Relations between Retired Agricultural Land, Water Quality, and Aquatic-Community Health, Minnesota River Basin

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The relative importance of agricultural land retirement on water quality and aquatic-community health was investigated in the Minnesota River Basin. Eighty-two sites, with drainage areas ranging from 4.3 to 2200 km^2, were examined for nutrient concentrations, measures of aquatic-community health (e.g., fish index of biotic integrity [IBI] scores), and environmental factors (e.g., drainage area and amount of agricultural land retirement). The relation of proximity of agricultural land retirement to the stream was determined by calculating the land retirement percent in various riparian zones. Spearman's rho results indicated that IBI score was not correlated to the percentage of agricultural land retirement at the basin scale ($p = 0.070$); however, IBI score was correlated to retired land percentage in the 50- to 400-m riparian zones surrounding the streams ($p < 0.05$), indicating that riparian agricultural land retirement may have more influence on aquatic-community health than does agricultural land retirement in upland areas. Multivariate analysis of covariance and analysis of covariance models indicated that other environmental factors (such as drainage area and lacustrine and palustrine features) commonly were correlated to aquatic-community health measures, as were in-stream factors (standard deviation of water depth and substrate type). These results indicate that although agricultural land retirement is significantly related to fish communities as measured by the IBI scores, a combination of basin, riparian, and in-stream factors act together to influence IBI scores.

Stream conditions are influenced by interactions among physical and chemical factors. In the Minnesota River Basin, the loss of riparian vegetation and natural land cover has reduced habitat, modified hydrologic conditions, and changed water quality (Stark et al., 1996; Stauffer et al., 2000; Lee et al., 2001). Agricultural activities in general have resulted in widespread degradation of water quality (Capel et al., 2008). Three important spatial scales that influence physical, chemical, and biological conditions are basinwide conditions, riparian (area adjacent to the stream) conditions, and in-stream conditions. Numerous researchers have investigated the relation of land use to water quality at different spatial scales (Sliva and Williams, 2001; Bolstad and Swank, 1997; King et al., 2005) and the spatial arrangement of land cover on stream ecosystems (O'Neill et al., 1997). More specifically, some investigators have compared land cover within certain distances from a stream or sampling site (Omernik et al., 1981; Schuft et al., 1999; Sponseller et al., 2001).

The Minnesota River Basin, located primarily in the State of Minnesota (Fig. 1), is part of the Midwest Corn Belt, one of the most productive and intensively managed agricultural regions in the world. Agricultural activities have changed landscapes throughout the Midwest, and streams in the Minnesota River Basin commonly have diminished aquatic resources due to intense physical and chemical perturbations (Stauffer et al., 2000). To address concerns about degradation of agricultural streams, Federal and State of Minnesota programs have encouraged agricultural land retirement through the Conservation Reserve Program (CRP) (Lant, 1991), the Reinvest in Minnesota Program (Korczak and Gran, 1986), and the Conservation Reserve Enhancement Program (Smith, 2000). In the Minnesota River Basin, agricultural land taken out of production through conservation programs typically is planted in native grasses. Retired agricultural land cover may be important to water quality, aquatic habitat, reduction of sediment, and reduction of overland runoff.

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Abbreviations: CRP, Conservation Reserve Program; IBI, index of biotic integrity; IDAS, Invertebrate Data Analysis System; MANCOVA, multivariate analysis of covariance; MPCA, Minnesota Pollution Control Agency; NLCD, National Land Cover Data; STATSGO, State Soil Geographic database; TSS, total suspended solids.

Fig. 1. The Minnesota River Basin and location of sites sampled during 2001 to 2003.

Agricultural land retirement programs have been successful at taking environmentally sensitive land out of agricultural production (Minnesota Board of Water and Soil Resources, 2004, 2005), and agricultural land retirement is directly correlated with improved stream quality in the Minnesota River Basin (Christensen et al., 2009). Information of this type is critical for water-resource managers when making decisions on placement of retired land to maximize water and aquatic-resource quality. The U.S. Geological Survey, in cooperation with the Minnesota Board of Water and Soil Resources, conducted a quantitative assessment of the effect of retiring agricultural land and replacing it with native grasses on the aquatic community in three subbasins of the Minnesota River Basin (Christensen et al., 2009). Although assessments of this type on a narrow scale have shown significant results (Christensen et al., 2009), a broader study focusing on a larger spatial scale was needed to understand the relation between retired agricultural land and the aquatic-community health in the Minnesota River Basin.

Background

Much of the Minnesota River Basin is planted in row crops, such as corn, soybeans, and sugar beets (32% corn, 29% soybeans, and 1% sugar beets [U.S. Department of Agriculture, 2002]). In 1996, research showed that most crops in the Minnesota River

Basin were harvested in the fall, leaving the fields bare and prone to erosion (University of Minnesota, 1996). Crop residue protects soil from erosion (Karlen et al., 2009), and producers have increasingly used various conservation tillage practices, such as no-till or strip-till, in which some crop residue is left on the field (Randall and Vetsch, 2006).

Annual precipitation in the Minnesota River Basin is about 61 cm (University of Minnesota, 2009) and generally ranges from about 56 cm yr^{-1} in the western part of the basin to about 79 cm yr^{-1} in the eastern part of the basin. In poorly drained areas with high rainfall where sedimentation is a problem, management practices other than tillage may be required to reduce sediment losses (Randall et al., 1995). Riparian buffers (the zone of vegetation adjacent to the stream) have been proposed as an effective method of safeguarding a stream and its aquatic organisms against excess sediment and nutrient loading. Riparian buffers and grass filters slow overland sediment transport (Mankin et al., 2007; Komor and Hansen, 2003), attenuate nutrients through plant uptake (Mankin et al., 2007; Gregory et al., 1991), and provide channel shading (Gregory et al., 1991).

In addition to riparian buffers, upland areas that are environmentally sensitive have been taken out of agricultural production. Land adjacent to lakes and wetlands, marginal pastureland, hillsides, and areas with substantial risk of groundwater contamination qualify for certain agricultural land retirement programs. In this article, land that is within 400 m of a stream or river is considered riparian; upland areas are those that are not directly adjacent to a river or stream but may be connected to a lake or wetland.

Research has shown that riparian buffers have improved water quality and have provided benefits to fish (Lee et al., 2001; Stauffer et al., 2000) and wildlife (Haufler, 2005). Mayer et al. (2006) concluded that riparian buffers are effective at improving water quality by reducing nitrogen levels in groundwater and streams. Although the effectiveness of riparian buffers may be lessened in subsurface tile-drained areas where the tile bypasses the buffers, these buffers may reduce sediment-laden snow from reaching the ditches and provide a setback, preventing field operations from occurring near ditches or other surface-water bodies. Buffers also can moderate peak flows, which in turn helps moderate water temperature (Stewart et al., 2006), an important component of fish habitat (Flint and Flint, 2008). Numerous wildlife species use CRP land (Burger, 2005; Clark and Reader, 2005), and benefits to fish through enhanced water quality have been documented (Allen, 2005). The evidence of improvement is particularly abundant for bird populations (Ferrand and Ryan, 2005; Johnson, 2005; Reynolds, 2005).

Although benefits have been reported by the authors above, responses to land retirement programs vary (Haufler, 2005), and studies have been confined to small areas (Johnson, 2005) or individual sub-basins (Christensen et al., 2009). There is a need for more studies to look at conservation programs on a basin level (Allen, 2005). In 1990–1992 and 2001, fish communities from 31 Minnesota River Basin sites were sampled as part of the Minnesota River Assessment Project (Feist and Niemela, 2002). A comparison between fish index of biotic integrity (IBI) scores from 1990 to 1992 and 2001 showed no significant increase in IBI scores despite a decade of restoration efforts. Feist and Niemela (2002) suggest that elapsed time may not have been sufficient to improve biological conditions. Changes or improvements in water quality may not occur for several years after land is retired because plant communities need time to become established (e.g., Meals and Dressing, 2010). Therefore, more recent data may provide information related to the length of time needed for improvement in biological conditions.

