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Impacts of Rotational Grazing and Riparian Buffers on Physicochemical and Biological Characteristics of Southeastern Minnesota, USA, Streams

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ABSTRACT / We assessed the relationship between riparian management and stream quality along five southeastern Minnesota streams in 1995 and 1996. Specifically, we examined the effect of rotationally and continuously grazed pastures and different types of riparian buffer strips on water chemistry, physical habitat, benthic macroinvertebrates, and fish as indicators of stream quality. We collected data at 17 sites under different combinations of grazing and riparian management, using a longitudinal design on three streams and a paired watershed design on two others. Continuous and rotational grazing were compared along one longitudinal study stream and at the paired watershed. Riparian buffer management, fenced trees (wood buffer), fenced grass, and unfenced rotationally grazed areas were the focus along the two remaining longitudinal streams. Principal components analysis (PCA) of water chemistry, physical habitat, and biotic data indicated a local management effect. The ordinations separated continuous grazing from sites with rotational grazing and sites with wood buffers from those with grass buffers or rotationally grazed areas. Fecal coliform and turbidity were consistently higher at continuously grazed than rotationally grazed sites. Percent fines in the streambed were significantly higher at sites with wood buffers than grass and rotationally grazed areas, and canopy cover was similar at sites with wood and grass buffers. Benthic macroinvertebrate metrics were significant but were not consistent across grazing and riparian buffer management types. Fish density and abundance were related to riparian buffer type, rather than grazing practices. Our study has potentially important implications for stream restoration programs in the midwestern United States. Our comparisons suggest further consideration and study of a combination of grass and wood riparian buffer strips as midwestern stream management options, rather than universally installing wood buffers in every instance.

Nonpoint source pollution associated with agricultural production is a major threat to water quality in the United States (Lovejoy and others 1997). In Minnesota, nonpoint source pollution was identified as the most serious water-quality problem in the state. Of the 12,241 river miles assessed by the Minnesota Nonpoint Source Management Program from 1983 to 1993, 58% were determined to be impaired for at least one designated use category (e.g., swimmable, fishable, or drinkable). Surveys of natural resource professionals indicate that 90% of the river miles evaluated were impaired (or threatened with impairment) by agriculture.

According to Clark (1998), evidence is building that indicates that unrestricted livestock access to streams and rivers accounts for a relatively modest amount of nonpoint source pollution in humid temperate regions. To examine the impacts of different grazing and riparian management practices on stream water quality, we participated in a larger study in which farmers and researchers collaborated to monitor the ecological, financial, and social aspects of rotational grazing in southeastern Minnesota. Farmers wanted to know if recently adopted changes in their grazing practices and riparian management were effecting stream quality. Participating farmers had recently converted from row

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crop agriculture to rotational grazing. Rotational grazing, as defined in this study, is a method of farming in which a pasture is partitioned into small 2- to 3-acre paddocks. Livestock are allowed to graze a paddock for one to three days, before being rotated to another paddock. The original paddock is then rested for an extended period, often 30 days or longer. Under this grazing strategy, animals spend shorter periods of time in or near streams, allowing heavier growth of riparian vegetation and reduced inputs of waste.

The effects of grazing on stream quality have been extensively studied, primarily in the western United States. Cattle have been implicated in degradation of streambank soils and vegetation, which affect channel morphology, water chemistry, and fish and aquatic insect habitat (Kauffman and Krueger 1984, Trimble and Mendel 1995, Fitch and Adams 1998, Strand and Merritt 1999, Belsky and others 1999). Cattle manure and urine in or along a stream can result in elevated phosphorus and nitrogen levels (Lemly 1982), increased fecal coliform counts (Tiedemann and others 1987, 1988), and a decline in dissolved oxygen concentrations (Fleischner 1994, Harris and others 1994). Heavy grazing along streambanks can lead to increased sediment deposition in the streambed (Winegar 1977). Trampling of streambanks can eliminate overhanging vegetation, increasing the potential for increased water temperatures (Kauffman and Krueger 1984, Wohl and Carline 1996).

While the concepts learned in the western United States may apply to the midwest, relatively few studies have examined and compared effects of different types of riparian management strategies in this region (Osborne and Kovacic 1993, Peterjohn and Correll 1984, Rabeni and Smale 1995, Schlosser and Karr 1981). The authors are not aware of published studies that have examined the effects of rotational grazing on stream quality in the midwestern United States. Information gained in the western United States may not be readily transferable to the Midwest because of differences in management objectives, vegetation, geomorphology, and climate.

Historical accounts suggest that many midwestern streams were bordered by a mix of grass and forest riparian areas imbedded in rolling prairie or savanna with loess soils in both upstream and downstream reaches (US Surveyor General 1847–1908). These historical accounts of midwestern streams conform with Wiley and others' (1990) contemporary observations of the spatial structure and function of midwestern streams. Accordingly, headwater riparian vegetation is dominated by grasses, production-to-respiration ratios decrease downstream, and water temperature reaches equilibrium with air temperature, with little variation downstream. In contrast, the spatial and functional configuration of western streams conform with the river continuum concept developed by Vannote and others (1980). In western stream systems, headwaters are more steeply graded and, unlike midwestern streams, are dominated by wooded vegetation.

Intensive agriculture in the Midwest from the late 19th to the early part of the 20th century increased sediment movement in stream channels in the Western Corn Belt Plains Ecoregion (Trimble 1993, Waters 1977) and from tributaries to upper main valleys in the Driftless Area Ecoregion (Omernik and Gallant 1988). Sediment deposition from tributaries to upper main valleys often led to aggradation of streambanks followed by erosion, increasing the downstream sediment yield. Soil conservation practices, which emerged in the 1930s, often excluded cattle from and introduced trees to riparian areas to stabilize streambanks. Fire was eliminated from these riparian areas, which are now often dominated by a dense canopy of early successional tree species such as box elder (*Acer negundo*). These densely wooded riparian areas prevent the establishment of ground cover and display accelerated erosion.

