM; decaying leaves and wood) samples were collected at each site. Each CPOM sample was covered with water and agitated; then the water, suspended debris, and organisms were poured through a 0.419-mm-mesh sieve. This procedure was repeated five times to ensure that most invertebrates were dislodged. All invertebrates and debris from the kick net and CPOM samples were preserved in 70% ethanol for later sorting and identification.

In the laboratory, invertebrates were sorted, identified, and counted following the 100 fixed-count method of Plafkin et al. (1989) for the kick net and CPOM samples. This methodology may underestimate site quality by underestimating taxa richness, but relative comparisons among sites should not be biased (Hilsenhoff 1987, Plafkin et al. 1989, Sovell and Vondracek 1999). Invertebrates other than chironomids were identified to the taxonomic level where Hilsenhoff (1987) assigned tolerance values, usually genus or species. Chironomidae were classified to subfamily as either blood red or not-blood red and assigned tolerance values based on a family level biotic index (Hilsenhoff 1987). All identified organisms were classified to functional feeding group to calculate RBP scores following Merritt and Cummins (1996). For our analysis, we used a mean of the three riffle and three CPOM samples at each site.

RBP metrics were calculated for each sample; however, only mean site values are reported. Mean values for each site were compared to a reference site to generate a score for that site. Due to the lack of a suitable reference site in the area, the highest scoring site among those sampled was used as a reference. This may potentially bias scoring to make all sites appear higher in quality than if a truly high-quality reference site was used. However, this study is intended to compare sites with each other and not with sites outside the region; thus any introduced bias should have little impact on interpretation.

Fish

Fish were collected in late June/early July using a backpack electrofisher in 150-m stream reaches, rather than the minimum of 35 times MSW recommended by Lyons (1992a). Only for sites that were <4.3 m wide did we satisfy Lyons' (1992a) protocol, but for the following reasons we contend an adequate distance was covered in the streams that were >4.3 m wide. Species diversity was low, especially at coldwater sites, making it easier to capture all species present. Furthermore, sampling three riffle/pool sequences is considered adequate (Lyons 1992a), and we fulfilled this condition at all sites. Most captured fish were identified on site and released. Specimens not identified were preserved in 70% ethanol and identified in the laboratory following Smith (1989).

Coldwater and warmwater fish assemblages differ in their functional components and response to water quality degradation. Therefore, data were analyzed with two separate IBIs. Sites designated as trout waters by the Minnesota Department of Natural Resources and that contained trout were classified as coldwater sites. All other sites were classified as warmwater. For warmwater sites, we used an IBI developed by Lyons (1992b). For

Table 2. Values for physical habitat variables significantly different across buffer types in Whitewater River watershed in 1996 and 1997

Variable	Buffer Type (mean ± SE)			P
	Wood	Grass	Grazed	
1996				
Width-to-depth	7.7 ± 3.0	8.2 ± 4.7	8.3 ± 3.9	0.905
% Cover	10.8 ± 4.7	17.9 ± 7.4	12.8 ± 6.0	0.752
% Fines	52.8 ± 4.5	22.8 ± 7.1	46.7 ± 5.9	0.030
Embeddedness	66.2 ± 5.5	53.65 ± 10.91	38.4 ± 8.7	0.041
Exposed bank	31.7 ± 5.1	12.4 ± 8.0	10.0 ± 6.6	0.072
Overhanging vegetation	0.10 ± 0.11	0.85 ± 0.18	0.20 ± 0.14	0.052
1997				
Width-to-depth	6.1 ± 0.5	4.1 ± 0.5	5.0 ± 0.5	0.078
% Cover	7.7 ± 1.8	17.4 ± 2.0	7.9 ± 1.8	0.014
% Fines	60.0 ± 4.0	38.7 ± 4.3	59.8 ± 4.0	0.019
Embeddedness	69.5 ± 3.3	54.9 ± 3.5	72.6 ± 3.2	0.018
Exposed bank	44.6 ± 4.5	4.1 ± 4.8	27.7 ± 4.4	<0.001
Overhanging vegetation	0.32 ± 0.06	0.81 ± 0.07	0.27 ± 0.06	<0.001

coldwater sites, we used a coldwater IBI developed for use in streams in the upper midwestern United States (Mundahl and Simon 1999). Because comparisons between scores from the two IBIs are difficult due to differences in metrics, analyses of fish data were performed separately for warmwater and coldwater sites.

Statistical Analysis

Physical habitat, invertebrate, and fish scores among sites were compared using a factorial analysis of covariance (ANCOVA). The ANCOVA model used buffer and upland land use as main effects and included buffer × upland interaction, with buffer width as a covariate and stream segment as a blocking variable. Due to high variability common in observational field experiments, all tests were considered significant at $P = 0.10$. When a significant difference was found using ANCOVA, groups were compared using Tukey's Honestly Significant Difference test for multiple comparisons. ANCOVA and multiple comparisons were carried out using the MACANOVA statistical program (Bingham and Oehlert 1998).