Christensen et al. (2009) demonstrated that total nitrogen concentrations were highest at sampling sites with little to no agricultural land retirement in the basin and lower at sampling sites that had a higher percentage of agricultural land retirement in the basin. However, total phosphorus concentrations were not directly related to differing percentages of agricultural land retirement. Fish IBI scores increased as agricultural land retirement percentages in the local riparian area increased. Data and analysis from Christensen et al. (2009) and Feist and Niemela (2002) were used as a foundation for this evaluation of the association between agricultural land retirement programs and stream quality throughout the Minnesota River Basin.

Objectives

The objectives of this study were to determine if agricultural land retirement was related to differences in water quality and aquatic-community health among the streams of the Minnesota River Basin by (i) analyzing retired land at several spatial scales defined by distance from the streams on water quality and aquatic-community health and by (ii) examining other basin, riparian, or in-stream variables that were related to water quality and aquatic-community health. This study will help scientists and basin managers understand factors influencing aquatic communities in the Minnesota River Basin by documenting the relations between agricultural land retirement and aquatic-community health.

Materials and Methods

The Minnesota Pollution Control Agency (MPCA) maintains a network of biological monitoring sites. The results of water quality, fish, invertebrate, and physical habitat sampling from 334 sites in the Minnesota River Basin were obtained from the MPCA in June 2007. The data were collected from July 1990 to October 2006 during the open water season, generally from June through September. From this initial population of sites, many were omitted from the dataset. Sites with drainage areas outside the State of Minnesota were eliminated, including many sites on the main stem of the Minnesota, Blue Earth, and Lac Qui Parle Rivers. Certain sites were eliminated to reduce factors that might confound the effect of land retirement programs. For example, sites on intermittent streams, sites near tributaries to lakes, and sites less than 25 km downstream from a wastewater treatment plant were excluded. Sites within 3.65 km of another site on the same stream were excluded from the database. Because the majority of the land retirement contracts commenced in 1998, all data collected before 1997 were excluded. For sites sampled more than once, only data for the latest sample collection date were retained. Finally, only sites with water quality, fish, and physical habitat data collected in the same sampling season were retained. Data from 82 sites in the Minnesota River Basin, sampled in June, July, or August between 2001 and 2003, were used for the quantitative analysis (Fig. 1; Table 1).

Water Quality and Aquatic Biology Data Collection

Water quality sampling was conducted by the MPCA immediately before fish sampling at all sites according to sampling protocols established by the MPCA (Minnesota Pollution Control Agency, 2002). Samples were collected from a single point judged to represent the water quality of the total instantaneous flow at the cross section. Samples were analyzed by the Minnesota Department of Health for ammonia, total nitrogen, total phosphorus, and total suspended solids (TSS) using U.S. Environmental Protection Agency methods EPA 350.1 (U.S. Environmental Protection Agency, 1993a), EPA

Table 1. Selected Minnesota River Basin sampling sites, watershed areas, and sample collection dates, 2001–2003. Sites are in order of increasing watershed area.

MPCA† site no.	Stream name‡	Watershed area	Fish, habitat, and water sample collection date	Invertebrate collection date	MPCA site no.	Stream name	Watershed area	Fish, habitat, and water sample collection date	Invertebrate collection date
		km^2					km^2		
01MN004	Co. Ditch #70	4.26	12 July 2001	12 Sept. 2002	01MN012	Middle Br Rush R	179	9 July2001	18 Sept. 2001
03MN066	Co. Ditch # 44	6.94	29 July 2003	25 Aug. 2003	03MN002	Pomme de Terre R	190	5 Aug. 2003	18 Aug. 2003
03MN035	Dutch Charley Cr	19.2	9 July 2003	26 Aug. 2003	03MN030	Carver Cr	191	11 Aug. 2003	18 Aug. 2003
01MN028	Co. Ditch #42	20.6	28 June 2001	17 Sept. 2002	03MN074	Le Sueur Cr	194	22 July 2003	20 Aug. 2003
01MN053	Judicial Ditch #33	20.6	26 June 2001	12 Sept. 2001	03MN038	S Br Yellow Medicine R	196	26 June 2003	26 Aug. 2003
90MN008	Signalness Cr	21.1	8/27/2001	13 Sept. 2001	03MN026	Judicial Ditch # 1A	197	11 Aug. 2003	26 Aug. 2003
01MN007	Highwater Cr	22.2	27 June 2001	5 Sept. 2001	03MN067	Rice Cr	201	8 July 2003	25 Aug. 2003
91MN112	Robert Cr	25.4	16 Aug. 2001	17 Sept. 2002	01MN042	Cottonwood R	202	14 Aug. 2001	5 Sept. 2001
01MN008	East Cr	26.9	24 July 2001	17 Sept. 2001	03MN069	Nicollet Cr	204	16 July 2003	25 Aug. 2003
01MN011	Spring Cr	27.9	27 June 2001	13 Sept. 2001	03MN024	Middle Br Rush R	209	14 July 2003	26 Aug. 2003
01MN038	Judicial Ditch #30	30.3	28 June 2001	4 Sept. 2001	03MN025	S Br Rush R	216	14 July 2003	23 Aug. 2003
91MN059	Co. Ditch 46A	31.1	26 July 2001	18 Sept. 2001	03MN039	S Br Yellow Medicine R	220	26 June 2003	26 Aug. 2003
01MN040	trib to Le Sueur R	34.0	26 July 2001	10 Sept. 2001	01MN061	Ten Mile Cr	229	18 July 2001	26 Sept. 2001
03MN021	Co. Ditch #42	35.9	16 July 2003	9 Sept. 2003	92MN083	Center Cr	238	25 July 2001	12 Sept. 2001
03MN058	N Fork Nine Mile Cr	37.0	12 Aug. 2003	18 Aug. 2003	03MN018	W Fork Beaver Cr	254	16 July 2003	18 Aug. 2003
01MN014	Rice Cr	41.1	11 July 2001	10 Sept. 2001	03MN027	N Br Rush R	254	14 July 2003	26 Aug. 2003
03MN059	S Fork Nine Mile Cr	42.6	12 Aug. 2003	18 Aug. 2003	03MN004	Little Chippewa R	282	23 July 2003	19 Aug. 2003
03MN022	Co. Ditch # 56	50.4	14 July 2003	25 Aug. 2003	03MN013	Mud Creek	295	24 July 2003	20 Aug. 2003
91MN056	Little Cottonwood R	54.7	13 Aug. 2001	6 Sept. 2001	01MN051	Stony Run	312	16 July 2001	13 Sept. 2001
01MN054	E Br Blue Earth R	55.7	12 July 2001	10 Sept. 2001	92MN025	Redwood R	338	1 Aug. 2001	5 Sept. 2001
90MN117	Credit R	59.8	8 Aug. 2001	17 Sept. 2002	01MN041	Chetomba Cr	350	3 July 2001	13 Sept. 2001
01MN021	Canby Cr	67.4	22 Aug. 2001	26 Sept. 2001	01MN032	Muddy Cr	365	19 July 2001	13 Sept. 2001
01MN025	E Fork Beaver Cr	68.8	9 July 2001	6 Sept. 2001	90MN099	S Fork Watonwan R	484	8 Aug. 2001	12 Sept. 2002
90MN111	Buffalo Cr	71.0	26 July 2001	17 Sept. 2002	03MN070	Le Sueur R	484	7 July 2003	25 Aug. 2003
03MN037	Three Mile Cr	73.5	30 June 2003	20 Aug. 2003	03MN012	E Br Chippewa R	530	23 July 2003	20 Aug. 2003
03MN056	trib to Chippewa R	77.7	25 June 2003	20 Aug. 2003	03MN034	Dutch Charley Cr	540	10 July 2003	26 Aug. 2003
03MN041	S Br Yellow Medicine R	84.8	30 June 2003	19 Aug. 2003	92MN030	Redwood R	662	31 July 2001	5 Sept. 2001
03MN040	S Br Yellow Medicine R	90.4	30 June 2003	19 Aug. 2003	92MN031	Redwood R	689	1 Aug. 2001	5 Sept. 2001
03MN049	Judicial Ditch # 10	93.1	16 July 2003	9 Sept. 2003	03MN032	Sleepy Eye Cr	700	9 July 2003	26 Aug. 2003
01MN055	Boiling Springs Cr	93.8	26 June 2001	12 Sept. 2001	01MN039	Cobb R	764	11 July 2001	10 Sept. 2001
03MN033	Mound Cr	97.0	9 July 2003	26 Aug. 2003	03MN015	Shakopee Cr	789	20 Aug. 2003	19 Aug. 2003
01MN003	Judicial Ditch #8	98.7	16 July 2001	14 Sept. 2001	03MN003	Pomme de Terre R	933	19 Aug. 2003	18 Aug. 2003
03MN061	Judicial Ditch # 1	104	9 July 2003	25 Aug. 2003	03MN028	Rush R	1040	21 July 2003	9 Sept. 2003
03MN042	N Br Yellow Medicine R	107	5 Aug. 2003	19 Aug. 2003	03MN048	Yellow Medicine R	1140	17 July 2003	25 Aug. 2003
91MN067	Mound Cr	113	13 Aug. 2001	18 Sept. 2001	92MN038	Redwood R	1210	22 Aug. 2001	5 Sept. 2001
03MN014	Judicial Ditch # 8	120	23 July 2003	9 Sept. 2003	92MN041	Redwood R	1320	23 Aug. 2001	5 Sept. 2001
01MN037	Ten Mile Cr	141	2 July 2001	6 Sept. 2001	92MN044	Redwood R	1580	21 Aug. 2001	5 Sept. 2001
92MN022	Redwood R	144	14 Aug. 2001	26 Sept. 2001	92MN049	Redwood R	1810	15 Aug. 2001	5 Sept. 2001
03MN031	Judicial Ditch #30	149	15 July 2003	26 Aug. 2003	03MN009	Chippewa R	1840	20 Aug. 2003	19 Aug. 2003
01MN047	Watonwan R	157	13 Aug. 2001	11 Sept. 2001	03MN010	Chippewa R	1920	20 Aug. 2003	19 Aug. 2003
03MN036	Cottonwood R	178	30 June 2003	20 Aug. 2003	03MN068	Watonwan R	2190	15 July 2003	9 Sept. 2003