This study, addresses two specific questions about the influence of land use on stream quality: Is there a significant difference between the impacts of rotationally and continuously grazed pastures? Is there a significant difference between rotational grazing and different types of riparian buffer strips, e.g., grass or wooded? To address these two questions, we investigated several indicators of stream quality, including water chemistry, bank and channel physical habitat, and the composition of macroinvertebrate and fish communities. Increased runoff (Rauzi and Hanson 1966, Sartz and Tolsted 1974), sediment loading (Winegar 1977, Waters 1995), and an increase in width-to-depth ratios (Marcuson 1977, Platts 1979) are examples of well documented impacts of livestock grazing that affect stream chemical and physical habitat. The use of fish and benthic macroinvertebrate indicators for stream monitoring is more recent than water chemistry monitoring, but is now widely applied (e.g., Ohio Environmental Protection Agency 1987). Benthic macroinvertebrates and fish are indirectly affected by land management practices through impacts to water chemistry and physical habitat. Benthic macroinvertebrate indices characterize the water quality of stream segments based on the community composition and tolerance (or intolerance) of macroinvertebrate taxa to organic pollution and habitat degradation (Hilsenhoff 1977, 1982, Plafkin and others 1989). Similarly, fish community

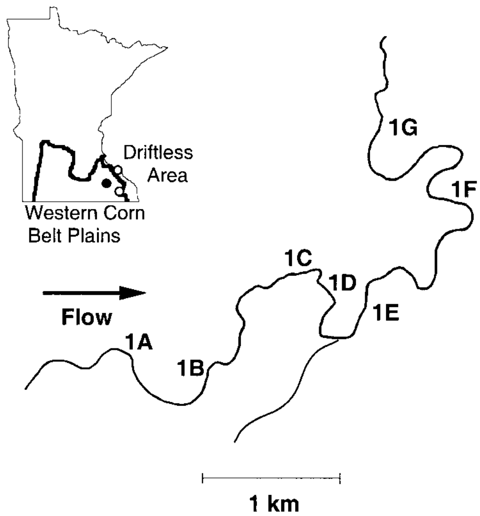

Figure 1. Location of sampling sites for stream 1 (closed circle on inset). Sites A, B, E, F, and G: rotational grazing; C and D: continuous grazing.

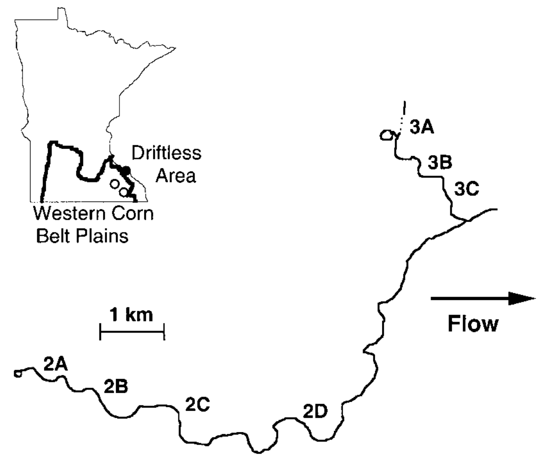

Figure 2. Location of sampling sites for streams 2 and 3 (closed circle on inset). Sites 2A and 2B: grass buffer/rotational grazing in 1995, rotational grazing to stream edge in 1996; 2C: rotational grazing to stream edge; 2D: wood buffer; 3A: rotational grazing; 3B: grass buffer/rotational grazing; and 3C: wood buffer.

structure and composition are used as indicators of stream water quality (Karr 1981).

Methods

Study Locations

Study sites were established on farms located on five streams in southeastern Minnesota. A longitudinal (upstream/downstream) study design was established on three streams, and a paired watershed design was used on two streams. Streams were located in two ecoregions, which reflect differences in geology, hydrology, and vegetation (Omernik and Gallant 1988). Our longitudinal design included one stream (Figure 1) that drains a broad glaciated plain in southern Minnesota, within the Western Corn Belt Plains. The two remaining streams with a longitudinal design were located in the same watershed (Figure 2) within the unglaciated Driftless Area, adjacent to the Western Corn Belt Plains. The two adjacent streams used for the paired design were within the Western Corn Belt Plains (Figure 3). Land use in the watersheds is dominated by row crop agriculture; however, grassland and deciduous forest were also present (Table 1). The riparian zones are a mix of wooded, grazed, or grass buffers.

Treatments (rotational grazing, grass and wood buffers) were not randomly selected, rather they were dependent on the land management practices of farmers participating in the larger project. Reference sites were chosen on nearby farms based on similar soil type and proximity to participating farms. Sites within treatments and references were selected to include similar geomorphic units. At both rotationally and continuously grazed sites, cows grazed to the stream edge and had access to streams. Grass and wood buffer sites were fenced to exclude cows.