We investigated effects of instream physical habitat on invertebrates and fish using redundancy analysis (RDA), a constrained form of multiple regression of a multivariate set of predictor variables on a multivariate set of response variables, in the Canoco computer program (ter Braak 1987–1992). All variables were standardized to unit variance prior to analysis to correct for measurements taken with different units. RDA was performed using instream physical habitat measures as explanatory variables and either invertebrate RBP or fish IBI metrics as response variables. The significance of the RDA model was tested using a Monte Carlo permutation (ter Braak 1987–1992). F values generated from each Monte Carlo permutation are compared with the F value for the original set. If the original F value was among the highest 5% of 100 random sets, the ordination was considered significant (ter Braak 1987–1992).

Variables with a correlation >0.50 from the RDA were considered strong predictors. Strong predictors were regressed versus either RBP or IBI score using linear and multiple regression to illustrate the relationship between physical habitat variables and fish or invertebrate assemblages.

Results

Physical Habitat

No significant differences were noted in physical habitat variables between sites in different upland land use. However, significant differences in physical habitat measures were detected among sites with different riparian buffer types (Table 2). Buffer width was used as a covariate because greater benefits were expected from wider buffers (e.g., Petersen et al. 1992). This relationship was confirmed; percent fines and embeddedness were both significantly and negatively correlated with buffer width in both 1996 ($r^2 = 0.342$, $P = 0.008$) and 1997 ($r^2 = 0.248$, $P = 0.010$). In general, physical habitat characteristics along grass-buffer sites were significantly different from wooded sites, whereas grazed buffers were intermediate.

Width-to-depth ratio and percent cover were significantly different across buffer types in 1997 (Table 2). Width-to-depth ratios were significantly higher (P = 0.078) along wood-buffered sites than in grass-buffered sites, whereas grazed sites were not significantly different from wooded or grass sites. Percent cover was significantly higher (P = 0.014) along grass-buffered sites than along wood sites or along grazed sites. A portion of the difference can be accounted for by overhanging vegetation length, which was significantly higher (P < 0.001) along grass-buffered sites than along either wooded or grazed sites.

Instream substrate was significantly different across buffer types in both 1996 and 1997, primarily due to differences in fine particles (clay, silt, and sand ≤0.062 mm) (Table 2). In 1996, percent fines were significantly lower (P = 0.030) along grass-buffered sites than along wooded sites; grazed sites were intermediate and not significantly different from other buffer types. In 1997, percent fines were significantly lower (P = 0.019) along grass-buffered sites than either wooded or grazed buffers. This relationship was evident even though grass buffers were narrower than grazed or wooded areas. Embeddedness was significantly lower (P = 0.041) in 1996 along grass-buffered sites than along grazed sites, with wood sites intermediate. In 1997, embeddedness was significantly lower (P = 0.018) along grass buffers than along grazed or wooded sites. Although embeddedness was lower, mean values along grass buffers were >40% in 1996 and 1997.

The amount of exposed streambank was significantly different across buffer types (Table 2). In 1996, percent exposed streambank was significantly higher (P = 0.072) along wooded sites than along grazed or grass-buffered sites. In 1997, percent exposed streambank was significantly lower (P < 0.001) along grass-buffered sites than along wooded or grazed sites.

Invertebrates

We collected 100 invertebrate taxa across all sites in 1996 and 1997 (Nerbonne 1999). We found no significant differences in RBP scores (range 6–38 of a possible 48) across upland land use or riparian buffer type in either 1996 or 1997, indicating similar invertebrate assemblages. The Monte Carlo test of the RDA using 1996 data was not significant (P = 0.400); thus, no conclusions could be drawn. However, the RDA of invertebrate data collected in 1997 indicated significant correlations (P ≤ 0.010) between several physical habitat and invertebrate metrics (Figure 2). The first two axes of the RDA for 1997 physical habitat variables explained 51% of the variance in invertebrate metrics, with the first axis explaining 47% of the variance.