† Minnesota Pollution Control Agency.

‡ Br, branch; Co., County; Cr, Creek; E, east; N, north; R, River; S, south; trib, tributary.

351.2 (U.S. Environmental Protection Agency, 1993c), EPA 365.1 (U.S. Environmental Protection Agency, 1993b), and Standard Method 2540D (American Public Health Association et al., 1998), respectively.

Fish community sampling was conducted according to sampling protocols established by the MPCA (Minnesota Pollution Control Agency, 2009). To ensure that a representative sample of the fish community was collected, a stream reach was established by multiplying the average stream width by 35. This stream reach determination was used based on habitat assessment protocol (Simonson et al., 1993; Leopold et al., 1964), indicating that 35 times the stream width usually allows for the inclusion of three riffle/run/pool sequences. A collection pass along the stream reach was then made with one of four types of electrofishing gear depending on stream size. These four types were a Backpack Electrofisher (small wadeable streams less than 8 m across), a Stream Shocker (medium-sized wadeable streams greater than 8 m across), a Miniboom (small to large unwadeable streams and rivers, where poor access did not allow the use of a boat), and a Boom Shocker (larger unwadeable rivers with boat access). All fish captured were identified to the species level by MPCA staff. Two individuals of each fish species were retained for species verification by Dr. Andrew Simons (Bell Museum, St. Paul, MN), and the rest were returned to the stream.

The IBI scores were used to measure fish community response and community health. The original IBI (Karr, 1981) comprised 12 metrics, each with a range of sensitivities to differing types of environmental degradation. The IBI scores for this study were calculated following Minnesota River Assessment Project procedures established by the MPCA specifically for the Minnesota River Basin (Bailey et al., 1993), using 8 of the 12 metrics related to the composition and structure of the fish community. The eight metrics used are described in more detail in Christensen et al. (2009). The IBI score was determined by calculating the biological attributes, converting the attribute values to metric scores, and then adding the metric scores to generate the final index value. The IBI scores can provide an indication that a stream fish community is potentially degraded by environmental stressors. However, the total score cannot predict which individual stressors are causing the response (O'Reilly et al., 2007).

The MPCA Biological Monitoring Unit collected invertebrate samples according to standard operating procedures (Minnesota Pollution Control Agency, 2011). Stream invertebrates were characterized by collecting a composite sample from as many as five different habitat types: hard bottom, aquatic macrophytes, undercut banks, snags, and leaf packs (Minnesota Pollution Control Agency, 2011). Sampling consisted of dividing 20 sampling efforts equally among dominant habitats in the reach. The sample material was preserved immediately with 100% reagent alcohol. Samples were sent to Rhithron Associates, Inc., Missoula, Montana, invertebrate taxonomy laboratory where they were counted and identified to the genus level. Macroinvertebrate metrics were calculated according to Chirhart (2003). Invertebrate data were restructured for analyses in the U.S. Geological Survey Invertebrate Data Analysis System (IDAS) (Cuffney, 2003). The IDAS software was developed to provide an accurate, consistent, and efficient mechanism for analyzing invertebrate data collected as part of the U.S. Geological Survey National Water-Quality Assessment Program. The IDAS program allows the user to calculate abundances, community metrics, diversity, and similarity.

Database Construction Methods

Basins were delineated for each of the 82 sites using hydro-enhanced digital elevation models and a geographic information system (GIS). Three GIS layers were overlaid: National Land Cover Dataset (NLCD), State Soil Geographic database (STATSGO), and a coverage of land retirement programs. The NLCD is composed of 16 land-cover classifications (U.S. Geological Survey, 2003). The NLCD classifications were aggregated into five new categories: barren and developed land, cultivated cropland, pasture/hay/grassland, forest/shrub, and open water/wetlands.

A GIS STATSGO soil layer was used to derive soil drainage information for each basin. STATSGO is divided into seven distinct soil hydrology classes based on similarities in soil drainage and type (Natural Resources Conservation Service, 2003). The seven hydrological classes were reclassified into six soil hydrology classes to combine mixed hydrology classes and to add a class for open water. Following the methods established by Christensen et al. (2009), the basins were overlaid onto a reclassified STATSGO layer from which a percentage of each class was derived. For the statistical analyses in this report, three soil hydrology classes were used: soil hydrology class C (slow infiltration rates and fine soils with layers impeding downward movement of water), soil hydrology class D (very slow infiltration rates and clayey soils with a high water table), and soil hydrology class M (mixed hydrology or areas of combined A/D, B/D, and C/D soil hydrology classes).

Conservation Reserve Program data for Minnesota were obtained in 2007 from the Farm Services Agency (St. Paul, MN) and included a GIS layer of land with CRP contracts. Similar agricultural land retirement data were obtained for the Conservation Reserve Enhancement Program and Reinvest in Minnesota Program from the Minnesota Board of Water and Soil Resources. Some smaller amounts of retired agricultural land in other programs, such as nonwetland Wildlife Management Areas and U.S. Fish and Wildlife Service Waterfowl Production Areas, were obtained from the Minnesota Department of Natural Resources (St. Paul, MN). All data were overlaid in a GIS to form a single-part layer of retired agricultural land and then clipped to the basins. The percentage of retired land within each basin was calculated according to the methods established by Christensen et al. (2009). A GIS model was developed to create riparian zones of 400, 300, 200, 100, and 50 m bordering all streams in the basins. The riparian zones were then overlaid with the agricultural land retirement data to calculate an intensity of retired land within each basin and each riparian zone.

Statistical Methods

Spearman's rho correlations (Helsel and Hirsch, 1992; Tibco Software, 2008) were used to analyze agricultural land retirement at several spatial scales (objective 1). The amount of agricultural land retirement in each sub-basin and within each of the riparian zones (independent variables) was compared with each water quality, fish, and invertebrate variable (dependent variables) using a significance level of $p = 0.05$.