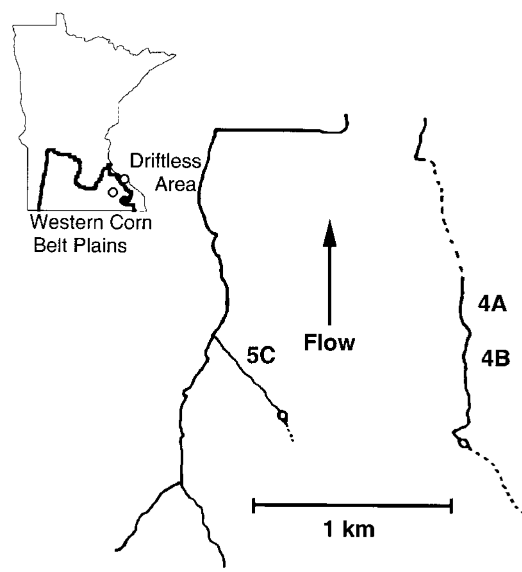

Figure 3. Location of sampling sites for streams 4 and 5 (closed circle on inset). A: Wood buffer, B: rotational grazing, and C: continuous grazing. Note: the distance between streams is not to scale; streams are 2 km apart, but in adjacent watersheds.

Table 1. Description of study sites, streams and watersheds

Stream	Study design	Stream order	Treatment/ management	Ecoregion/ county	Land use	Sites (*N*)
1	Longitudinal	3rd–4th	Grazing/(rotational vs continuous)	Western Corn Belt Plain/Dodge	90% row crop	7
2	Longitudinal	3rd	Riparian/(rotational vs grass vs wood)	Driftless Area/Wabasha	47% row crop 39% wooded 13% grassland	4
3	Longitudinal	3rd	As for stream 2	As for stream 2	As for stream 2	3
4	Paired	1st	Grazing/(rotational vs continuous) and (Riparian wood vs rotational)	Western Corn Belt Plain/Winona	46% row crop 34% grassland 18% wooded	2
5	Paired	1st	Grazing/(rotational vs continuous)	Western Corn Belt Plain/Olmsted	66% row crop 26% grassland	1

Data Collection

Water chemistry. Water chemistry was sampled monthly at each site from May through September 1995. Dissolved oxygen, pH, conductivity, and temperature were measured in the field using a YSI model 54 oxygen meter, an Orion model SA250 pH meter, and a YSI model 53 conductivity and temperature meter, respectively. Water samples were collected, stored on ice, and analyzed in the lab (HACH DR/2000 photometer) for nitrate, sulfate, orthophosphate, ammonia, and chloride. Turbidity was measured in the lab with a HACH 2100A Turbidimeter. Organic carbon and fecal coliform samples were collected in acid washed or autoclave sterilized containers, stored on ice, and cultured for analysis within 30 h of collection at a state-certified laboratory.

Physical habitat. Physical habitat measurements were taken along 10–13 transects at each site during July 1995 and 1996 (Table 2), following the protocol of Simonson and others (1994). Depth, velocity, substrate composition, and substrate embeddedness were recorded at four or five points along each transect (Simonson and others 1994). Depth (meters) was measured using a top-setting wading rod. Velocity (meters per second) was measured at the 0.6 depth using a Marsh-McBirney 2000 flowmeter. A modified Wentworth scale was used to visually assign substrate composition (to the nearest 5%) within a 30 × 30-cm area at each transect point. Fines were defined as particles <6.4 mm. A substrate embeddedness rating (0–100% to the nearest 20%) was assigned at each transect point where gravel, cobble, or boulders were present (Simonson and others 1994). Width-to-depth ratios were calculated using the average depth and width for each transect. Canopy cover was visually estimated to the nearest 5% at each transect. The percentage of exposed streambank soil was calculated by measuring both the exposed soil and total streambank lengths.

Benthic macroinvertebrates. We used a modified Hess sampler to take three benthic macroinvertebrate samples from a riffle at each site in June and September 1995 and June 1996. Samples were preserved in Kahles solution for 24–48 h, rinsed, and stored in 80% ethyl alcohol. A sample was sorted by evenly spreading its contents in a gridded plastic pan with water and picking 100 organisms randomly from a minimum of three separate grids (Hilsenhoff 1982). Macroinvertebrates were identified to family (Hilsenhoff 1988). A subset of Rapid Bioassessment Protocol (RBP) metrics were used to characterize water quality at each site (Plafkin and others 1989). Metrics included Hilsenhoff's Family Biotic Index (FBI), total number of families present (total taxa); total number of families within the groups Ephemeroptera, Plecoptera, and Trichoptera (EPT index); percent contribution of the dominant family to the total number of organisms (percent dominance); and ratio of EPT to Chironomidae (EPT/C). For all metrics except FBI, a higher value indicates better water quality.

Fish. Fish were sampled at each site by electrofishing in late July or August (Lyons 1992), except on stream 1 where only five of seven sites were sampled. We recorded the number of species and abundance of each, and calculated fish densities (number $\cdot$ m^{-2}) for each site.

Data Analysis

A principal components analysis (PCA) was used to identify patterns in water chemistry, physical habitat, and benthic macroinvertebrates across streams, years, and riparian use. Only variables with principal compo-

