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Influence of riparian forest buffers on macroinvertebrate and fish community structure in Illinois agricultural streams

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PROJECT DESCRIPTION

Agriculture has caused extensive landscape changes and is the leading cause of degradation to stream ecosystems in the United States. Stream degradation from agricultural activities is a serious concern in Illinois because agricultural land use dominates the state. Physical and chemical stream alterations from agricultural activities, such as channelization and removal of riparian vegetation, may be linked to declines in species diversity, changes in community composition, and loss of ecosystem integrity. Forested riparian buffers can improve stream water quality in agricultural areas by reducing soil erosion and filtering runoff before it enters a stream. In addition, forested riparian zones can moderate stream water temperature, supply important inputs of organic matter, and provide critical habitat that connects aquatic and terrestrial communities. Although riparian forest buffers are promoted to improve both water quality and ecological integrity, there has been much debate as to whether protecting land in the riparian zone is sufficient to mitigate large scale watershed agricultural disturbances. Previous studies have reached conflicting conclusions and suggest that more research in needed to understand the extent to which riparian areas influence community structure in agricultural streams.

This study examines nine agricultural headwater streams in Illinois with a gradient of riparian forest (range from 16-92%) to determine the effect of riparian forest and agricultural land use on stream community structure. Sampling was conducted seasonally to investigate temporal patterns in community structure related to land use, in-stream habitat, and water quality parameters. Over two years of sampling have been completed and results suggest macroinvertebrate and fish community structure are sensitive to agricultural influences at both the riparian zone and watershed scale. Macroinvertebrate and fish abundances were highest in streams with the lowest percent riparian forest and highest percent watershed agriculture. In contrast, streams with the highest percent riparian forest had the highest biotic integrity and most complex trophic structure. Results of this study provide information on the influence of riparian forest on stream ecosystems and have implications for managing and restoring riparian areas in Illinois agricultural watersheds.

INTRODUCTION

Many streams in the Midwestern United States have been impacted by the conversion of native vegetation to agriculture, particularly in Illinois where agriculture covers more than seventy six percent of the state's total area (USDA 2001). Agricultural development has profound effects on streams ecosystems because it significantly alters physical, chemical, and hydrological characteristics. An estimated one-third of Illinois streams have been altered by channelization for drainage or irrigation of farmlands (Swift 1984). Riparian areas comprise a significant portion of the remaining forested vegetation in the state (Iverson et al. 2001), but are generally confined to the lower reaches of watersheds (Wiley et al. 1990). Channelization, artificial drainage, and removal of riparian vegetation in agricultural areas can increase the severity and frequency of floods, create 'flashier' storm flows, and change the normal flow regime of a stream (Allan 2004). In addition, agricultural runoff contributes high loads of sediments, nutrients, and pesticides that degrade stream water quality (Osborne and Wiley 1988).

The physical and chemical stream alterations from agricultural activities impair habitat quality and alter resource availability and, therefore, affect stream biological communities. Agricultural streams generally exhibit communities with decreased species diversity, low overall abundance, and increased relative abundance of pollution tolerant taxa (Karr 1981). There are often shifts in macroinvertebrate functional feeding groups, such as increases in scrapers and grazers, in response to increases in algal biomass (Delong and Brusven 1998, Sponseller 2001). Fish trophic structure also typically changes in response to agricultural disturbance with omniovores and algivores that can spawn on fine sediments replacing benthic invertivores that require clean spawning substrate (Schlosser 1982). The dramatic change in community composition in agricultural areas is a principle threat to freshwater biodiversity and ecological integrity of stream ecosystems. Increasing awareness of the economic and ecological losses arising from stream degradation has fueled added pressures to implement land management strategies to conserve and restore flowing waters.

Riparian vegetation 'buffers', typically 30-50 m wide alongside streams, are widely recommended and promoted in agricultural areas as a method for improving degraded stream ecosystems (NRC 2002, Bernhardt et al 2005). Riparian buffers can improve stream water quality in agricultural areas by reducing soil erosion and filtering nutrients and pesticides out of runoff before it enters a stream (Schlosser and Karr 1981, Osborne and Kovacic 1993, Lovell and Sullivan 2006). Riparian vegetation can also stabilize stream banks, restore channel morphology, and protect natural flow patterns (Lyons et al. 2000). Forested riparian zones can moderate stream water temperature through shading, supply important inputs of organic matter such as leaves and seeds, and provide critical habitat that connects aquatic and terrestrial communities (Sweeney 1993, Naiman and Decamps 1997). Previous studies have addressed the size of riparian buffers needed to reduce agricultural pollutants (e.g., Lee et al. 2004), but few have determined the effectiveness of riparian forest buffers to ameliorate the impacts of agricultural activities on stream biological communities.