Basin, riparian, and in-stream variables commonly are interrelated and influence several aquatic-community measures. Therefore, to address our second objective, to determine other physical factors that influence the efficacy of retired land in influencing aquatic-community health, a series of multivariate analysis of covariance (MANCOVA) models were performed on the data (Tibco Software, 2008).

The MANCOVA models were used to answer the question of whether water quality and aquatic-community health were related to agricultural land retirement despite relations to other environmental factors. For these models, each site was placed into a retired land intensity category on the basis of percentage of agricultural land retirement in the 50-m riparian zone, either high intensity (>10%) or low intensity (<10%). An overall MANCOVA on the water quality data is not included because the water quality data did not meet the assumption of correlation among dependent variables. The multivariate test statistics Wilk's lambda (Λ), F-statistic, and *p* value are used to evaluate MANCOVA. Wilk's lambda expresses the proportion of unexplained variance in the dependent measures (Grimm and Yarnold, 1995). Univariate ANCOVA models were computed with a single dependent variable (Tabachnik and Fiddell, 2007) following the MANCOVA analysis. Univariate ANCOVAs can identify individual variables that are not significant (Grimm and Yarnold, 1995), though a significant multivariate effect exists.

Analysis of covariance (and similarly the MANCOVA) is a parametric test (Helsel and Hirsch, 1992). Most parametric tests have the underlying assumption that the data are normally distributed. Some of the variables (for example, cultivated cropland basinwide and fish species with deformities, lesions, or tumors) did not meet the assumption of normality as defined by the Shapiro-Wilk test (Tibco Software, 2008). Variables were not transformed because the transformations did not always improve normality. For this data analysis, MANCOVA and ANCOVA and other parametric tests that are commonly used to assess water quality response (Bishop et al., 2005; Johnson et al., 2009) were chosen because there are no equivalent nonparametric tests. The resulting test statistics are valid; however, *p* values are not exact.

Results

Water Quality

Of the 24 possible correlations between agricultural land retirement (six variables) and water quality (four variables), only total nitrogen concentration was significantly negatively correlated (rho = –0.449; *p* = 0.000) to agricultural land retirement at the basin level (Table 2). Ammonia, total

Table 2. Spearman's rho correlations between land retirement variables and water quality or fish variables, Minnesota River Basin, 2001–2003.

	Agricultural land retirement percent					
	50-m riparian zone	100-m riparian zone	200-m riparian zone	300-m riparian zone	400-m riparian zone	Basinwide
Total ammonia						
rho	0.016	0.080	0.124	0.137	0.136	0.127
p value	0.882	0.474	0.264	0.216	0.221	0.252
Total N						
rho	–0.236	–0.341	–0.411	–0.423	–0.432	–0.449
p value	0.034*	0.002*	0.000*	0.000*	0.000*	0.000*
Total P						
rho	0.016	0.080	0.124	0.137	0.136	0.078
p value	0.882	0.474	0.264	0.216	0.221	0.482
TSS						
rho	0.200	0.240	0.259	0.272	0.258	0.210
p value	0.072	0.031*	0.020*	0.014*	0.020*	0.058
Fish index of biotic integrity						
rho	0.286	0.272	0.249	0.244	0.240	0.201
p value	0.010*	0.014*	0.025*	0.028*	0.031*	0.070
Native species, %						
rho	0.276	0.319	0.342	0.364	0.357	0.333
p value	0.013*	0.004*	0.002*	0.001*	0.001*	0.003*
Tolerant species, %						
rho	–0.230	–0.162	–0.129	–0.143	–0.143	–0.126
p value	0.038*	0.146	0.245	0.197	0.197	0.257
Simple lithophilic spawning species, %						
rho	0.309	0.299	0.267	0.247	0.246	0.183
p value	0.005*	0.007*	0.016*	0.026*	0.027*	0.100
Species with deformities, lesions, and tumors, %						
rho	0.334	0.348	0.343	0.351	0.354	0.321
p value	0.003*	0.002*	0.002*	0.002*	0.001*	0.004*
Number per minute of tolerant species						
rho	0.031	–0.001	–0.024	–0.019	–0.017	–0.005
p value	0.782	0.996	0.832	0.863	0.875	0.963

* Significant at the 0.05 probability level.

phosphorus, and total suspended solids (TSS) concentrations were not related to agricultural land retirement on a basin level. Total nitrogen was negatively correlated to agricultural land retirement in the 50- to 400-m riparian zones, and TSS concentration was positively correlated to the amount of agricultural land retirement in the 100- to 400-m riparian influence zones. The negative correlation indicates a decreasing total nitrogen concentration with increasing land retirement, whereas a positive correlation indicates an increasing TSS concentration with increasing land retirement. Spearman's rho correlations for all 33 independent variables and 27 dependent variables are included in the supplementary material.

Aquatic-Community Health

Of the 36 possible correlations between agricultural land retirement (six variables) and fish community attributes (six variables), 23 were significant ($p < 0.05$): 22 were positive, and one was negative. Retired land within the 50-m riparian zone was related to the most fish response variables (five variables).

Fish IBI scores ranged from 8 to 75. Larger numbers indicate better aquatic resource quality than lower numbers. The relation between IBI scores and land retirement was statistically significant (p = 0.010–0.031) for the 50- to 400-m riparian zones but was not significant for the percentage land retirement in the basin (rho = 0.201; p = 0.070).

Percentage of native fish species and species with deformities lesions and tumors were significantly correlated to all land retirement variables. Percentage of native species had a stronger relation for the larger spatial scales (p = 0.001–0.004) than for the 50-m riparian zone (p = 0.013). Tolerant fish species had a negative correlation with land retirement in the 50-m riparian zone (rho = –0.230; p = 0.038), with no significant correlation at the other spatial scales. Simple lithophilic spawning fish species were significantly correlated to land retirement in the 50- to 400-m riparian zones with a stronger relation (p = 0.005) at the 50-m riparian zone and gradually decreasing in strength at the larger spatial scales (p = 0.007–0.027). The collection effort (number collected per minute) of tolerant species was not correlated to land retirement at any spatial scale.

Findings from the Spearman's analysis support the idea that agricultural land retirement is related to fish community composition in the Minnesota River Basin. Results indicate that the percentage of agricultural land retirement adjacent to streams has a positive influence on fish community composition as indicated by a positive association with the fish IBI score.

Agricultural land retirement was not correlated to any invertebrate community attributes (17 variables), and these variables are not shown in Table 2. Invertebrate community attributes were primarily influenced by in-stream conditions, such as streambed substrate (see supplementary material), rather than broader land-use conditions, such as percentage of agricultural land retirement.

MANCOVA and ANCOVA Models

Total ammonia concentration was significantly related to four independent variables (ANCOVA): (i) average number of shaded densiometer points (p = 0.027), (ii) average percentage of emergent macrophytes per site (p = 0.028), (iii) average percentage of undercut bank (p = 0.043), and (iv) substrate type at the thalweg (p = 0.004) (Table 3). Total nitrogen concentrations were significantly related to four independent variables: (i) retired lands intensity category (p = 0.022), (ii) percentage of lacustrine area basinwide (p = 0.000), (iii) percentage of palustrine area basinwide (p = 0.009), and (iv) hydrology class M basinwide (p = 0.002). A disadvantage of an ANCOVA analysis is that it does not indicate whether the relation is positive or negative. The Spearman's correlation described previously indicated that the relation between retired land intensity and nitrogen concentration is negative: when retired land percentages increased, nitrogen concentrations decreased. Total phosphorus concentrations were significantly related to drainage area (p = 0.002) but were not related to any other independent variable. Total suspended solids concentrations were significantly related to two independent variables: (i) drainage area (p = 0.000) and (ii) percentage of soil hydrology class D basinwide (p = 0.015).