Table 2. Mean physical habitat values (±2 SD) for each site in 1995 and 1996

Site	Width:depth ratio	Bank angle (°)	Exposed bank (%)	Canopy cover (%)	Embed. (%)	Fines (%)	Velocity (m/sec)
1995							
1A	53 (46)	152 (24)	31 (64)	37 (82)	80 (50)	30 (50)	0.08 (0.18)
1B	47 (46)	146 (32)	56 (70)	13 (56)	68 (66)	37 (62)	0.08 (0.26)
1C	43 (40)	153 (50)	36 (76)	12 (58)	58 (74)	41 (70)	0.04 (0.28)
1D	66 (64)	163 (32)	62 (84)	22 (70)	73 (58)	62 (56)	0.08 (0.18)
1E	60 (64)	143 (30)	47 (60)	45 (84)	72 (46)	44 (52)	0.11 (0.16)
1F	67 (190)	148 (32)	50 (54)	35 (72)	39 (56)	31 (44)	0.16 (0.26)
1G	52 (72)	144 (34)	49 (68)	24 (70)	41 (74)	28 (46)	0.20 (0.44)
2A	8 (10)	139 (44)	1 (10)	45 (46)	68 (64)	45 (48)	0.17 (0.32)
2B	14 (20)	140 (36)	11 (52)	49 (76)	56 (86)	50 (50)	0.02 (0.36)
2C	11 (14)	153 (14)	23 (72)	8 (44)	76 (46)	53 (58)	0.09 (0.56)
2D	33 (40)	142 (40)	29 (48)	33 (88)	74 (48)	77 (44)	0.21 (0.42)
3A	10 (10)	133 (66)	59 (74)	97 (14)	83 (56)	55 (52)	0.06 (0.14)
3B	7 (8)	129 (56)	7 (30)	82 (56)	58 (68)	70 (54)	0.11 (0.16)
3C	28 (38)	143 (50)	87 (38)	100 (0)	100 (0)	90 (32)	0.17 (0.18)
4A	22 (22)	134 (26)	96 (20)	100 (0)	81 (48)	57 (54)	0.05 (0.10)
4B	9 (6)	150 (28)	2 (8)	98 (12)	68 (40)	42 (38)	0.08 (0.14)
5C	15 (0)	129 (68)	38 (68)	0 (0)	79 (46)	35 (48)	0.10 (0.20)
1996							
1A	13 (10)	156 (42)	40 (40)	12 (52)	44 (16)	27 (12)	0.05 (0.20)
1B	15 (18)	148 (44)	46 (68)	20 (72)	59 (22)	52 (18)	0.07 (0.30)
1C	16 (28)	162 (26)	43 (56)	8 (56)	75 (16)	41 (16)	0.06 (0.18)
1D	17 (20)	162 (18)	47 (86)	28 (58)	85 (14)	58 (14)	0.06 (0.18)
1E	21 (18)	151 (18)	54 (64)	36 (48)	72 (14)	36 (14)	0.07 (0.10)
1F	22 (28)	160 (10)	23 (50)	33 (54)	49 (16)	16 (8)	0.15 (0.24)
1G	20 (24)	142 (34)	47 (58)	33 (54)	24 (14)	15 (10)	0.15 (0.28)
2A	3 (1)	156 (20)	5 (28)	18 (36)	70 (67)	52 (51)	0.19 (0.70)
2B	5 (9)	149 (27)	22 (63)	37 (96)	77 (60)	60 (58)	0.12 (0.33)
2C	6 (6)	145 (34)	14 (50)	8 (29)	68 (68)	63 (69)	0.09 (0.25)
2D	14 (5)	155 (22)	24 (65)	50 (69)	70 (48)	80 (35)	0.20 (0.36)
3A	3 (2)	169 (8)	12 (39)	37 (51)	63 (69)	58 (37)	0.08 (0.19)
3B	3 (4)	139 (60)	29 (47)	49 (52)	75 (65)	61 (48)	0.14 (0.19)
3C	9 (6)	146 (39)	64 (68)	98 (6)	100 (0)	87 (40)	0.14 (0.19)
4A	7 (11)	137 (31)	95 (15)	89 (41)	78 (50)	53 (35)	0.05 (0.11)
4B	4 (3)	162 (16)	10 (44)	100 (0)	5 (22)	17 (17)	0.04 (0.12)
5C	6 (4)	155 (35)	50 (67)	41 (66)	60 (66)	28 (40)	0.04 (0.13)

nent loadings greater than positive or negative 0.30 were considered important in the ordinations.

A Kruskal-Wallis test (Kruskal and Wallis 1952), the nonparametric analog of an analysis of variance, was used to determine if physical habitat measures and macroinvertebrate metrics identified as important in the PCA differed within streams. Where significant differences were identified ($P < 0.10$) with the Kruskal-Wallis tests, Dunn's multiple comparisons test (Dunn 1964) was used to compare pairs of sites within each stream. Summaries of water chemistry data, fish density and fish numbers are presented.

Results

Principal components analysis distinguished water chemistry, physical habitat and invertebrate metrics among sites. Continuous grazing sites differed from rotational grazing, and sites with wood buffers differed from sites with grass buffers or rotationally grazed areas. These differences indicate a local riparian effect on water quality. PCA analysis also identified a year effect for physical habitat and macroinvertebrate metrics. The year effect may result from differences in precipitation between years; stream discharge was lower in 1996 than in 1995.

Water Chemistry

Dissolved oxygen concentrations were consistently above the state water quality standard of 5 mg/liter, and mean nitrate concentrations were consistently below the 10 mg N/liter drinking water standard set by the US Environmental Protection Agency at all sites (Sovell 1997). Only fecal coliform and turbidity dem-

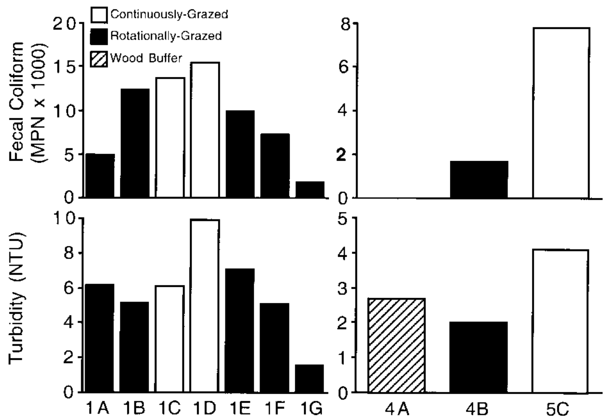

Figure 4. Fecal coliform (upper panels) and turbidity (lower panels) at each site for streams 1, 4, and 5 in 1995. Note differences in scale on *y* axes. Fecal coliform is reported as most probable number (MPN).