This study examines headwater streams with a gradient of riparian forest buffer and watershed agriculture to determine the extent to which riparian forest buffers can mitigate the effects of agricultural land use on stream ecosystems. The assumption of riparian buffer policies is that protecting land adjacent to streams is sufficient to mitigate watershed-wide disturbances to water resources and stream ecosystems. However, some studies have found that riparian forests are not sufficient for protecting stream ecosystems in highly disturbed areas (Allan and Johnson 1997, Harding et al. 1998, Roy et al. 2006). Changes in riparian vegetation and agricultural land use may have the greatest impact on headwater streams because their watersheds are small so they are strongly influenced by land use disturbances (Meyers et al. 2007). This project focused on headwater streams because, in addition to being sensitive to land use changes, they play a vital role in protecting the biotic integrity of downstream reaches (Wipfli 2007). We predicted that streams with low percent riparian forest buffer will have altered stream habitat conditions, higher nutrient levels, and increased algal production compared to higher forest buffer sites. These abiotic changes were expected to affect the abundance, distribution, and trophic structure of macroinvertebrate and fish communities.

OBJECTIVES

1) Conduct seasonal (Spring, Summer, Fall) assessments of macroinvertebrate and fish communities in nine headwater streams that range from high to low percent riparian forest buffer and watershed agricultural land use.

2) Quantify stream habitat characteristics, water chemistry parameters, and basal energy sources to investigate instream habitat and water quality differences among streams.

3) Compare seasonal and annual variation in macroinvertebrate and fish community composition among streams related to land use, instream habitat, and water quality parameters.

METHODS

Site Selection and Study Design

The nine study sites are second and third order streams located in the Embarras River Watershed in east-central Illinois. Land use within the watershed is dominated by row-crop and small grain agriculture (73.5%) with low levels of urban development (1.8%). This watershed was chosen because it has considerable corridors of intact natural riparian forest. Individual study sites were selected using ArcView GIS 9.1 (ESRI 2005) based on the Land Cover of Illinois 1999-2000 Classification on-line database compiled by the Illinois Department of Natural Resources and the Illinois State Geological Survey (IDOA 2001). Land use was inferred from land cover information. Land use categories were created based on the overall proportion of agriculture (row crop + non-row crops), forest (upland + lowland wooded forest), and urban development (residential + commercial). Land use proportions for each site were calculated for the riparian zone (land use within 30 m of the stream) and watershed level spatial scale (the entire area upstream of each site). Riparian buffer width size was selected based on resolution of land cover data (30-m pixels) and because 30 m is the minimum size recommended by the USDA for protection of water quality (Welch 1991). The study reaches were selected to have similar watershed area (range 27-40 km^2) in an attempt to minimize potential differences between sites unrelated to land use. The nine sites were equally divided into three distinct land use groups that cover the available range of riparian forest (16-92%) and agricultural land use (48-89%) in the Embarras Watershed (Figure 1). All sites selected had very little (<1%) urban development. This study design was chosen to provide information on the influence of land use changes in the riparian zone and watershed level spatial scale.

Land Use Group Descriptions

HIGH BUFFER – LOW AGRICULTURE STREAMS: This group of streams have high percentages of riparian forest buffer and the lowest amounts of watershed agriculture (<51%) in the entire Embarras River watershed.

Bennett Creek: This stream is located in Stoy, IL 0.5 mile south of U.S. Route 33.

Brushy Creek: This stream is located south of Flat Rock, IL 1.3 mile east of U.S. Route 1.

Honey Creek: This stream is located 4.5 miles south of Robinson, IL near New Hebron, IL.

HIGH BUFFER – HIGH AGRICULTURE STREAMS: Streams in this group have high percentages of riparian forest buffer (>60%) and high percentages of watershed agriculture (>70%).

East Crooked Creek: It is located 5 miles south of Greenup, IL 4.5 miles east of U.S. Rt.130.

Lost Creek: This stream is located 2.5 miles north of Greenup, IL 1 mile east of U.S. Rt.130.

Panther Creek: This stream is located near Yale, IL 1.5 mile west of U.S. Route 49.

LOW BUFFER – HIGH AGRICULTURE STREAMS: These streams have low percentages of riparian forest buffer (<35%) and high amounts of watershed agriculture (>75%).

Bear Creek: This steam is located northwest of Toledo, IL 2.7 miles north of U.S. Rt. 121.

Cottonwood Creek: This stream is located near Bradbury, IL 1.3 mile east of Co Rd. 1200 E.

West Crooked Creek: This stream is located near Hidalgo, IL 1 mile east of U.S. Route 130.

Physical and Chemical Characterization

Sampling for this study was conducted seasonally (Spring, Summer, Fall) over two years to explore patterns of temporal variation. To minimize influence of flow conditions on sampling efficiency and habitat measurements, sampling only took place when streams were considered to be at base flow. Habitat and hydrology measurements followed measures adapted from the U.S. Geological Survey (USGS) protocol for the National Water-Quality Assessment (NAWQA) Program (Fitzpatrick et al. 1998). At each site, a representative 100 meter reach was selected to include two riffle-pool sequences. Mean reach width, depth, flow, and substrate size were estimated based on measurements taken at 10 equally spaced cross-stream transects. Stream depth and flow velocity (FLOW-MATE, Marsh-McBirney, Fredrick, MD) were measured at five equally spaced points along each transect. At each sampling point, the dominant substrate surrounding the point was classified as sand or silt (particle size < 2mm), gravel (2-16mm), pebble (17-64mm), cobble (65-256 mm), boulder (> 256 mm), or bedrock (Wentworth 1922).