Two MANCOVA models were performed, one to examine the independent variables that affect fish health and one to examine the independent variables that affect invertebrate health (Table 4). The MANCOVA allows the treatment of all response variables (fish IBI score, percentage of native species per site, etc.) at the same time. Results of the first MANCOVA model indicated that five independent variables were related to fish health: (i) drainage area (L = 0.442; F = 11.0), (ii) retired land intensity category (Λ = 0.714; F = 3.47), (iii) percentage of cultivated cropland basinwide (Λ = 0.791; F = 2.28), (iv) standard deviation of water depth (Λ = 0.787; F = 2.34), and (v) percentage of average submergent macrophytes (Λ = 0.777; F = 2.49). Results of the second MANCOVA model indicated that six independent variables were significantly related to invertebrate health: (i) drainage area (Λ = 0.430; F = 3.20), (ii) percentage of lacustrine area basinwide (Λ = 0.545; F = 2.01), (iii) percentage of palustrine area basinwide (Λ = 0.539; F = 2.06), (iv) sinuosity (Λ = 0.547; F = 2.00), (v) standard deviation of water depth (Λ = 0.337; F = 4.75), and (vi) percentage of average submergent macrophytes (Λ = 0.518; F = 2.24).

Because a MANCOVA might show a significant multivariate effect when there is no univariate effect (Grimm and Yarnold, 1995), MANCOVA models were followed by univariate ANCOVA models for each dependent variable (Tables 5 and 6). Univariate ANCOVAs are not included for dependent variables that showed no significant effect (total abundance composed of scrapers, total abundance composed of diptera, and total abundance composed of oligochaeta). By performing the MANCOVA models first, the risk of Type 1 error (false positives) was reduced. Any relation between the dependent variables is ignored with an ANCOVA model (e.g., the relation between native fish species percentage and fish IBI score). Results of the ANCOVA model for IBI scores (Table 5) indicated that retired land intensity category (p = 0.009) and percentage of cultivated cropland basinwide (p = 0.001) were as significantly related to the univariate response (IBI score; Table 5) as they were to the multivariate response (all fish-response variables; Table 4). For native species percentage, drainage area (p = 0.000) and retired land intensity category (p = 0.007) were significantly related. Other fish response variables were related to drainage area (percentage of tolerant species) or intensity category (percentages of simple lithophilic spawning fish species and species with deformities, lesions, and tumors per site), but no other fish response variables were related to both drainage area and intensity category.

Univariate ANCOVA models performed on invertebrate health metrics (Table 6) varied from no relation with basin, riparian, or in-stream variables (total abundance composed of scrapers, total abundance composed of diptera, and total abundance composed of oligochaeta) to as many as nine statistically significant independent variables (percentage abundance of tolerant species). The most common significant variables for invertebrate health metrics with the univariate ANCOVA models were drainage area, followed by standard deviation of water depth, confirming the MANCOVA model. However, the next most significant independent variable was gradient, which was not indicated in the MANCOVA model.

Discussion

The first objective of this study, to analyze agricultural land retirement relative to proximity to streams, was achieved through the analysis of agricultural land retirement percentage in riparian zones. In general, agricultural land retirement percentages are greater close to streams. This was anticipated because agricultural land retirement programs often target the sensitive land next to water bodies.

The strong negative relation between total nitrogen and percentage of retired land (Table 2) was not unexpected because land that is planted in native grasses tends to capture solids and sediment (and related nutrients) in runoff before it reaches the stream. Mankin et al. (2007) reported that riparian buffers consisting of native grasses and shrubs removed sediment, nutrient, and chemical pollutants from upland surface runoff. Although data for this study were collected during base flow conditions, residual concentrations of total nitrogen from surface runoff could be apparent during base flow. Total nitrogen commonly is sorbed to soils and, when associated with solids, is trapped by native grasses. Native grasses in programs such as CRP also are present in the landscape year round, in contrast to farm fields that may be bare for part of the year and therefore susceptible to erosion. Total nitrogen may be negatively related to retired land partially because native grasses are not fertilized and row crops are fertilized. The relation between total nitrogen was not as strongly correlated to land retirement in the 50-m riparian zone as land retirement at larger spatial scales (100-m through basinwide), which may indicate that narrow riparian zones may not be as effective for removal of total nitrogen as wider riparian zones.

Total phosphorus and TSS, a measure of solids and sediment in the stream, were not related to agricultural land retirement at

Table 3. Results of univariate analysis of covariance models examining independent variables and water quality variables.

Independent variable	Total NH_4 concentration		Total N concentration		Total P concentration		TSS†	
	F	*p* Value	F	*p* Value	F	*p* Value	F	*p* Value
Retired land intensity category (high or low)‡	97.0	0.992	5.58	0.022*	0.285	0.595	0.851	0.360
Basin variables								
Drainage area, km^2	2.33	0.132	3.02	0.088	11.1	0.002*	26.9	0.000*
Lacustrine (lake) area basinwide, %	2.65	0.109	19.6	0.000*	0.226	0.637	1.15	0.288
Palustrine (wetland) area basinwide, %	1.65	0.204	7.28	0.009*	0.582	0.449	0.075	0.786
Barren and developed land basinwide, %	0.068	0.795	1.58	0.214	0.060	0.808	0.302	0.585
Cultivated cropland basinwide, %	0.416	0.521	2.70	0.106	0.001	0.975	0.800	0.375
Forest and shrub land basinwide, %	0.270	0.605	0.570	0.453	0.654	0.422	0.578	0.450
Soil hydrology class C basinwide, %	0.750	0.390	3.82	0.056	1.92	0.171	0.912	0.344
Soil hydrology class D basinwide, %	0.590	0.446	2.66	0.108	0.053	0.818	6.30	0.015*
Soil hydrology class M basinwide, %	0.093	0.762	10.9	0.002*	1.86	0.178	0.029	0.866
Riparian variables								
Sinuosity	1.09	0.301	0.146	0.704	0.017	0.897	2.72	0.105
Gradient, m km^{-1}	1.93	0.170	0.153	0.697	0.872	0.354	0.094	0.761
Avg. stream width, m	0.546	0.463	2.55	0.116	2.51	0.119	0.195	0.660
SD of stream width, m	0.007	0.932	0.002	0.963	0.498	0.483	0.330	0.568
Avg. water depth at the thalweg, cm	0.190	0.664	0.787	0.379	0.017	0.898	0.500	0.482
SD of water depth at the thalweg, cm	0.286	0.595	0.260	0.612	0.087	0.769	3.81	0.056
In-stream variables								
Avg. number of shaded densiometer points	5.18	0.027*	0.001	0.981	0.011	0.917	2.88	0.095
Avg. submergent macrophytes, %	0.321	0.573	0.220	0.641	2.63	0.110	2.93	0.092
Avg. emergent macrophytes, %	5.06	0.028*	0.068	0.796	2.20	0.144	0.139	0.711
Avg. undercut bank, %	4.28	0.043*	0.146	0.703	0.749	0.390	0.797	0.376
Avg. overhanging vegetation, %	0.284	0.596	3.00	0.088	1.84	0.181	0.130	0.719
Avg. woody debris, %	0.687	0.411	2.55	0.116	2.39	0.128	0.150	0.700
Avg. of the % of boulders along transect	0.001	0.976	0.880	0.352	1.22	0.275	0.422	0.519
Substrate type at thalweg	9.12	0.004*	0.005	0.945	1.27	0.265	1.29	0.260

* Significant at the 0.05 probability level.

† Total suspended solids.

‡ Retired lands intensity category is on the basis of the percentage of retired land in the 50-m riparian zone.

the basin level. Young and Briggs (2008) suggested that buffers were effective at reducing available (or dissolved) phosphorus concentrations, but the effect on particulate (or total) phosphorus concentrations was unclear. Another theory is that phosphorus may respond slowly to land retirement, perhaps due to desorption of the sediment that has been stored in the channel. However, studies showing temporal change in land retirement compared with total phosphorus concentrations are lacking. Alternatively, there may be differences in surface geology and thus differences in phosphorus and solids reaching the stream. Total suspended solids concentrations were positively correlated to land retirement in the 100- to 400-m riparian zones. This may be a reflection of the soil type in the basins because soil hydrology class D was positively correlated with TSS concentration (see supplementary material). For this study, base flow conditions were targeted for all sites, and TSS concentrations generally were low, with a few outliers (range, 1–170 mg L^{-1}; median, 15 mg L^{-1}). Schottler and Engstrom (2002) found that near-channel sediment sources can be significant in the Minnesota River Basin. It is possible that channel conditions and soil hydrology type have a greater relation to TSS concentration than to land retirement. A reduction in TSS or sediment may be evident in areas with similar soil types.