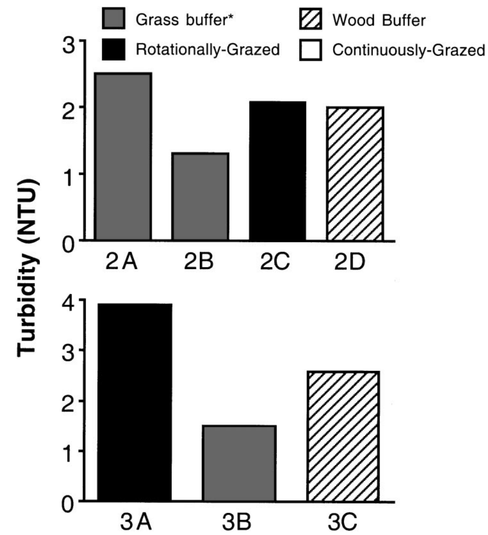

Figure 5. Mean turbidity for streams 2 and 3 at each site in 1995. Note differences in scale on *y* axes. Stations 2A and 2B rotationally grazed to waters edge in 1996.

onstrated consistent differences among land-use and buffer conditions and are the only water chemistry variables discussed further.

Rotational vs continuous grazing. Mean values for fecal coliform and turbidity were higher at continuously grazed sites than at rotationally grazed sites in streams 1 and 4 (Figure 4). In stream 1, both fecal coliform and turbidity increased between the upstream and downstream continuously grazed sites.

Buffer comparisons. Mean turbidity was generally lower along grass buffered sites than for wooded sites. Mean turbidity was lowest at the downstream grass buffer site in stream 2, intermediate at the wood (2D) and rotationally grazed site (2C), and highest at the upstream grass buffer site (2A), which was likely influenced by upstream practices. Turbidity dropped dramatically along the grass buffer sites (2A and 2B) in 1995 (Figure 5). Along stream 3, monthly turbidity was lowest at the grass buffer site 3B, intermediate at the wood buffer site 3C, and highest at the upstream rotationally grazed site 3A (Figure 5).

Physical Habitat

Differences among sites within streams were detected for 20 of 28 Kruskal-Wallis tests for physical habitat variables measured in 1995, and for 22 of 28 tests in 1996 (Table 3). Percent fines were significantly different within each stream during 1995 and 1996 (Table 3). Velocity (streams 1–3) and width-to-depth ratios (streams 2–4) were also significantly different for both years and are discussed for appropriate comparisons below.

Rotational vs continuous grazing. Generally, percentage of fines was highest at continuously grazed sites. In both years, percentage of fines was consistently high at the downstream continuously grazed site (1D) in relation to all other sites (Figure 6). In streams 4 and 5, the continuously grazed site (5C) had the highest percentage of fines in 1996, but was similar to the rotationally grazed site in 1995. Percentage of exposed streambank soil was significantly higher at the continuously grazed site (5C) than at the rotationally grazed site (4B) for both 1995 and 1996 (Figure 7).

Buffer comparison. Percentage of fines was generally significantly higher at wood buffer sites than for grass buffer or rotationally grazed sites. In 1995 and 1996 for streams 2 and 3, percentage of fines at the wood buffer sites (2D and 3C) was higher than other sites (Figure 8). Note also that the percentage of fines was consistently high at the wood buffer site (4A, Figure 6).

Percentage of canopy cover was generally similar for grass and wood-buffers. In 1995, canopy cover was significantly lower at the rotationally grazed site (2C) than at the grass and wood buffer sites in stream 2. Grass buffers along our sites formed an almost complete canopy, comparable to wood buffered sites, even on streams up to 4 m wide, such as site 2B (Figure 9). However, in 1996 when fenced grass buffers were removed, there was no significant difference between sites 2A and 2C (Figure 10). For streams 4 and 5, canopy cover was significantly lower at the continuously grazed site than at the rotationally grazed and wood buffer sites in both years (Figure 10).

Table 3. *P* values based on Kruskal-Wallis tests for physical parameters for each system in 1995 and 1996[a]

	System							
	1995				1996			
Physical parameter	1	2	3	4	1	2	3	4
	—	—	—	—				
Velocity (cm/sec)	<0.001	<0.001	<0.001	—	<0.001	<0.001	0.016	
Embeddedness (%)	<0.001		0.002	—	<0.001			0.005
Fines (%)	<0.001	<0.001	<0.001	0.073	<0.001	<0.001	<0.001	<0.001
Bank angle	0.028			0.058	<0.001		<0.001	0.002
Canopy cover (%)		0.001	0.040	<0.001	<0.001	0.054	0.002	<0.001
Exposed streambank (%)		0.010	<0.001	<0.001	0.005		0.010	<0.001
Width-to-depth ratio		<0.001	<0.001	<0.001		<0.001	<0.001	0.056

[a]Blank cells indicate $P > 0.10$.