During each sampling period, water temperature, dissolved oxygen, conductivity, and pH were measured using a hand held meter. In addition, water samples were collected for determining dissolved reactive phosphorous (SRP) and total phosphorous (TP) concentrations using the ascorbic acid method (APHA et al. 2005) and nitrate-nitrogen (NO_3-N) concentration was analyzed using second-derivative spectroscopy (Crumpton et al. 1992). To compare thermal regimes among land use type, stream water temperatures were recorded every 30 minutes at each site during summer months using submerged temperature loggers (HOBO Stowaway, Onset Computer, Massachusetts, USA). Basal energy source availability was determined for each study reach by measuring the amount of allochthonous material (benthic and suspended organic matter) compared to autochthonous production (periphyton) at each sampling time. Benthic particulate organic matter was sampled using sediment cores and divided into coarse particulate organic matter (CPOM > 1mm) and fine particulate organic matter (FPOM < 1mm) (Delong and Brusven 1993). Periphyton was collected using the inverted petri dish method for sampling epipsammic habitats (Moulton et al. 2002) and chlorophyll *a* and ash-free dry mass (AFDM) was measured to estimate algal biomass and primary productivity.

Stream Community Assessment

Within a selected 100-m reach at each site, benthic macroinvertebrates were collected using a 500-μm mesh Kicknet and Hess sampler. Multi-habitat kicknet sampling followed methods described in Barbour et al. (2002) where 20 kick samples were taken at locations reflecting the proportion of the microhabitat type present in each stream site. Net contents were pooled and the composite sample preserved in 80% ethanol. Six Hess samples (500-μm mesh, 350mm diameter) were also taken to provide more quantitative estimates of macroinvertebrate density. Macroinvertebrates collected were identified and subsequently classified into functional feeding groups according to Merritt and Cummins (1996). Fish populations were sampled over the same 100-m reach using a Smith-Root backpack electrofishing unit. Blocking seines were placed at the ends of the reach to recover fish missed by dip netting. Fish that could not be identified on site were preserved in 10% formalin and subsequently identified in the laboratory. Fish species were classified into guilds based on dominant food source (Gerking 1994).

Data Analyses

Macroinvertebrate and fish biotic metrics (e.g., taxa richness and IBI) were used to determine community differences related to land use, in-stream habitat, water chemistry, and basal energy sources. Data was transformed to approximate normality or to improve variance homogeneity before analysis. Abundance, biotic metrics, and select physical/chemical variables were compared among stream land use group and sample periods with repeated-measures analysis of variance (ANOVA) using PROC MIXED (SAS Institute Inc. 2009). Akaike Information Criterion (AIC) values were used to select the most appropriate covariance structure for within-subject variation in the repeated measures model.

RESULTS

Physical and Chemical Characterization

More than two years of sampling and laboratory processing have been completed. Results suggest that most of the hydrological characteristics (mean depth, velocity, width) were similar among streams ($P > 0.05$, Table 1). However, maximum depth was significantly greater in streams in the two high forest buffer land use groups compared to the low forest buffer streams ($P < 0.05$, Table 1). Although there were significant seasonal differences among many of the abiotic parameters, relationships among streams with different land use type were relatively consistent. Low Buffer – High Agriculture streams had significantly higher daily maximum temperature, nitrate-nitrogen levels, and chlorophyll *a* concentrations ($P < 0.05$, Table 1, Figure 2A) than the two high forest buffer land use groups. High Buffer – Low Agriculture streams consistently had the lowest total phosphorus levels, however, differences in total phosphorus concentrations among land use groups were not significant (Table 1).

Stream Community Assessment

Organisms representing 70 different families of macroinvertebrates and 36 species of fish were identified in the nine headwater streams sampled within the Embarras River watershed

(Appendix 1 and 2). Taxa richness in individual sample sites ranged from 5 – 19 fish species and 13 – 32 macroinvertebrate taxa for a single sampling date and from 15 – 27 fish species and 29 – 53 macroinvertebrate taxa over the course of the study. Macroinvertebrate and fish richness and diversity indices were not significantly different among land use groups, but there were significant differences among sampling periods (Table 2). Fish species richness was significantly higher in summer 2008 compared to spring 2008 (t = 2.34, P = 0.03) and summer 2009 (t = 3.26, P = 0.004) and macroinvertebrate richness was significantly higher in fall 2008 compared to summer 2007 (t = 3.89, P < 0.001), spring 2008 (t = 5.58, P < 0.001), and summer 2008 (t = 4.32, P < 0.001).

Macroinvertebrate density differed significantly among land use groups (Table 2) and was significantly greater in Low Buffer – High Agriculture streams compared to streams with high forest buffer (Figure 2B). The highest densities of fish were also found in Low Buffer – High Agriculture streams, but differences in fish density among land use groups were only marginally significant (P = 0.07, Table 2, Figure 2C). Although the highest abundances of organisms were present in streams with low forest buffer, macroinvertebrate and fish assemblages were dominated by pollution tolerant taxa, such as chironomids and creek chubs.