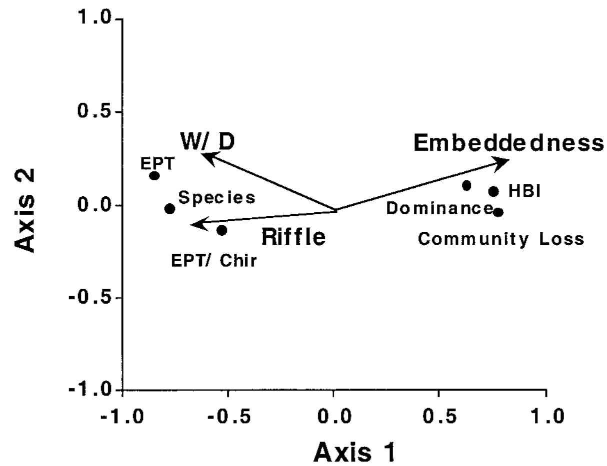

Figure 2. The first two RDA axes for invertebrate metrics from 1997. RBP metrics are displayed as closed circles and physical habitat as arrows. EPT = Ephemeroptera, Plecoptera, and Trichoptera; EPT/chir = number of EPT/number of Chironomidae; HBI = Hilsenhoff biotc index; W/D = width-to-depth ratio.

Within the first axis, most of the variation in invertebrate metrics was explained by embeddedness and to a lesser extent by percent riffle and width-to-depth ratio (Figure 2). HBI, community loss, and percent dominance, all indicators of a pollution tolerant assemblage, were positively associated with embeddedness and negatively associated with width-to-depth ratio and percent riffle habitat. Number of EPT taxa, number of species, and ratio of EPT to chironomids, all indicators of a pollution intolerant assemblage, were negatively associated with embeddedness and positively associated with width-to-depth ratio and percent riffle.

Physical habitat variables that explained a majority of variance in invertebrate data were significantly correlated with RBP scores, but explained only 25%–36% of the variance (Table 3). RBP scores were positively correlated with width-to-depth ratio (P = 0.010) and negatively correlated with embeddedness (P = 0.019) and percent fines (P = 0.067). Including width-to-depth ratio and either percent fines or embeddedness in a multiple-regression versus RBP score increased variance explained to 42% when embeddedness was used or to 37% with percent fines. Adding percent riffle as a third explanatory variable resulted in only negligible increases in variance explained. Embeddedness and percent fines were highly correlated and therefore not included in the same multiple regression.

Fish

We collected 25 species of fish across all sites in 1996 and 1997 (Nerbonne 1999). ANCOVA revealed no sig-

Table 3. *P* and r^2 values for rapid bioassessment protocol (RBP) and index of biotic integrity (IBI) scores in relation to width-to-depth ratio, percent fines, and embeddedness

Variable	*P*	r^2
RBP		
Width-to-depth ratio	0.010	0.247
Percent fines	0.067	0.279
Embeddedness	0.019	0.364
IBI		
Width-to-depth ratio	0.002	0.689
Percent fines	0.058	0.288
Embeddedness	0.070	0.230

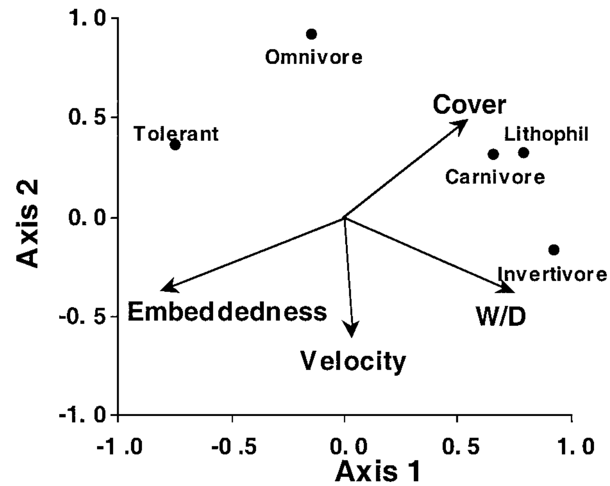

Figure 3. The first two RDA axes for warmwater fish metrics from 1997. IBI metrics are displayed as closed circles and physical habitat as arrows. W/D = width-to-depth ratio.

nificant differences in coldwater (range 5–110 of a possible 120) or warmwater (2–37 of a possible 100) IBI scores across upland land use or riparian buffer types in either 1996 or 1997.

In both years, coldwater sites were too few in number to perform RDA analysis. The Monte Carlo permutation of the RDA using warmwater fish data collected in 1996 was not significant ($P = 0.49$); therefore, no conclusions could be drawn. However, for warmwater sites in 1997, the first RDA axis explained 38% of the variance in warmwater metrics, and the second axis explained 23%. The Monte Carlo permutation indicated that the model was significant ($P = 0.01$). The first axis was associated with embeddedness, cover, and width-to-depth ratio (Figure 3). Percent carnivore, percent lithophil, and percent invertivore were negatively correlated with higher embeddedness. Percent cover was positively associated with percent lithophil and percent carnivore, whereas width-to-depth ratio was positively associated with percent invertivore and negatively associated with percent tolerant. The second axis was primarily associated with velocity, which was negatively associated with percent omnivore.