Correlations for some of the fish variables with land retirement were stronger in the riparian zones of 50 to 400 m (fish IBI and simple lithophilic spawning fish; Table 2) than in the basin, indicating that land retirement in the riparian areas may have a greater effect on fish health than does land retirement at the basin level. Invertebrate variables were not related to land retirement percent at any of the spatial scales, indicating that other factors are affecting invertebrate health measures.

The second objective of the study was to determine whether factors other than retired land variables influence aquatic-resource quality. To capture the inherent complexity in these biological systems, multivariate models have been used to explain these interactions (O'Reilly et al., 2007). The MANCOVA and subsequent ANCOVA models identified several other basin, riparian, and in-stream variables that were related to water quality and aquatic community health.

The relation between fish-response variables and agricultural land retirement is complex. Many of the streams in the study area have been straightened for a variety of reasons related to enhanced drainage. Channel straightening results in removal of substrate, pool and riffle areas, and riparian vegetation, potentially resulting in a reduction in the diversity of aquatic life (Watson et al., 1999). Riparian conditions, such as sinuosity, that

Table 4. Results from multivariate analysis of covaiance models examining watershed, riparian, and in-stream variables related to fish health and invertebrate health among sites in the Minnesota River Basin, 2001–2003.

Independent variable	MANCOVA1† (fish health)				MANCOVA2 (invertebrate health)			
	Wilk's Lambda	F	DF	*p* Value	Wilk's Lambda	F	DF	*p* Value
Retired land intensity category (high or low)‡	0.714	3.47	6, 52	0.006*	0.695	1.06	17, 41	0.422
Basin variables								
Drainage area, km^2	0.442	11.0	6, 52	0.000*	0.430	3.20	17, 41	0.001*
Lacustrine (lake) area basinwide, %	0.805	2.10	6, 52	0.069	0.545	2.01	17, 41	0.034*
Palustrine (wetland) area basinwide, %	0.991	0.079	6, 52	0.998	0.539	2.06	17, 41	0.030*
Barren and developed land basinwide, %	0.974	0.234	6, 52	0.963	0.778	0.689	17, 41	0.795
Cultivated cropland basinwide, %	0.791	2.28	6, 52	0.049*	0.737	0.863	17, 41	0.618
Forest and shrub land basinwide, %	0.872	1.27	6, 52	0.287	0.575	1.78	17, 41	0.065
Soil hydrology class C basinwide, %	0.939	0.560	6, 52	0.760	0.627	1.43	17, 41	0.170
Soil hydrology class D basinwide, %	0.963	0.334	6, 52	0.916	0.619	1.48	17, 41	0.149
Soil hydrology class M basinwide, %	0.867	1.33	6, 52	0.263	0.579	1.75	17, 41	0.071
Riparian variables								
Sinuosity	0.792	2.28	6, 52	0.050	0.547	2.00	17, 41	0.036*
Gradient, m km^{-1}	0.827	1.81	6, 52	0.116	0.605	1.58	17, 41	0.117
Avg. stream width, m	0.811	2.02	6, 52	0.079	0.652	1.29	17, 41	0.249
SD of stream width, m	0.935	0.605	6, 52	0.725	0.765	0.741	17, 41	0.744
Average water depth at the thalweg, cm	0.825	1.84	6, 52	0.110	0.662	1.23	17, 41	0.285
SD of water depth at the thalweg, cm	0.787	2.34	6, 52	0.045*	0.337	4.75	17, 41	0.000*
In-stream variables								
Avg. number of shaded densiometer points	0.879	1.19	6, 52	0.326	0.650	1.30	17, 41	0.242
Avg. submergent macrophytes, %	0.777	2.49	6, 52	0.034*	0.518	2.24	17, 41	0.018*
Avg. emergent macrophytes, %	0.880	1.18	6, 52	0.331	0.595	1.64	17, 41	0.097
Avg. undercut bank, %	0.972	0.248	6, 52	0.958	0.669	1.19	17, 41	0.312
Avg. overhanging vegetation, %	0.803	2.13	6, 52	0.066	0.743	0.837	17, 41	0.645
Avg. woody debris, %	0.851	1.51	6, 52	0.192	0.663	1.23	17, 41	0.287
Avg. of the % of boulders along transect	0.910	0.858	6, 52	0.532	0.585	1.71	17, 41	0.081
Substrate type at thalweg	0.929	0.659	6, 52	0.683	0.749	0.81	17, 41	0.676

* Significant at the 0.05 probability level.

† Multivariate analysis of covariance. MANCOVA1, fish variables; MANCOVA2, invertebrate variables.

‡ Retired land intensity category is on the basis of the 50-m riparian zone.

were positively correlated to fish IBI scores (see supplementary material) may be greater in areas with more agricultural land retirement as the streams are allowed to regain a more natural channel. Simple lithophilic spawning fish species can be sensitive to siltation because they spawn over gravel substrates. These species offer no parental care to the developing embryos. The positive relation between simple lithophils and retired land intensity (Table 5) may be an indication that land retirement reduces the amount of silt and sediment reaching the stream and thus has a positive effect on these sensitive species of fish.

The percentage of species with deformities, lesions, and tumors, a measure of fish health, was significantly correlated with several independent variables, including drainage area. This indicates that agricultural land retirement may positively influence the diversity of fish in the stream, but not necessarily the health of the fish. The percentages of average submergent macrophytes were significantly negatively related to fish variables (Table 4 and supplementary material). Excessive growth of macrophytes in streams is due to nutrient enrichment, and this may limit in-stream habitat for fish and result in lower IBI scores.

This study found no clear correlation between invertebrate community response variables in streams and retired agricultural land variables. Invertebrate response variables were strongly associated with more local physical factors in the riparian area and within the stream, as also observed by Allan and Castillo (1995). Invertebrate response variables that indicate good resource quality, such as the percentage of intolerant species, and percent ephemeroptera, plecotera, and trichoptera increased as riparian characteristics such as sinuosity, gradient, standard deviation of stream width, standard deviation of stream depth, and stream shading increased. These indicators of good quality also increased as in-stream habitat (woody debris and boulders) increased. Similar to fish response variables, the stream reaches with more macrophyte cover resulted in fewer sensitive invertebrate taxa. The more sinuous and shaded channels, with variability in habitat and intact riparian area, likely support more sensitive invertebrate species. These results indicate that stressors that degrade habitat may have a strong influence on invertebrate community composition and that fish may be better indicators of basin and riparian land use than invertebrates.

Table 5. Results of univariate analysis of covariance models examining independent variables and fish health.