Biotic integrity scores and fish trophic structure were significantly different among land use groups (Table 2). High Buffer - Low Agriculture streams had significantly better biotic integrity than the Low Buffer – High Agriculture streams, as indicated by FBI and IBI scores, while High Buffer – High Agriculture sites were intermediate (Figure 3). High Buffer –Low Agriculture streams also had significantly greater proportions of benthic invertivores (P < 0.05), such as blackside darters, and piscivores (P < 0.05), such as grass pickerel and largemouth bass, compared to land use groups with high watershed agriculture (Figure 4). These results indicate a balanced, trophically diverse ecosystem in streams with high levels of riparian forest buffer and low levels of watershed agriculture (Figure 4). In contrast, Low Buffer – High Agriculture streams had significantly greater proportions of herbivores-detritivores ($P < 0.05$, Figure 4), such as central stonerollers, than both high forest buffer land use groups.

DISCUSSION

Our results suggest that headwater stream ecosystems are sensitive to changes in riparian forest and watershed agriculture. However, comparisons among the three land use groups indicate that riparian forest buffer may have the strongest influence on key in-stream parameters (e.g., nutrients), overall abundances of macroinvertebrate and fish, and community composition. Despite a large range in watershed agriculture (48-79%) between the two High Buffer land use groups, streams with high levels of riparian forest had similar responses for many of the abiotic and biotic parameters. Streams with low percentages of riparian forest had significantly higher maximum daily temperatures and greater algal productivity (chlorophyll *a*) due to increase light levels compared to streams with high forest buffer. Streams with high percent forest buffer had significantly lower nitrate-nitrogen levels indicating that riparian vegetation may be effectively reducing nutrient runoff.

Riparian buffers have been recommended as one of the most effective tools for mitigating agricultural pollution and improving aquatic ecosystems (Lee et al. 2004, Mayer et al. 2007). However, there has been considerable debate as to whether relatively narrow riparian areas can protect stream communities from watershed scale agricultural disturbances. Although some studies have found watershed-wide land use more important than stream buffers for maintaining and restoring stream ecosystems (Richards et al 1996; Stephenson and Morin 2009), our results support previous studies that have concluded land use within riparian buffers is the most important factor in driving differences in stream communities (Lammert and Allan 1999, Bunn and Davies 2000, Sponseller et al 2001, Wilson and Xenopoulos 2008). Macroinvertebrate and fish species richness and diversity did not differ among land use groups, but there were differences in abundances and community composition related to the presence or absence of riparian forest buffers. Total macroinvertebrate and fish abundance has often been found to increase in streams where canopy cover has been reduced by riparian forest clearing (e.g. Lorin and Kennedy 2009), due primarily to increased primary production. Therefore, it was not surprising that we found the greatest densities of macroinvertebrate and fish in streams with low forest buffer. Assemblages in these streams were dominated by highly tolerant species (e.g., chironomidae larvae and creek chub) that may have been able to proliferate in the absence of top predators at those sites.

Patterns of fish trophic structure differed among land use groups. Streams with low forest buffer and high watershed agriculture had the greatest proportions of herbivore-detritivores, presumably taking advantage of higher algal biomass in those streams. In contrast, streams with high percent forest buffer and relatively low watershed agriculture had the greatest proportions of benthic invertivores and piscivores. Although most hydrological characteristics (width, flow, average depth) were similar among land use groups, maximum depth was significantly greater in streams with high percent forest buffer. Riparian trees along the bank influence channel morphology by slowing water velocities and creating deeper pools in areas where dense roots systems or fallen trees occur (Lyons et al. 2000). Woody debris, in the form of logs and other wooden structures, is often placed into streams to improve fisheries (Lyons et al. 2000). In addition to providing a refuge and supporting fish diversity (Pusey and Arthington

2003), the presence of woody debris in streams has been shown to be an important determinant of growth rates for piscivorous fish (Persson and Eklov 1995) and may be used as cover for ambush predators (Pusey and Arthington 2003). The increased maximum depth and habitat structure from root masses and woody debris may explain the higher proportions of top predators in our study streams with the highest percent riparian forest.

Current efforts to manage and restore stream ecosystems require a better understanding of how stream communities are influenced by riparian and watershed land use. Results of our study suggest that differences in riparian forest buffer have the greatest effect on environmental parameters among agricultural headwater streams and have the strongest influences on macroinvertebrate and fish abundance and community composition. However, streams with the lowest percent watershed agriculture had the best macroinvertebrate FBI and fish IBI scores and were the most trophically diverse streams indicating that variation in the amount of watershed agriculture also influences stream biotic integrity and trophic structure. Taken together, our results suggest that headwater ecosystems are sensitive to agricultural impacts at both the riparian zone and watershed scale, but variation in riparian forest buffers has the greatest effect on stream communities. We are currently completing a third year of sampling at the same sites to determine if patterns of variation among stream communities are consistent across multiple years. Future analysis of this data will include a combination of multivariate techniques (e.g., PCA, NMDS) and regression analyses to compare environmental variables and species compositions among streams to identify potential mechanisms for patterns of variation in community structure related to riparian forest and agricultural land use.

The results of this research are of use to scientists investigating the functional role of riparian buffers in aquatic ecosystem protection, as well as use by agencies that administer policies that promote establishment of riparian buffers. Watershed managers have widely adopted the use of riparian forest buffers as a Best Management Practice for protecting stream ecosystems (Lowrance et al. 1997), but in the Midwestern United States $<20\%$ of the natural riparian plant communities still exist (Burkart et al. 1994). Our results indicate that riparian forest buffers may be able to mitigate the effects of agricultural land use on headwater stream ecosystems in Illinois. Because headwaters influence water quality and provide resources for downstream ecosystems, protection of riparian forest buffers along headwater streams have potential benefits for improving aquatic resources in reservoirs and larger rivers downstream. Therefore, we recommend continued restoration of riparian forest buffers (through programs such as the USDA's Conservation Reserve and Enhancement Program; NRC 2002) as an effective stream management strategy in areas of Illinois modified by agricultural land use.