IBI scores were positively correlated with width-to-depth ratio ($P = 0.002$, Table 3) and negatively correlated with percent fines ($P = 0.058$) and embeddedness ($P = 0.070$) in 1997. Including either percent fines or embeddedness and width-to-depth ratio in a multiple regression versus IBI scores increased variance explained to 70%–71%.

Discussion

Riparian land use and management appear to be more important than upland land use in shaping instream physical habitat at local scales in the Whitewater River basin. Differences in physical habitat across buffer types can be viewed as responses within a hierarchical structure where finer-scale physical characteristics are nested within larger-scale influences (Frissell et al. 1986, Hawkins et al. 1993). While large-scale processes determine the potential range of states for nested levels, local-scale characteristics can exert considerable influence on conditions within that range. Within the constraints set at the watershed level, local-scale riparian characteristics resulted in differences in instream physical habitat in our study. Physical habitat variables that significantly related to riparian land use fell into three interrelated categories: channel characteristics, substrate, and instream cover.

During the early twentieth century, erosion rates in the region were extremely high and a large amount of alluvium was deposited in stream valleys (Waters 1977); however since 1940 upland sediment movement has been significantly reduced (Trimble and Lund 1982). In response to the reduced sediment delivery, trout have expanded into their former range after improvements in upland management practices since the 1970s (Thorn et al. 1997). Streams are now cutting into banks of the historically deposited alluvial sediment (Trimble 1993). A study in the Driftless Area, including the Whitewater River, by Trimble (1997) suggests that grass buffers would reduce sediment yields relative to wooded buffers. Trimble (1997) found that width-to-depth ratios of streams in the Driftless Area were significantly lower along grass-buffered sites than sites with a wooded buffer. Grass channels tended to stay in place, while wooded channels tended to widen over time. Differences in width-to-depth ratio were attributed to grass streambanks holding alluvial soil in place. Outside the Driftless Area, streams with grass-buffers have been shown to have narrower and deeper channels than wood-buffered streams (Peterson 1993, Sweeny 1993, Davis-Colley 1997, see also review by Lyons et al. 2000).

Channel shape may also explain differences in fine sediment in the streambed across buffer types. We found that the width-to-depth ratio was higher along wooded than along grass-buffered sites, possibly due to streambank erosion along wooded sites. As stream width increases relative to depth, the velocity of the stream is reduced and the settling velocity for fine particles may be reached (Gordon et al. 1992). In contrast, deeper and narrower channels found along grass buffers may have higher water velocities, facilitating sediment transport (Allan 1995). Although not quantified in our study, bank, bed, and wood can increase hydraulic roughness, which can result in streambeds dominated by fine particles (Buffington and Montgomery 1999).

Stream substrate composition was related to buffer width and vegetation type. Riparian buffers can reduce sedimentation into the streambed via two pathways: reducing channel erosion by increasing streambank stability or by filtering sediment from overland runoff. We found lower percent fines in relation to increased buffer width, which suggests greater filtration of sediment from overland runoff in wider buffers (Erman et al. 1977, Neibling and Alberts 1979, Dillaha et al. 1989, Osborne and Kovacic 1993). Relative to vegetation type, percent fines were lowest along grass buffers and highest along wood buffers. In our study, any buffer that was at least 150 m wide had relatively low (<50%) fine material in the streambed; however only grass buffers had <50% fines in the streambed when buffer width was <100 m (Nerbonne 1999). Grass buffer strips have been shown to be highly effective at filtering suspended sediment from agricultural runoff (e.g., Osborne and Kovacic 1993, Barling and Moore 1994, Rabeni and Smale 1995, Daniels and Gilliam 1996) and reducing streambank erosion (Trimble 1993, 1997). Grazed sites reflected intermediate fine sediment in the streambed likely due to the reduced ability of buffers to filter sediment (Barling and Moore 1994, Trimble and Mendel 1995).

Higher percent fines and exposed streambank soil found along wooded buffers may be related to the quality of buffers in the Whitewater basin. A majority of wooded buffers was dominated by boxelder, *Acer negundo*, a fast-growing, short-lived, invasive tree species. Dense growth of boxelder in wooded buffers typically shades the understory completely, and although no quantitative surveys of vegetation were conducted, only sparse understory vegetation was observed in wooded buffers. The lack of understory may limit filtration of sediment from overland runoff and could lead to higher percent fines in the stream.

Although many riparian studies have associated wooded buffers with improved water quality (Montgomery 1997), there are compelling differences between these studies and ours. Comparisons have often been made across different types of buffers, but they either did not consider grass buffers (Schlosser and Karr 1981, Peterjohn and Correl 1984) or they contrasted wooded and grass buffers only in the western United States (e.g., Beschta and Platts 1986, Erman et al. 1977), where trees provide the dominant riparian vegetation.