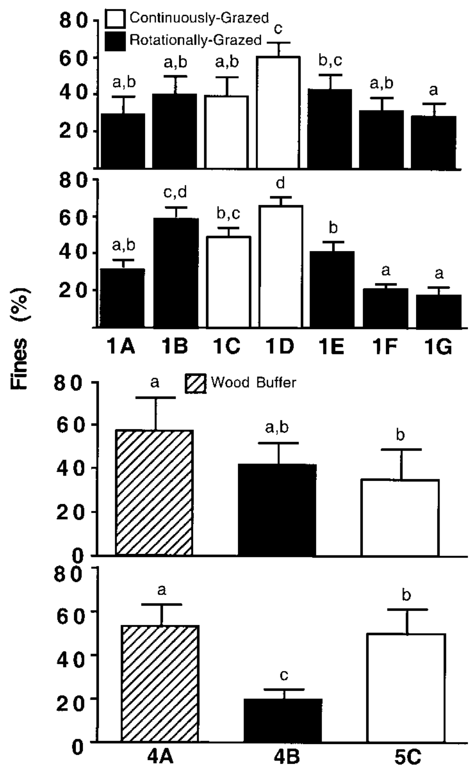

Figure 6. Percent fines at each site for streams 1, 4, and 5. Histograms with the same letter are not significantly different. 1995 first and third panels, 1996 second and fourth panels.

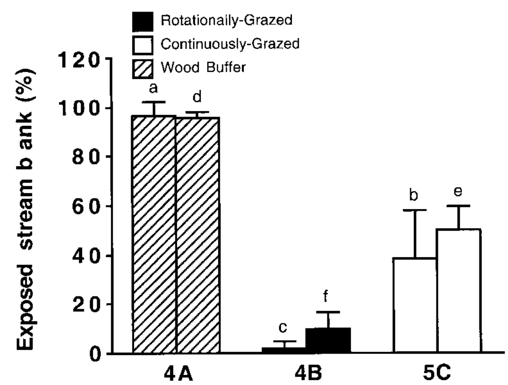

Figure 7. Percent exposed streambank at each site for streams 4 and 5. Histograms to the left: 1995, to the right: 1996. Histograms with the same letter are not significantly different.

Benthic Macroinvertebrates

Benthic macroinvertebrate metrics did not indicate consistent differences across sites or sampling periods for rotationally vs continuously grazed sites or among buffer comparisons, although across the 20 Kruskal-Wallis comparisons for each sampling period there were 10, 12, and 7 significant outcomes for spring 1995, fall 1996, and spring 1996, respectively (Table 4).

Rotational vs continuous grazing. For spring 1995, only FBI and percentage of dominance metrics differed significantly among sites in stream 1; both metrics were significantly higher at the downstream continuously grazed site (1D) than at the downstream rotationally grazed site (1G). In fall 1995, Kruskal-Wallis tests indicated differences among sites for FBI and EPT/C, but there were no significant differences between pairs of sites. In spring 1996, only FBI differed significantly among sites; and although not significantly higher, FBI at the downstream continuously grazed site (1D) remained high (high values indicate impaired water quality). FBI was significantly lower at the rotationally grazed site (4B) than at the continuously grazed site (4C) in spring 1995.

Buffer comparison. In fall 1995, FBI was lower at the wood buffer site (2D) than at the rotationally grazed site (2C), and EPT/C was higher at the wood buffer site (2D) than at the upstream grass buffer site (2A). In

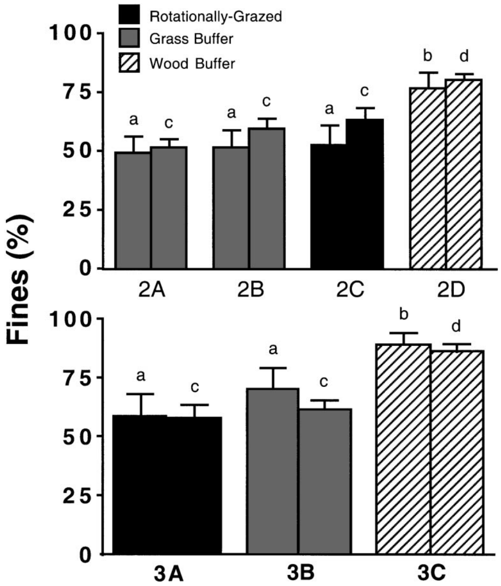

Figure 8. Percent fines at each site for streams 2 and 3. Histograms to the left: 1995, to the right: 1996. Histograms with the same letter are not significantly different. Sites 2A and 2B were rotationally grazed to the waters edge in 1996.

spring 1996, none of the macroinvertebrate metrics differed significantly for pairs of sites along stream 2.

In 1995, spring and fall macroinvertebrate metrics suggest less degraded water quality at the wood buffer site (3C) than at the rotationally grazed site (3A). For example, FBI for both spring and fall 1995 and percentage of dominance in spring 1995 were significantly lower at site 3C than at site 3A. However, in spring 1996 and for percentage of dominance in fall 1995 these trends did not hold.

Fish

Density and abundance of fish were lower at the wood buffer site in stream 3 for both years (Figure 11). No differences were noted for fish size within a species, number of species, abundance, or density within other streams.

Discussion

We observed watershed-level and annual climatic effects on stream water quality as reflected by differences in chemical, physical, and biotic indicators among catchments. Streams in this study are located in two ecoregions (Omernik and Gallant 1988) and are characterized by differences in geology and hydrology. Water chemistry values within the study streams were typical of agricultural catchments in southeastern Minnesota, characterized by high turbidity and large nutrient fluxes due to seasonal agricultural inputs and flashy stream discharge (McCollor and Heiskary 1993, Minnesota Pollution Control Agency, Minnesota Geological Survey and Minnesota Department of Natural Resources 1989, Troelstrup and Perry 1989, Minnesota Pollution Control Agency 1994, Sovell 1997). Recent studies have pointed out the importance of large-scale factors such as geology in differentiating streams (Rabeni and Sowa 1996, Richards and others 1996). However, within the context of large-scale watershed influences and annual differences in water discharge, impacts of local land management practices in relation to grazing or riparian areas were found within streams.