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Table 1. Physical and chemical characteristics of the three land use groups (Mean (SE), n = 3). Streams categorized into distinct land use groups based on percentages of watershed agricultural land use and riparian forest buffer (30 m). Different letters indicate a significant difference between land use groups ($P < 0.05$) based on repeated-measures ANOVA models.

	Land use group		
	High Buffer Low Agriculture	High Buffer High Agriculture	Low Buffer High Agriculture
Watershed area (km^2)	34.5 (4.5)	35.5 (3.5)	33.2 (1.2)
Riparian forest %	81 (4)	77 (6)	19 (4)
Watershed agricultural land use %	51 (2)	76 (2)	83 (4)
Mean velocity (m/s)	0.17 (0.03)	0.12 (0.03)	0.11 (0.03)
Channel width (m)	5.3 (0.5)	5.1 (0.5)	4.7 (0.5)
Depth (cm)	17 (2.3)	19 (2.4)	18 (2.6)
Maximum depth (cm)	28 (3.6)	28 (3.5)	24 (3.8)
Temperature (^{o}C)	21.6 (0.5)	21.1 (0.7)	22.4 (0.6)
Daily Maximum temperature (^{o}C)	24.6 (0.8)a	24.1 (1.0)a	28.1 (0.9)b
NO_3-N (mg/L)	0.45 (0.18)a	0.42 (0.19)a	1.51 (0.22)b
TP (mg/L)	0.10 (0.05)	0.22 (0.05)	0.26 (0.06)
Chlorophyll *a* (µg/cm2)	0.72 (0.14)a	0.83 (0.13)a	5.54 (0.16)b

Table 2. Results of repeated-measures ANOVA models for density, biotic metrics, and fish trophic groups with land use group and sample period as fixed effects (n = 3 for each land use group).

Parameter	Land Use Type		Sample Period		Land Use X Sample Period	
	$F_{2,7}$	P	$F_{4,21}$	P	$F_{8,21}$	P
Macroinvertebrate						
Density (individual/m^2)	**15.56**	**0.004**	**25.96**	**<.001**	**3.16**	**0.032**
Taxa Richness	5.11	0.059	**13.88**	**<.001**	1.10	0.404
Diversity (Shannon)	2.49	0.163	0.72	0.589	1.10	0.405
FBI	**6.75**	**0.021**	0.65	0.454	1.21	0.363
Fish						
Density (individual/m^2)	3.57	0.065	**1.28**	**0.310**	**1.79**	**0.137**
Species Richness	0.01	0.990	**3.83**	**0.017**	2.02	0.095
Diversity (Shannon)	0.33	0.729	**8.01**	**<.001**	**5.03**	**0.001**
IBI	**4.75**	**0.042**	**7.44**	**<.001**	2.03	0.093
Fish Trophic group						
% Herbivore-Detritivore	**5.44**	**0.038**	0.96	0.451	2.10	0.083
% Omnivore	0.65	0.553	**3.85**	**0.017**	0.59	0.776
% Invertivore	**10.12**	**0.009**	1.02	0.421	1.47	0.226
% Piscivore	**6.18**	**0.029**	**5.87**	**0.003**	0.97	0.488

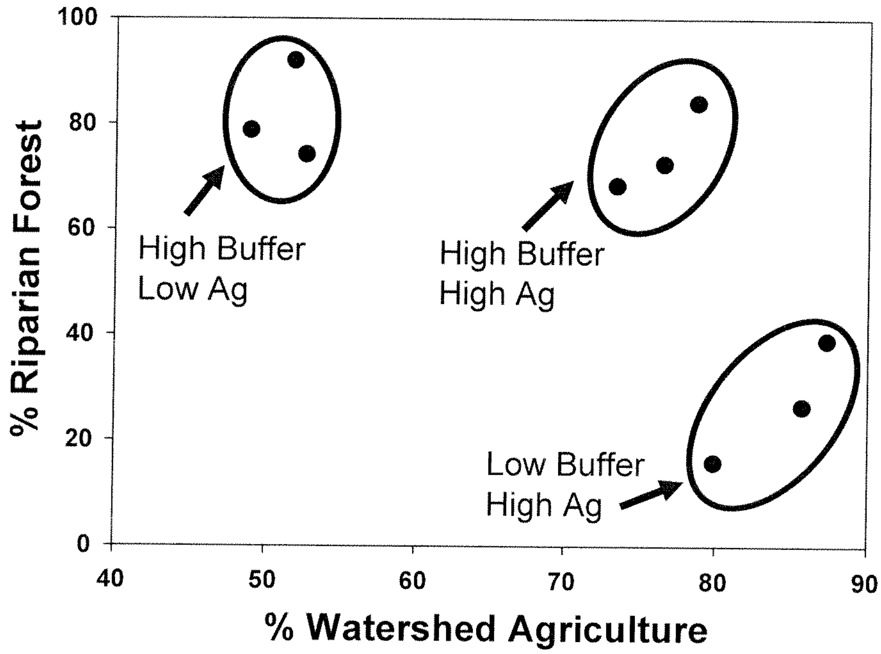

Figure 1. Relationship between percent riparian forest (within 30 m of stream) and percent watershed agriculture for the nine study streams within the Embarras River Watershed. Study sites were divided into three land use categories that cover the highest and lowest percent forest buffer and watershed agriculture within the entire study area.