Independent variable	Fish index of biotic integrity score		Native species (%)		Tolerant species (%)		Simple lithophilic spawning fish species (%)		Species with deformities, lesions, and tumors (%)		No. per minute of tolerant species	
	F	*p* Value	F	*p* Value	F	*p* Value	F	*p* Value	F	*p* Value	F	*p* Value
Retired land intensity category† (high or low)	7.27	0.009*	7.95	0.007*	2.82	0.099	6.78	0.012*	4.64	0.036*	0.006	0.939
Basin variables												
Drainage area, km^2	1.08	0.304	49.3	0.000*	5.48	0.023*	0.187	0.667	1.93	0.171	0.815	0.370
Lacustrine (lake) area basinwide, %	0.751	0.390	0.358	0.552	8.09	0.006*	3.06	0.086	0.109	0.743	2.88	0.095
Palustrine (wetland) area basinwide, %	0.096	0.758	0.395	0.532	0.019	0.891	0.030	0.864	74.0	0.993	0.019	0.892
Barren and developed land basinwide, %	0.126	0.724	0.018	0.894	0.148	0.702	0.698	0.407	0.192	0.663	0.109	0.743
Cultivated cropland basinwide, %	12.1	0.001*	2.96	0.091	2.23	0.141	0.795	0.376	0.319	0.575	1.28	0.263
Forest and shrub land basinwide, %	1.20	0.279	2.76	0.102	0.863	0.357	0.623	0.433	0.002	0.969	0.140	0.710
Soil hydrology class C basinwide, %	1.08	0.302	0.814	0.371	1.86	0.178	0.247	0.621	0.747	0.391	0.137	0.713
Soil hydrology class D basinwide, %	0.065	0.800	0.212	0.647	0.000	0.984	0.083	0.775	0.336	0.565	1.51	0.224
Soil hydrology class M basinwide, %	4.76	0.033*	2.37	0.129	1.25	0.269	0.012	0.914	1.57	0.215	0.137	0.713
Riparian variables												
Sinuosity	2.58	0.114	4.99	0.029*	6.03	0.017*	1.22	0.275	0.103	0.749	0.439	0.510
Gradient, m km^{-1}	2.67	0.108	0.238	0.628	0.179	0.674	0.903	0.346	3.61	0.063	4.02	0.050*
Avg. stream width, m	0.135	0.714	7.29	0.009*	0.296	0.589	2.08	0.155	2.63	0.110	0.197	0.659
SD of stream width, m	0.811	0.372	0.483	0.490	0.236	0.629	0.233	0.631	0.605	0.440	2.95	0.091
Avg. water depth at the thalweg, cm	2.38	0.128	4.29	0.043*	0.440	0.510	0.316	0.576	0.413	0.523	8.55	0.005*
SD of water depth at the thalweg, cm	1.93	0.170	3.95	0.052	7.99	0.006*	0.140	0.709	1.53	0.221	0.845	0.362
In-stream variables												
Avg. number of shaded densiometer points	0.523	0.472	0.531	0.469	1.34	0.252	0.018	0.893	0.009	0.924	7.20	0.010*
Avg. submergent macrophytes, %	3.46	0.068	4.34	0.042*	1.85	0.179	7.24	0.009	2.74	0.103	0.735	0.395
Avg. emergent macrophytes, %	1.21	0.276	4.30	0.043*	1.08	0.303	1.29	0.261	1.18	0.281	0.180	0.673
Avg. undercut bank, %	0.070	0.792	0.769	0.384	0.234	0.631	0.147	0.703	0.141	0.708	0.298	0.587
Avg. overhanging vegetation, %	0.214	0.646	1.43	0.237	3.23	0.078	3.21	0.078	0.598	0.443	0.423	0.518
Avg. woody debris, %	3.60	0.063	7.03	0.010*	0.474	0.494	0.349	0.557	1.14	0.290	0.945	0.335
Avg. of the % of boulders along transect	3.48	0.067	0.029	0.865	0.675	0.415	0.537	0.467	0.151	0.699	0.001	0.978
Substrate type at thalweg	2.62	0.111	2.34	0.132	2.03	0.160	0.009	0.925	0.159	0.692	1.07	0.305

* Significant at the 0.05 probability level.

† Retired land intensity category is on the basis of the 50-m riparian zone.

Table 6. Results of univariate analysis of covariance models examining independent variables (Ind. var.) and invertebrate health.

Ind. var.	TA composed of filtering-collectors (%)†		TA composed of shredders (%)		A-weighted EPA tolerance value		A of intolerant species (%)		A of tolerant species (%)		TA composed of the most abundant taxon (%)		TA composed of mayflies, stoneflies, & caddisflies (%)		TA composed of mayflies (%)		TA composed of caddisflies (%)		TA composed of odonates (%)		TA composed of coleoptera (%)		TA composed of tanytarsini midges (%)		TA composed of gastropods (%)		TA composed of amphipoda (%)	
	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p
Retired land intensity category‡ (high or low)	0.009	0.924	0.133	0.717	1.72	0.195	2.39	0.128	0.288	0.593	0.291	0.592	2.20	0.144	0.886	0.350	2.09	0.154	0.196	0.659	2.80	0.100	1.26	0.267	0.721	0.399	0.250	0.619
Basin variables																												
Drainage area, km^2	6.89	0.011*	11.7	0.001*	16.6	0.000*	14.8	0.000*	6.96	0.011*	0.110	0.741	6.83	0.011*	1.18	0.282	11.1	0.002*	5.79	0.019*	0.007	0.935	3.10	0.084	5.19	0.027*	2.24	0.140
Lacustrine (lake) area basinwide, %	5.13	0.027*	2.055	0.157	1.61	0.210	1.35	0.249	1.73	0.193	0.012	0.913	1.08	0.303	2.79	0.101	0.373	0.544	0.460	0.501	0.271	0.605	0.019	0.892	1.81	0.183	2.61	0.112
Palustrine (wetland) area basinwide, %	11.3	0.001*	0.640	0.427	11.9	0.001*	5.14	0.027*	12.4	0.001*	5.51	0.022*	3.49	0.067	0.050	0.824	11.2	0.001*	0.205	0.653	0.007	0.932	0.123	0.727	2.19	0.144	0.025	0.874
Barren and developed land basinwide, %	0.527	0.471	0.009	0.927	0.433	0.513	0.534	0.468	0.537	0.467	0.848	0.361	1.15	0.288	2.11	0.152	0.022	0.884	4.06	0.049*	0.447	0.506	2.59	0.113	0.448	0.506	0.764	0.386
Cultivated cropland basinwide, %	0.714	0.402	0.159	0.691	0.781	0.380	1.97	0.166	0.028	0.868	0.061	0.806	1.32	0.255	1.75	0.191	0.039	0.843	3.41	0.070	2.72	0.104	0.045	0.833	1.25	0.268	1.63	0.207
Forest and shrub land basinwide, %	6.89	0.011*	0.184	0.670	7.13	0.010*	3.79	0.056	6.38	0.014*	1.90	0.174	3.11	0.083	1.62	0.209	2.19	0.145	11.4	0.001*	0.266	0.608	0.256	0.615	2.01	0.161	0.799	0.375
Soil hydrology class C basinwide, %)	1.000	0.321	0.449	0.506	0.127	0.722	0.243	0.624	0.490	0.487	0.010	0.923	0.115	0.736	0.028	0.869	0.813	0.371	1.01	0.320	1.64	0.205	8.72	0.005*	2.09	0.154	0.720	0.400
Soil hydrology class D basinwide, %	0.324	0.571	0.024	0.878	1.43	0.237	0.000	0.988	2.24	0.140	0.765	0.386	0.231	0.633	1.32	0.255	0.895	0.348	0.873	0.354	6.52	0.013*	0.015	0.903	0.767	0.385	0.001	0.981
Soil hydrology class M basinwide, %	12.0	0.001*	1.73	0.193	7.50	0.008*	3.25	0.077	8.04	0.006*	0.761	0.387	0.861	0.357	0.116	0.735	5.54	0.022*	1.16	0.286	0.658	0.421	1.90	0.174	1.12	0.293	0.967	0.330
Riparian variables																												
Sinuosity	0.160	0.690	0.091	0.765	6.99	0.011*	1.96	0.167	5.35	0.024*	4.17	0.046*	1.94	0.169	2.87	0.096	0.000	0.989	2.89	0.094	6.65	0.013*	0.725	0.398	0.216	0.644	2.72	0.105
Gradient, m km^{-1}	10.7	0.002*	0.008	0.929	7.14	0.010*	4.79	0.033*	6.71	0.012*	4.72	0.034*	5.56	0.022*	0.724	0.398	11.0	0.002*	2.19	0.145	9.00	0.998	0.738	0.394	4.00	0.999	2.74	0.104
Avg. stream width, m	1.55	0.219	0.141	0.709	3.35	0.073	4.40	0.040*	0.468	0.496	0.957	0.332	3.00	0.088	1.29	0.261	2.46	0.122	1.08	0.303	0.027	0.871	0.852	0.360	3.03	0.087	1.20	0.279
SD of stream width, m	1.28	0.263	0.382	0.539	0.256	0.615	1.13	0.293	0.696	0.407	0.101	0.751	0.234	0.631	2.43	0.124	2.25	0.139	0.006	0.941	0.238	0.627	0.025	0.876	0.000	0.988	1.04	0.311
Avg. water depth at the thalweg, cm	0.636	0.428	2.23	0.141	3.79	0.057	0.550	0.461	5.69	0.020*	0.037	0.847	1.34	0.252	0.805	0.373	0.764	0.386	8.81	0.004*	0.875	0.353	0.570	0.453	0.543	0.464	10.3	0.002*
SD of water depth at the thalweg, cm	13.0	0.001*	0.459	0.501	47.3	0.000*	27.0	0.000*	44.8	0.000*	24.9	0.000*	14.3	0.000*	6.87	0.011*	11.3	0.001*	5.56	0.022*	0.545	0.464	0.124	0.726	3.02	0.088	30.0	0.000*

Table 6. Continued.