Although we found many differences in physical habitat across buffer types, the lack of differences in relation to upland land use was surprising. Numerous studies have shown strong influences of upland land use on physical habitat and biotic communities (Troelstrup and Perry 1989, Richards et al. 1993, Richards and Host 1994, Wang et al. 1997). Our study differs from previous work in that we considered only local (farmstead) upland land use, whereas other studies examined land use for the entire watershed. We concentrated on specific agricultural practices rather than broader categories such as forested, agricultural, and urban land use, possibly reducing the contrast between practices.

Finding strong local influence may have been due in part to our study design as well. Allan et al. (1997) found that the results of land-use studies may depend on the design of the experiment. Two studies, conducted within the same watershed, reached different conclusions: a study that examined several sites along only a few tributaries indicated that local riparian land use was important (Lammert 1995, see also Lammert and Allan 1999), whereas a study with one to two sites on several different tributaries suggested that watershed-wide land use was most important (Roth et al. 1996). Our study was similar to the former, so it is not surprising that we concluded that local riparian land use is more important.

We anticipated that differences in substrate quality across buffer types in both 1996 and 1997 would be reflected in fish and invertebrate assemblages. Contrary to our expectations, there were no differences in either RBP or IBI scores across land-use or buffer types. However, there are other examples where IBI scores did not reflect physical habitat or land-use changes (Karr et al. 1987, Shields et al. 1995, Lammert and Allen 1999). The lack of improvement in IBI scores could be due to the highly mobile nature of fish, which may explain the lack of differences in fish assemblages across buffer types in our study. Sensitive species could occupy degraded areas and utilize refuges of good habitat for critical portions of their life cycle, such as overwintering and spawning (Schlosser 1995).

Although the present study focused on local differences among sites, many factors that shape invertebrate and fish assemblages function on much broader scales. Watershed topography, surficial materials, and land use have strong control over many factors important to invertebrates and fish, such as temperature, discharge, flood frequency and magnitude, and delivery of sediment and nutrients (Troelstrup and Perry 1989). Differences in substrate among local buffer types may be less relevant to fish and invertebrate assemblages due to the overriding importance of watershed topography and surficial materials, which affect factors important to biota such as groundwater input, gradient, and substrate material. Several studies have shown differences in invertebrate (Richards et al. 1993, 1996, Richards and Host 1994, Lenat and Crawford 1994) and fish (Wang et al. 1997) assemblages, but comparisons were of land use among watersheds, not local sites. It may be that incorporation of broad-scale influences is necessary to explain variation in biotic communities.

Alternatively, the lack of differences in IBI and RBP scores across buffer types for percent fines and embeddedness may have been statistically different but not biologically different. For example, embeddedness was significantly different across buffer types, but along grass buffers, where embeddedness was lowest, embeddedness was often above 50%. When 50% of the substrate is covered, much of the interstitial space required by pollution sensitive invertebrates (Minshall 1984) and spawning fish (Muncy et al. 1979) is lost.

It is also possible that other factors besides channel characteristics, substrate, and instream cover are shaping biotic communities. Dissolved oxygen concentrations (Hilsenhoff 1982, 1987, Plafkin et al. 1989), nutrient loading (Lenat and Crawford 1994, Lemly 1982), and toxic substances such as pesticides related to broader-scale land use can all negatively impact fish and invertebrates and may explain the remaining variance in biotic communities. One or more factors may be shaping fish and invertebrate assemblages, but because many measurements were beyond the scope of this project, their impact on the biota is unknown and therefore may warrent further study.

In conclusion, this study suggests that to reduce sedimentation in agricultural areas, riparian management will show greater effectiveness at the local scale than at the upland scale. Although upland land use was not a significant factor in this study, the expansion of trout to their former range after improvements in upland management practices (Thorn et al. 1997) indicates the importance of maintaining and perhaps expanding upland BMPs. The ability of grass buffers to maintain streambank stability and low sediment content in stream substrates suggests they may be a viable riparian management option to add a diversity of habitats for fish and invertebrates (Lyons et al. 2000).

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Literature Cited

Allan, J. D. 1995. Stream Ecology: The structure and function of running waters. Chapman and Hall, London.

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. *Freshwater Biology* 37:149–161.

Amemiya, M. 1970. Land and water management for minimizing sediment. Pages 35–45 *in* Willrich, T. L. and G. E. Smith (ed.), Agricultural Practices and Water Quality. Iowa State Press, Ames.

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

Berkman, H. E., and C. F. Rabeni. 1987. Effect of siltation on stream fish communities. *Environmental Biology of Fishes* 18: 285–294.