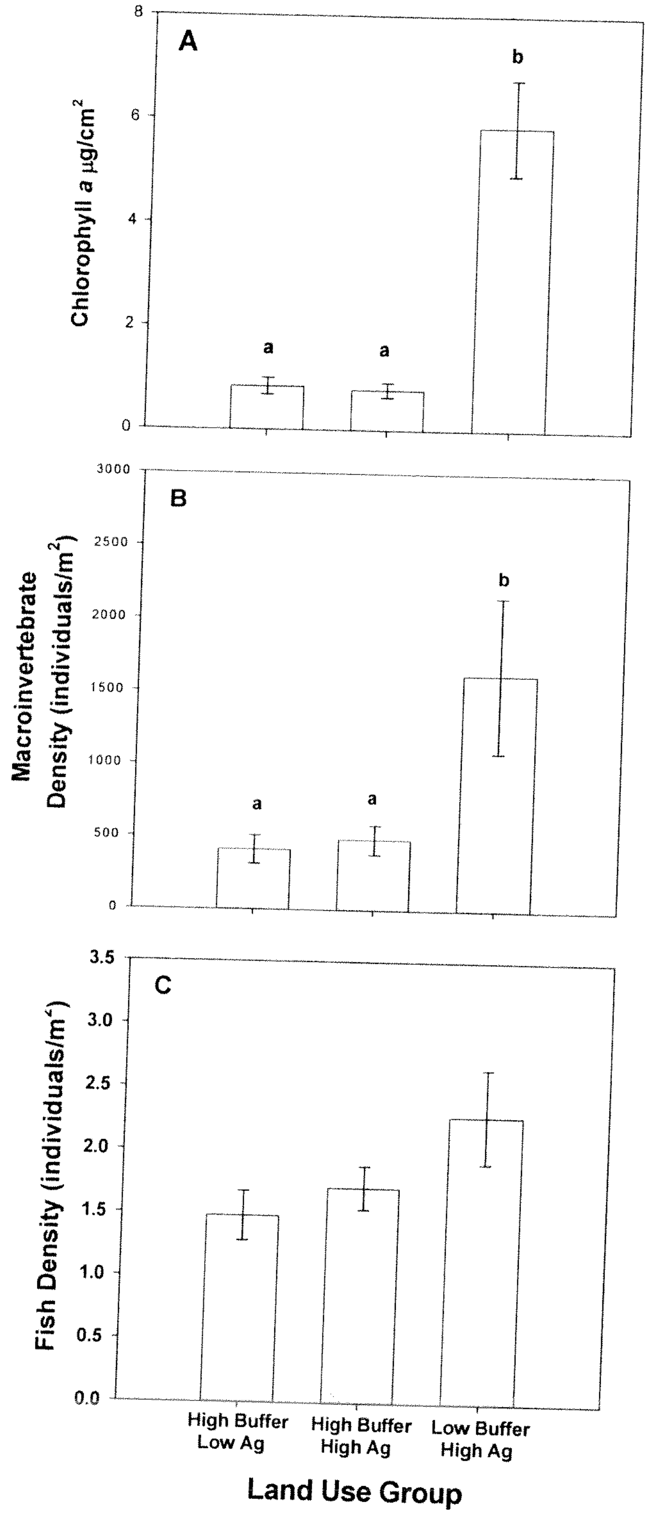

Figure 2. Means and standard errors of (A) chlorophyll *a* µg/cm^2 (B) macroinvertebrate density and (C) fish density for each land use group ($n = 3$) for all sampling periods combined ($n = 5$). Bars with different letters indicate a significant difference between land use groups ($P < 0.05$) based on repeated-measures ANOVA models (see Table 1 and 2).