Ind. var.	TA composed of filtering-collectors (%)†		TA composed of shredders (%)		A-weighted EPA tolerance value		A of intolerant species (%)		A of tolerant species (%)		TA composed of the most abundant taxon (%)		TA composed of mayflies, stoneflies, & caddisflies (%)		TA composed of mayflies (%)		TA composed of caddisflies (%)		TA composed of odonates (%)		TA composed of coleoptera (%)		TA composed of tanytarsini midges (%)		TA composed of gastropods (%)		TA composed of amphipoda (%)	
	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p	F	p
In-stream variables																												
Avg. n of shaded densiometer points	0.936	0.337	0.053	0.819	3.51	0.066	3.21	0.079	1.99	0.164	0.614	0.436	0.094	0.760	0.031	0.861	0.771	0.384	0.001	0.977	2.59	0.113	0.730	0.396	2.02	0.160	0.455	0.503
Avg. submergent macrophytes, %	0.010	0.922	6.32	0.015*	3.83	0.055	0.661	0.420	3.66	0.061	3.21	0.078	0.084	0.773	0.035	0.853	0.100	0.752	7.17	0.010*	0.467	0.497	1.14	0.290	2.17	0.146	6.25	0.015*
Avg. emergent macrophytes, %	3.76	0.058	1.18	0.283	5.49	0.023*	1.48	0.229	8.33	0.006*	10.3	0.002*	5.71	0.020*	3.23	0.078	3.62	0.062	0.167	0.684	0.008	0.927	1.65	0.204	0.002	0.968	6.53	0.013*
Avg. undercut bank, %	0.271	0.605	0.078	0.781	3.00	0.996	0.025	0.875	0.150	0.700	4.03	0.049*	0.158	0.692	61.0	0.994	0.579	0.450	0.353	0.555	1.71	0.196	0.032	0.858	0.290	0.593	0.403	0.528
Avg. overhanging vegetation, %	1.93	0.170	0.757	0.388	0.571	0.453	0.847	0.361	0.577	0.451	0.830	0.366	1.41	0.240	0.776	0.382	0.960	0.331	3.82	0.056	0.109	0.742	0.825	0.368	1.03	0.314	0.159	0.691
Avg. woody debris, %	0.007	0.932	1.83	0.181	2.07	0.156	1.18	0.282	1.89	0.175	0.043	0.836	0.053	0.819	0.120	0.730	1.07	0.306	3.57	0.064	1.55	0.218	0.110	0.742	0.022	0.883	1.52	0.222
Avg. of the % of boulders along transect	7.26	0.009*	0.155	0.696	0.183	0.670	0.867	0.356	0.200	0.656	0.625	0.432	0.698	0.407	0.687	0.411	8.99	0.004*	2.23	0.140	0.274	0.603	0.079	0.779	0.918	0.342	0.291	0.592
Substrate type at thalweg	1.44	0.235	2.24	0.140	2.67	0.108	0.826	0.367	1.80	0.185	2.18	0.146	1.94	0.170	0.957	0.332	1.52	0.223	1.34	0.252	0.110	0.741	0.312	0.578	0.053	0.818	1.99	0.163

* Significant at the 0.05 probability level.

† TA, total abundance; A, abundance

‡ Retired land category is on the basis of the 50-m riparian zone.

Basin-level variables that were related to fish-community health were drainage area and cultivated cropland basinwide (Table 4). Many fish and invertebrate variables vary with stream size, and drainage area is a good surrogate for stream size. For example, Karr et al. (1986) and Swift et al. (1986) found an increase in the number of fish species with increasing drainage area, and larger streams typically have a more complex habitat (Mathews, 1998). Drainage area was significant for many of the dependent variables; however, it was not significant for fish IBI score (Table 5) because the scoring criteria for IBI take drainage area into account.

Lacustrine and palustrine features are lake and wetland environments. Lacustrine area was related to some fish health metrics (see supplementary material) but not when the fish metrics were combined in the MANCOVA (Table 4). Presumably, basins with large percentage of lacustrine and palustrine features support a stronger biological community. Moreno-Mateos et al. (2008) suggested that reducing landscape homogeneity (in that example, increasing wetlands scattered throughout a basin) would lead to improved water quality. In simple terms, lacustrine and palustrine features provide more diverse habitat for organisms, so basins with more lacustrine and palustrine features should exhibit a greater aquatic-community response.

In-stream variables that commonly were related to aquatic-community health were standard deviation of water depth at the thalweg and substrate type. The standard deviation of water depth may be related to aquatic-community response because a higher standard deviation indicates that water depth is variable, and this may create microhabitats associated with varying velocities that are suitable for certain macroinvertebrate or fish species. Standard deviation of water depth was related to two fish variables—percentage of native species and percentage of tolerant species (Table 3)—and 12 invertebrate variables (Table 4). Substrate types were related to all the fish variables, with the exception of percentage of species with deformities, lesions, and tumors (Table 3), and to numerous invertebrate variables, including percentage abundance of tolerant and intolerant species (Table 4).

One limitation of this study was that only one sampling date was included for water quality at each of the 82 sites, which does not give a complete picture of water quality conditions in these basins. Precipitation during 2001–2003 was about 2 to 20 cm lower than normal from the west to the east side of the basin, based on 1971–2000 climate normals

(High Plains Regional Climate Center, 2012). In addition, because the ideal season to collect fish and invertebrate samples was chosen (June–August), the full range of water quality conditions was not characterized by the samples. Most nutrient loading occurs during spring runoff (Schilling and Zhang, 2004), and this may be the time of year when riparian buffers are most valuable. However, the nutrient concentrations in lower flow periods in late summer are important to consider, especially in systems where invertebrates are not thriving.

A second limitation was that certain variables did not meet the assumption of a normal distribution for the parametric MANCOVA and ANCOVA tests. Therefore, the p values are used as guides for indicating significance but are not exact values. However, the results of the nonparametric Spearman's tests generally confirmed the variables that influence water quality, fish, and invertebrate health. A third limitation was the lack of inventory available for tile drains in the Minnesota River Basin. Tile drains have been shown to affect water quality (Ahiablame et al., 2011) and in turn could have an effect on fish and invertebrate health. An assumption was made that all subbasins had a similar degree of tile drainage. In addition, there may be other factors that influence stream quality (for example, pesticides) that were not measured as part of this study.

The replacement of native vegetation with agricultural crops has caused numerous changes in streams. Removal of streamside vegetation may increase temperature and reduce woody debris (Allan and Castillo, 1995). Agriculture also has increased nutrient concentrations, due to fertilizers and animal wastes, and has increased soil erosion (Allan and Castillo, 1995). These perturbations may be mitigated by the replanting of native grasses. Buffers also can moderate peak flows, which in turn helps moderate water temperature (Stewart et al., 2006), leading to improved stream quality and aquatic habitat.

The results of this study indicate that agricultural land retirement programs may improve stream health as indicated by an increase in IBI scores and other fish attributes and decreased nitrogen concentrations. This may be an indication that when agricultural land retirement is concentrated near the stream, it may have a positive effect on selected water quality measurements and aquatic-community health.

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