Physicochemical and biotic characteristics differed in relation to riparian management. Within streams, rotationally grazed sites differed significantly from continuously grazed sites, and sites with wood buffers differed significantly from grass and rotationally grazed buffers. Specifically, rotational livestock grazing demonstrated reduced fecal coliform levels and turbidity when compared to adjacent continuously grazed stream segments. Instream and riparian physical habitat measures suggest an increased percentage of fine sediment in the streambed along wood buffers relative to other riparian conditions. Physical habitat distinctions between continuously and rotationally grazed areas were less clear, but indicate reduced fines in the streambed along rotationally grazed sites. Although fish density and abundance differed in relation to riparian buffer condition on stream 3, they did not appear to differ between continuous and rotational grazing.

Mean monthly turbidity (strongly correlated with total suspended solids for southern Minnesota streams) and fecal coliform density were consistently higher at continuously grazed sites, where cattle had free access to streams, than at rotationally grazed sites, where access was permitted but limited temporally as cattle moved through upland as well as riparian paddocks. On stream 1, turbidity was 40% higher at the downstream continuously grazed site than at other sites. Bacteria from livestock can enter streams in runoff or are deposited directly when animals have access to the stream (Sherer and others 1988). Fecal coliform half-lives can be as short as 11 days where sedimentation is reduced (Sherer and others 1992). The limited half-lives of coliform bacteria found with reduced fine sediment and limited cattle access may account for the lower bacterial densities along rotationally grazed sites in this study. This scenario is consistent with Johnson and others (1978), who observed a significant drop in stream fecal coliform counts as quickly as nine days after cattle were excluded from a riparian pasture. Thus, rotational grazing as practiced in the Midwest

Figure 9. Grass buffer at site 2B in 1995. Note that the channel is only visible in the lower forground.

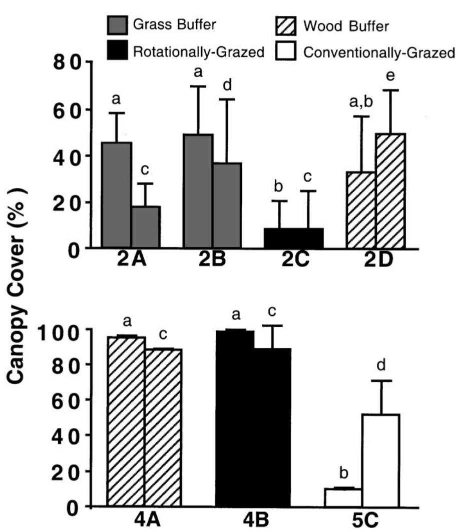

Figure 10. Percent canopy cover at each site for streams 2, 4, and 5. Histograms to the left: 1995, to the right: 1996. Histograms with the same letter are not significantly different. Note differences in scale on *y* axes.

may reduce fecal coliform abundance and erosion by effectively decreasing grazing intensity along streams.

The literature on buffer strips is dominated by considerations of forested riparian buffers (e.g., Schlosser and Karr 1981, Kauffman and Krueger 1984, Peterjohn and Correll 1984). This focus on wood buffer strips is related to temperature control and streambank stability provided by riparian woodlands. Physical habitat comparisons among riparian buffer types in our study challenge a widely held notion that wood buffer strips represent the optimum riparian condition for stream management and restoration (see Montgomery 1997). We noted that wood buffer sites were characterized by steep slopes, bare banks, little understory vegetation, and fine sediment-dominated streambeds. Lack of vegetative ground cover, due to almost complete canopy cover, along wood buffer sites may have reduced filtering of upland sediment and promoted erosion of streambank soils and subsequent deposition of fine materials in the stream channel. Daniels and Gilliam (1996) determined that grass buffers removed 50%–60% of the sediment entering the buffer and were more effective than mixed hardwood and pine buffers. Contributing factors for reduced sediment delivery to the streams in our study could be associated with the lower percentage of exposed soil on the streambanks (Owens and others 1989) and a reduction in runoff from the

Table 4. *P* values based on Kruskal-Wallis tests for benthic macroinvertebrate metrics for each system in 1995 and 1996[a]

	System							
	Spring				Fall			
	1	2	3	4	1	2	3	4
1995 metric								
Number of taxa	—	—	—	—	—	—	—	0.057
FBI	0.026	0.066	0.027	0.050	0.037	0.070	0.027	0.039
EPT	—	0.086	0.097	—	—	—	0.061	0.044
Dominance (%)	0.040	—	0.039	—	—	—	—	0.027
EPT/C	—	—	0.067	0.050	0.054	0.029	0.044	0.051
1996 metric								
Number of taxa	—	—	—	—				
FBI	0.053	—	0.060	—				
EPT	—	—	—	0.074				
Dominance (%)	0.072	—	0.051	—				
EPT/C	—	0.027	—	0.046				

[a]Blank cells indicate $P > 0.10$ except for fall 1996 when no data were collected. FBI = Hilsenhof's Family Biotic Index, EPT = Ephemeroptera, Plecoptera, Trichoptera; and EPT/C = Ephemeroptera, Plecoptera, Trichoptera/Chironomidae.