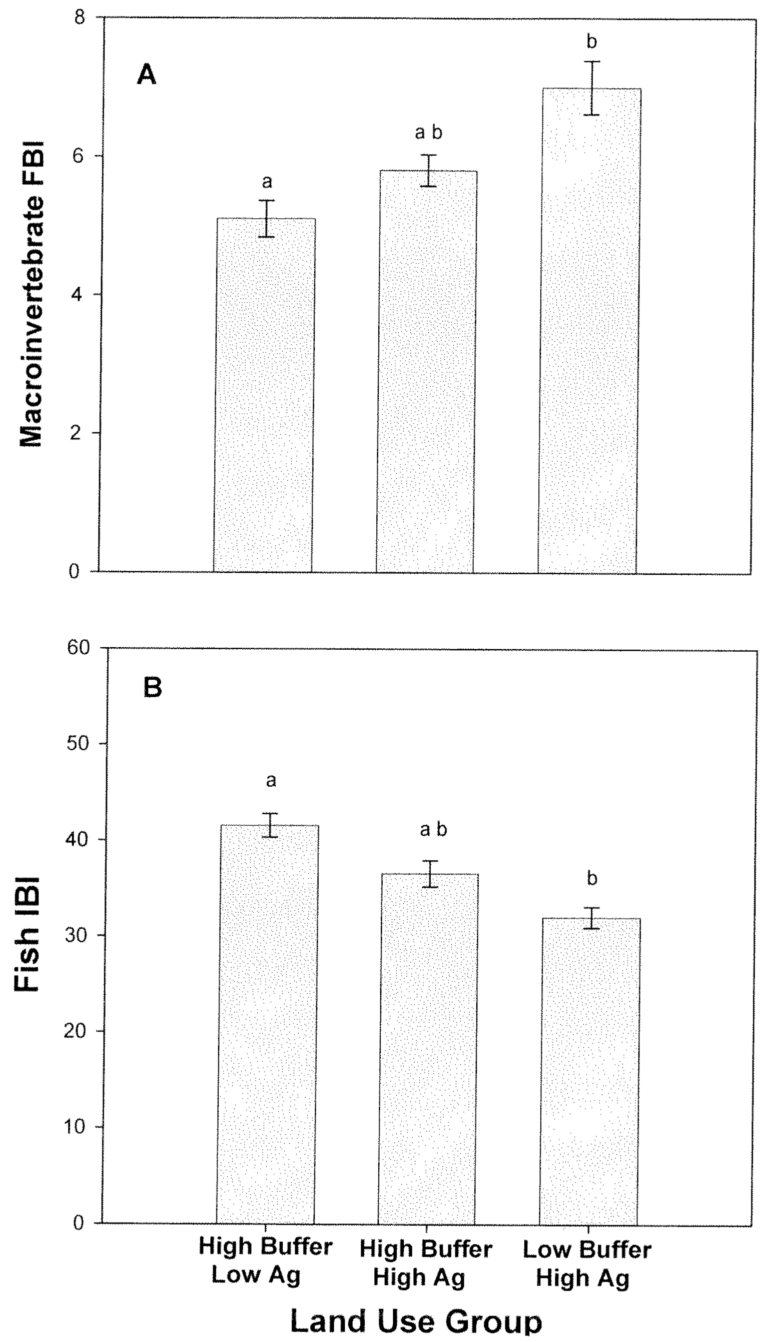

Figure 3. Means and standard errors of (A) macroinvertebrate family-level biotic index (FBI) and (B) fish index of biotic integrity (IBI) for each land use group (n = 3) for all sampling periods combined (n = 5). Bars with different letters indicate a significant difference between land use groups ($P < 0.05$) based on repeated-measures ANOVA models (see Table 2). Macroinvertebrate FBI is based on taxa tolerance to organic pollution (ranges 0 – 10) with higher values indicating poorer water quality (Hilsenhoff 1988). Fish IBI takes a multimetric approach with higher values indicating better water quality and improved ecological conditions (Karr 1981).