Beschta, R. L., and W. S. Platts. 1986. Morphological features of small streams: Significance and function. *Water Resources Bulletin* 22:369–380.

Bingham, C., and G. W. Oehlert. 1998. MACANOVA: A program for statistical analysis and matrix algebra. Department of Applied Statistics, University of Minnesota.

Blevins. R. L., W. W. Frye, P. L. Baldwin, and S. D. Robertson. 1990. Tillage effects on sediment and soluble nutrient losses from a Maury silt loam soil. *Journal of Environmental Quality* 19:683–686.

Bovee, K. D., and T. Cochnauer. 1977. Development and evaluation of weighted criteria, probability-of-use curves for instream flow assessments: Fisheries. US Fish and Wildlife Service Instream Flow Information Paper No.3. FWS/OBS-77/63.

Broussard, W. L., D. F. Farrell, H. W. Anderson, Jr., and P. E.

Felsheim. 1975. Water resources of the Root River watershed, southeastern Minnesota. Hydraulic Investigations Atlas HA-548. Department of Interior, US Geologic Survey, St. Paul, Minnesota.

Buffington, J. M., and D. R. Montgomery. 1999. Effects of hydraulic roughness on surface textures of gravel-bed rivers. *Water Resources Research* 35:3507–3521.

Cairns, J., and J. R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. Pages 10–27 *in* D. M. Rosenberg and V. H. Resh (ed.), Freshwater biomonitoring and benthic macroinvertebrates. Routledge, Chapman, and Hall, New York.

Chapman, D. W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. *Transactions of the American Fisheries Society* 117:1–21.

Cooper, C. M. 1987. Benthos in Bear Creek, Mississippi: effects of habitat variation and agricultural sediments. *Journal of Freshwater Ecology* 4:101–113.

Cordone, A. J., and D. W. Kelly. 1961. The influences of inorganic sediment on the aquatic life of streams. *California Fish and Game* 47:189–228.

Daniels, R. B., and J. W. Gilliam. 1996. Sediment and chemical load reduction by grass and riparian filters. *Soil Science Society of America Journal* 60:246–251.

Davis, W. S. 1995. Biological assessment and criteria: building on the past. Pages 15–29 *in* W. S. Davis and T. P. Simon (ed.), Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, Florida.

Davis-Colley, R. J. 1997. Stream channels are narrower in pasture than in forest. *New Zealand Journal of Marine and Freshwater Research* 31:599–608.

Dillaha, T. R., R. B. Reneau, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. *Transactions of the Society of Agricultural Engineers* 32:513–519.

Erman, D. C., J. D. Newbold, and K. B. Roby. 1977. Evaluation of streamside buffer strips for protecting aquatic organisms. University of California, Davis, Water Resources Center Contribution 186.

Fitch, L., and B. W. Adams. 1998. Can cows and fish coexist? *Canadian Journal of Plant Science* 78:191–198.

Frissell, C. A., W. J. Liss, C. E. Warren, and M. D. Hurley. 1986. A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. *Environmental Management* 10:199–214.

Gaynor, J. D., and W. I. Findlay. 1995. Soil and phosphorus loss from conservation and conventional tillage in corn production. *Journal of Environmental Quality* 24:734–741.

Gordon, N. D., T. A. McMahon, and B. L. Finlayson. 1992. Stream hydrology: An introduction for ecologists. John Wiley & Sons, Chichester, UK.

Hawkins, C. P., J. L. Kershner, P. A. Bisson, M. D. Bryant, and L. M. Decker. 1993. A hierarchical approach to classifying stream habitat features. *Fisheries* 18(6):3–12.

Hendicks, M. L., C. H. Hocutt, and J. R. Stauffer, Jr. 1980. Monitoring of fish in lotic habitats. Pages 205–231 *in* C. H. Hocutt and J. R. Stauffer, Jr. (ed.), Biological monitoring of fish. Lexington Books, D. C. Heath and Co., Lexington, Massachusetts.

Hilsenhoff, W. L. 1982. Using a biotic index to evaluate water quality in streams. Technical Bulletin No. 132. Wisconsin Department of Natural Resources, Madison, Wisconsin.

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

Johnson, H. P., and W. C. Moldenhauer. 1970. Pollution by sediment: Sources and the detactment processes. Pages 3–20 *in* T. L. Wilirich and G. E. Smith (ed.), Agricultural Practices and Water Quality. Iowa State Press, Ames.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6):21–27.

Karr, J. R. 1987. Biological monitoring and environmental assessment: A conceptual framework. *Environmental Management* 11:249–256.

Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. Assessing biological integrity in running waters: A method and its rationale. Illinois Natural History Survey Special Publication 5.

Karr, J. R., P. R. Yant, and K. D. Fausch. 1987. Spatial and temporal variability of the index of biotic integrity in three Midwestern streams. *Transactions of the American Fisheries Society* 116:1–11.