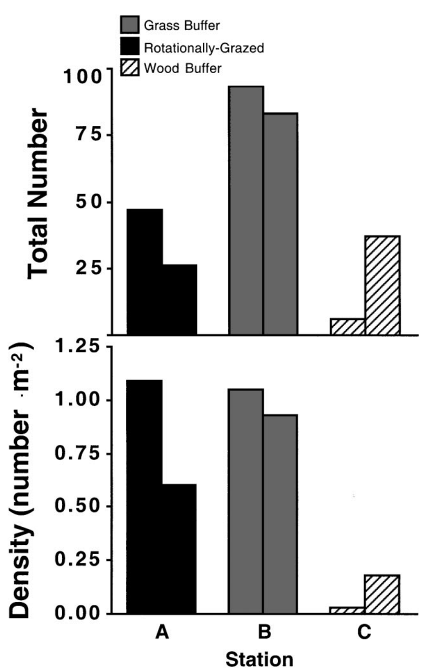

Figure 11. Total number of fish and fish density in stream 3. Histograms to the left: 1995, to the right: 1996.

grass buffers (sensu Kuhnle and others 1996). However, we did not quantify runoff in our study.

Trimble's (1993) distributed sediment model indicates intensive agriculture from the late 19th to the early 20th century increased sediment movement from tributaries to upper main valleys. Recognizing that flooding and downstream channel erosion are possible negative feedbacks currently still active, Trimble proposed that one land management option is to "convert riparian forest to grassland, allowing streams to create smaller channels, thus storing sediment."

On low-order streams, grass riparian buffers may also provide a canopy effect similar to that provided by wood buffer strips. Grass buffers along our sites formed an almost complete canopy, even on streams up to 4 m wide, and may have helped maintain lower water temperatures. Unfortunately, our study did not determine if there were any significant differences in temperature among sites. Grass buffers provide vegetative cover, stabilize streambank materials, and reduce erosion (Murgatroyd and Ternan 1983). Thus, small midwestern streams may benefit from grass buffers as a result of reduced solar insolation, streambank erosion, sedimentation, nutrient inputs, width-to-depth, and fecal coliform inputs.

Benthic macroinvertebrate metrics were not consistent among buffer types. Several macroinvertebrate metrics at rotationally and continuously grazed sites suggest less degraded water quality at the rotationally grazed sites, but were not consistent across sampling periods or within streams. Woody debris and organic matter may provide important structural material in streams with limited gravel or cobble and may have contributed to the observed indication of less degraded water quality at wood buffer sites, especially in 1995. The lack of consistency for invertebrate metrics may be related to substrate availability in the streambed within the study streams. Richards and others (1993) found that substrate was important in explaining macroinvertebrate community variation in regions dominated by fine material. Our study suggests that adequate substrate conditions may be limited in the streams studied, because many sites were dominated by fine materials. In streams with fine substrate, woody debris and organic matter can provide necessary food and hiding places for stream insects (Dudley and Anderson 1982, Reice 1974, 1980).

Fish species richness did not appear to be related to differences in grazing practices. However, fish density and abundance were related to riparian condition on stream 3, where substantially fewer fish were found at the wood-buffer site during both years. The extremely low density of fish may be related to the limited amount of suitable habitat available at the wood-buffer site (high width-to-depth ratio, percentage of fines, and embeddedness), and/or to reduced sampling efficiency. The close proximity (<100 m) to the other two sites along this stream (where density was higher) to the wood buffer site suggests that habitat is the limiting factor for fish.

Stream energy relationships among factors such as discharge, gradient, sinuosity, streambed particle distribution, and channel roughness, many of which were not measured during our study, influence riparian area physiognomy (Leopold and others 1964). An analysis of stream energy dynamics among riparian buffer strip types for our study is needed to determine the nature of the physical differences we found in relation to land and riparian management. Given the complexity of stream morphological processes and the historical changes that have occurred throughout the study area landscape, it is difficult to conclusively determine the exact relationship between recent changes in land management practices and stream physical condition.

The ability to understand stream dynamics in our study was limited by large-scale differences in hydrology and geology, upstream–downstream effects related to the spatial position of sites within watersheds, and the short time scale within which the study was executed. To completely understand stream system dynamics, spatial and temporal relationships (Rabeni and Sowa 1996) including the relationship of upstream to downstream land use must be considered (Vannote and others 1980, Wiley and others 1990). Paired watershed studies offer the potential to avoid upstream–downstream influences, but present the problem of selecting a pair of watersheds that produce a strong comparison (Spooner and others 1985).

The length of time for changes to occur as a result of management practices must be considered. While a relatively short-time lag will occur for water chemistry changes, a somewhat longer time is required for physical habitat changes to be evident. Biotic communities will respond only after changes are reflected in water chemistry and physical habitat. Changes in fish community composition may require two to three generations (4–10 years) for most species found in our streams to fully respond to improved water chemistry and physical habitat. The short duration of our study and the relatively short time following land management change to rotational grazing may have limited the ability to observe changes. Long-term stream monitoring may provide further insight into potential ecosystem changes that occur as a result of changes in upland and riparian management.

In conclusion, our study has implications for the planning and implementation of stream restoration programs in the midwestern United States. Successful stream and riparian restorations require the incorporation of disturbance regimes and rely upon examples of intact systems as references from which programs can be designed (Sousa 1984, Kauffman and others 1997). In the midwestern United States it is difficult to find examples of intact systems, given the duration and extent of alterations that have occurred within lotic systems. In a region where agriculture plays an important economic and social role, it is unlikely to be eliminated as an anthropological disturbance. Our study suggests that the incorporation of rotational grazing may provide a useful disturbance tool for restoring stream systems to their "potential natural community" state (Kauffman and others 1997). Together, these factors support the consideration of, and further research on, the composition of vegetation in riparian areas and their connection with midwestern streams.

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