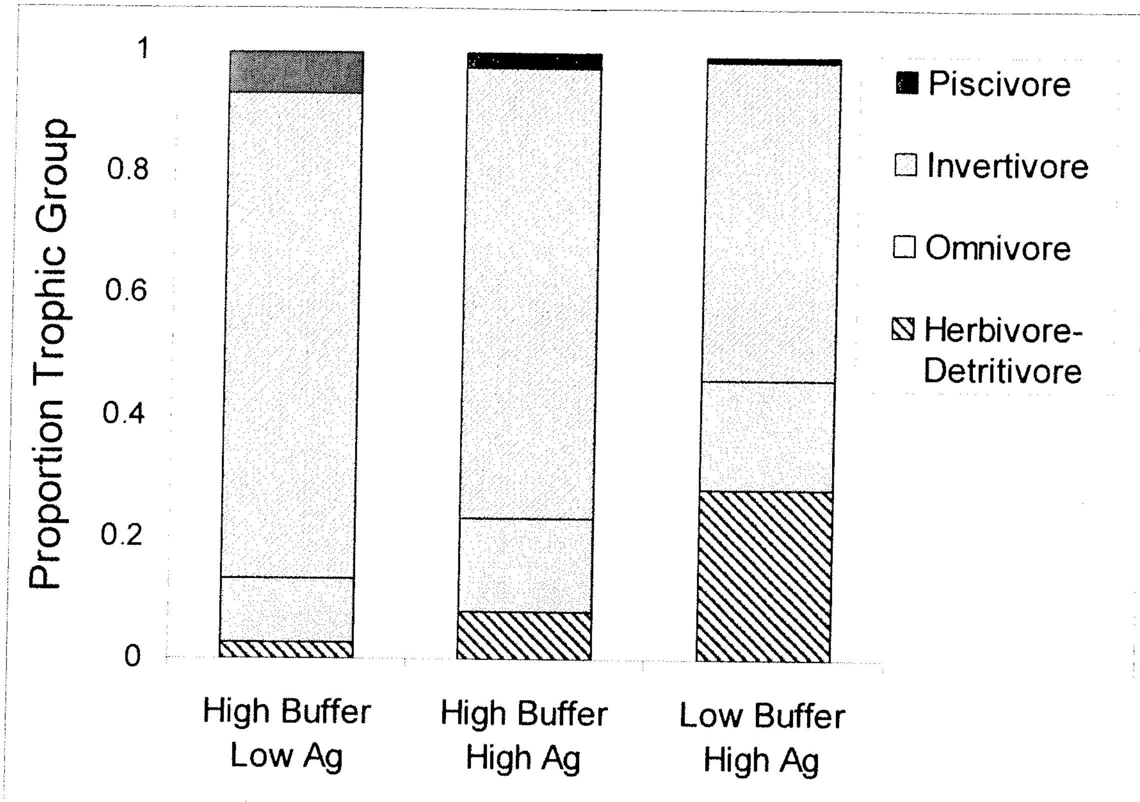

Figure 4. Mean proportions of fish trophic groups for each land use group ($n = 3$), all sampling combined ($n = 5$).

Appendix 1. Macroinvertebrate taxa collected in headwaters streams in the Embarras Watershed.

Order	Family
Amphipoda	Gammaridae
	Hyalellidae
Annelida	Hirudinea
	Oligochaeta
Bivalvia	Sphaeriidae
Coleoptera	Chrysumelidae
	Curculionidae
	Dryopidae
	Dytiscidae
	Elmidae
	Gyrinidae
	Haliplidae
	Hydrophilidae
	Scirtidae
Collembola	
Decapoda	Cambaridae
	Palaemonidae
Diptera	Ceratopogonidae
	Chironomidae
	Culicidae
	Curculionidae
	Dolichopodidae
	Empididae
	Ephydridae
	Muscidae
	Psychodidae
	Sciomyzidae
	Simuliidae
	Stratiomyidae
	Syrphidae
	Tabanidae
	Tipulidae
Ephemeroptera	Baetidae
	Caenidae
	Ephemerellidae
	Ephemeridae
	Heptageniidae
	Leptophlebiidae
	Tricorythidae

Order	Family
Gastropoda	Ancylidae
	Hydrobiidae
	Lymnaeidae
	Physidae
	Planorbidae
	Viviparidae
Hemiptera	Belostomatidae
	Corixidae
	Gerridae
	Hebridae
	Notonectidae
	Pleidae
	Saldidae
	Veliidae
Isopoda	Asellidae
Megaloptera	Corydalidae
Nematoda	
Nematomorpha	
Odonata	Aeshnidae
	Calopterygidae
	Coenagrionidae
	Cordulegastridae
	Gomphidae
	Lestidae
	Libellulidae
Plecoptera	Leuctridae
	Perlidae
	Perlodidae
Trichoptera	Hydropsychidae
	Hydroptilidae
	Leptoceridae
	Limnephilidae
	Philopotamidae
	Phryganeidae

Appendix 2. Fish species collected in headwater streams in the Embarras River Watershed.

Common name	**Scientific name**
Bowfin	*Amia calva*
Grass pickerel	*Esox americanus vermiculatus*
Common carp	*Cyprinnus carpio*
Golden shiner	*Notemigonus crysoleucas*
Creek chub	*Semotilus atromaculatus*
Central stoneroller	*Campostoma anomalum*
Suckermouth minnow	*Phenacobius mirabilis*
Striped shiner	*Luxilus chrysocephalus*
Redfin shiner	*Lythrurus umbratilus*
Bluntnose minnow	*Pimephales notatus*
Emerald shiner	*Notropis atherinoides*
Sand shiner	*Notropis ludibundus*
Silverjaw minnow	*Notropis buccatus*
Spotfin shiner	*Notropis spilopterus*
Steelcolor shiner	*Notropis whipplei*
River carpsucker	*Carpiodes carpio*
White sucker	*Catostomus commersoni*
Spotted sucker	*Minytrema melanops*
Creek chubsucker	*Erimyzon oblongus*
Yellow bullhead	*Ameiurus natalis*
Black bullhead	*Ameiurus melas*
Pirate perch	*Aphredoderus sayanus*
Blackstripe topminnow	*Fundulus notatus*
Mosquitofish	*Gambusia affinis*
Largemouth bass	*Micropterus salmoides*
Spotted bass	*Micropterus punctulatus*
Warmouth	*Lepomis gulosus*
Green sunfish	*Lepomis cyanellus*
Bluegill	*Lepomis macrochirus*
Longear sunfish	*Lepomis megalotis*
Blackside darter	*Percina maculata*
Johnny darter	*Etheostoma nigrum*
Orangethroat darter	*Etheostoma spectabile*
Rainbow darter	*Etheostoma caeruleum*
Banded darter	*Etheostoma zonale*

PROJECT EXPENDITURES

D7414 IDNR 10-024W
596006-375009-191100

Project Period 10/24/2009-6/30/2010	Supplies	Travel	Wages/F.B.	F&A	Total
Requested Budget	90.00	160.00	1,547.00	180.00	1,977.00

Date	Vendor/Description	Supplies	Travel	Wages/F.B.	F&A	Total
3/25/2010	Cabelas One Cabela Dr., Sidney, NE 69160 Item: Chest Waders	90.00			9.00	99.00
	Personnel Payroll - Technician			1,542.13	154.21	1,696.34
	Travel - Eden Effert		159.34		15.93	175.27
	TOTAL EXPENDITURES					1,970.62