Kauffman, J. B., and W. C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications—a review. *Journal of Range Management* 37:430–438.

Lammert, M. 1995. Assessing land use and habitat effects of fish and macroinvertebrate assemblages: Stream biotic integrity in an agricultural watershed. M S thesis. University of Michigan. Ann Arbor, Michigan.

Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: Effects of scale in measuring the influence of land use/cover and habitat structure of fish and macroinvertebrates. *Environmental Management* 23:257–270.

Lemly, A. D. 1982. Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. *Hydrobiotogia* 87:222–245.

Lenat, D. R., and J. K. Crawford. 1994. Effects of land use on water quality and aquatic biota of three North Carolina piedmont streams. *Hydrobiologia* 294:185–199.

Lenat, D. R., D. L. Penrose, and K. W. Eagleson. 1979. Biological evaluation of non-point source pollutants in North Carolina streams and rivers. Biological Series 102. North Carolina Department of Natural Resources and Community Development, Raleigh.

Lenat, D. R., D. L. Penrose, and K. W. Eagleson. 1981. Variable effects on sediment addition on stream benthos. *Hydrobiologia* 79:187–194.

Lyons, J. 1992a. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. *North American Journal of Fisheries Management* 12:198–203.

Lyons, J. 1992b. Using the index of biotic integrity to measure environmental quality in warmwater streams of Wisconsin. North Central Forest Experiment Station General Technical Report NC-149. US Department of Agriculture, Forest Service, St. Paul, Minnesota. 51 pp.

Lyons, J., S. W. Trimble, and L. K. Paine. 2000. Grass versus trees: managing riparian areas to benefit streams of central North America. *Journal of the American Water Resources Association* 36:919–930.

Marcuson, P. E. 1977. The effect of cattle grazing on brown trout in Rock Creek, Montana. Montana Department of Fish and Game Fisheries Division Special Report Project No. F-20-R-21, II-a.

Merritt, R. W., and K. W. Cummins. 1996. An introduction to the aquatic insects of North America. Kendall/Hunt Publishing, Dubuque, Iowa.

Minshall, G. W. 1984. Aquatic insect-substratum relationships. Pages 356–400 *in* V. H. Resh and D. M. Rosenberg (ed.), The ecology of aquatic insects. Praeger, New York.

Minnesota Pollution Control Agency. 1996. 319 water quality monitoring in the Whitewater River watershed project. St. Paul, Minnesota.

Montgomery, D. R. 1997. What's best on the banks? *Nature* 388:328–329.

Muncy, R. J., G. J. Atchison, R. V. Bulkley, B. W. Menzel, L. G. Perry, and R. C. Summerfelt. 1979. Effects of suspended solids and sediment on reproduction and early life of warmwater fishes: A review. US Environmental Protection Agency, Washington, DC. EPA Report 600/3-79-042.

Mundahl, N. D., and T. P. Simon. 1999. Development and application of an index of biotic integrity for coldwater streams of the upper midwestern United States. Pages 383–415 *in* T. P. Simon (ed.), Assessing the sustainability and biological integrity of water resource quality using fish assemblages. Lewis Publishers, Boca Raton, Florida.

Neibling, W. H., and E. E. Alberts. 1979. Composition and yields of soil particles transported through sod strips. Paper No. 79-2065. American Society of Agricultural Engineers, St. Joseph, Michigan.

Nerbonne, B. A. 1999. Effects of land use and sediment on the distribution of benthic invertebrates and fish in the Whitewater River Watershed of Minnesota. M S thesis. University of Minnesota. St. Paul, Minnesota.

Ornernik, J. M., and A. L. Gallant. 1988. Ecoregions of the upper Midwest states. US Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon. EPAI600/3-88-037.

Osborne, L. L., and D. A. Kovacic. 1993. Riparian vegetated buffer strips in water quality restoration and stream management. *Freshwater Biology* 29:243–258.

Peterjohn, W. T., and D. L. Correl. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology* 65:1466–1475.

Petersen, R. C., L. B. M. Pertersen, and J. Lacoursiere. 1992. A building block model for stream restoration. Pages 293–309 *in* P. J. Boon, P. Calow, and G. E. Petts (ed.), River conservation and management. John Wiley & Sons, New York.

Peterson, A. M. 1993. Effects of electric transmission rights-of way on trout in forested headwater streams in New York. *North American Journal of Fisheries Management* 13:581–585.

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. US Environmental Protection Agency, Washington, DC. EPA-444/4-89-001.

Platts, W. S., and W. F. Megahan. 1975. Time trends in riverbed sediment composition in salmon and steelhead spawning areas: South Fork Salmon River, Idaho. *Transactions of the North American Wildlife Conference* 40:229–239.

Platts, W. S., W. F. Megahan, and G. W. Minshall, 1983. Methods for evaluating stream, riparian, and biotic conditions. USDA, Forest Service, Ogden, Utah. Intermountain Forest and Range Experiment Station General Technical Report INT-138.

Platts, W. S., C. Armour, G. D. Booth, M. Bryant, J. L. Bufford, P. Culpin, S. Jensen, G. W. Lienkaemper, G. W. Minshall, S. B. Monsen, R. L. Nelson, J. R. Sedell, and J. S. Tuhy. 1987. Methods for evaluating riparian habitats with applications to management. USDA, Forest Service, Ogden, Utah. Intermountain Research Station General Technical Report INT-221.

Rabeni, C. F., and M. A. Smale. 1995. Effects of siltation on stream fishes and the potential mitigating role of the buffering riparian zone. *Hydrobiologia* 303:211–219.

Razavian, D. 1990. Hydrologic response of an agricultural watershed to various hydrologic and management conditions. *Water Resources Bulletin* 26:777–785.

Richards, C., and G. Host. 1994. Examing land use influences on stream habitats and macroinvertebrates: A GIS approach. *Water Resources Bulletin* 30:729–738.

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.

Richards, C., L. Johnson, and G. E. Host. 1996. Landscape-scale influences on stream habitats and biota. *Canadian Journal of Fisheries Aquatic Science* 53(Suppl. 1):295–311.

Roth, N. E., J. D. Allan, and D. L. Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology* 11:141–156.

Schlosser, I. J. 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. *Hydrobiologia* 303:71–81.

Schlosser, I. J., and J. R. Karr. 1981. Riparian vegetation and channel morphology impact on spatial patterns of water quality in agricultural watersheds. *Environmental Management* 5:233–243.

Shields, F. D., Jr., S. S. Knight, and C. M. Cooper. 1995. Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams. *Hydrobiologia* 312:191–208.

Simonson, T. D., J. Lyons, and P. D. Kanehl. 1994. Guidelines for evaluating fish habitat in Wisconsin streams. USDA, Forest Service. St. Paul, Minnesota. North Central Experiment Station General Technical Report NC-164, 36 pp.

Smith, G. R. 1989. Key to the non-game fishes of Michigan. University of Michigan, Ann Arbor, 52 pp.

Sovell, L. A., and B. Vondracek. 1999. Evaluation of the fixed count method for rapid bioassessment protocol III with benthic macroinvertebrate metrics. *Journal of the North American Benthological Society* 18:420–426.

Strand, M., and R. W. Merritt. 1999. Impacts of livestock grazing activities on stream insect communities and the riverine environment. *American Entomologist* 45:13–29.

Sweeny, 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 Philadelphia* 144:291–340.

ter Braak, C. J. F. 1987–1992. CANOCO—a FORTRAN program for canonical community ordianation. Microcomputer Power, Ithaca, New York.

Thorn, W. C., C. S. Anderson, W. E. Lorenzen, D. L. Hendickson, 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. 1993. The distributed sediment budget model and watershed management in the Paleozoic Plateau of the Upper Midwestern United States. *Physical Geography* 14:285–303.

Trimble, S. W. 1997. Stream channel erosion and change resulting from riparian forests. *Geology* 25:467–469.

Trimble, S. W. 1999. Decreased rates of alluvial sediment storage in the Coon Creek basin, Wisconsin 1975–93. *Science* 285:1244–1246.

Trimble, S. W., and S. W. Lund. 1982. Soil conservation and the reduction of erosion and sedimentation in the Coon Creek Basin, Wisconsin. Geological Survey Professional Paper 1234.

Trimble, S. W., and A. C. Mendel. 1995. The cow as a geomorphic agent: A critical review. *Geomorphology* 13:233–253.

Troelstrup, N. H., and J. A. Perry. 1989. Water quality in southeastern Minnesota streams: Observations along a gradient of land use and geology. *Journal of the Minnesota Academy of Science* 55:6–31.

US Environmental Protection Agency. 1996. National water quality inventory of 1996. US EPA, Washington, DC.

Wang, L., J. Lyons, P. Kanehl, and R. Gatti. 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. *Fisheries* 22(6):6–12.

Waters, T. F. 1977. The streams and rivers of Minnesota. University of Minnesota Press, Minneapolis, 373 pp.

Waters, T. F. 1995. Sediment in streams. American Fisheries Society Monograph 7. American Fisheries Society, Bethesda, Maryland, 251 pp.

Wilkin, D. C., and S. J. Hebel. 1982. Erosion, redeposition, and delivery of sediment to Midwestern streams. *Water Resources Research* 18:1278–1282.