Encouraging Farmers to Produce Environmental Benefits from Agriculture

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*The Unit is jointly sponsored by the U.S. Geological Survey Biological Resources Division, the University

of Minnesota, the Minnesota Department of Natural Resources, and the Wildlife Management Institute.

Abstract

Farmers produce many things in abundance for which they receive income, including food, feed, fiber and fuel. Similarly, farmers generate environmental benefits such as improved water quality by reducing soil and nutrient loss and improved wildlife habitat by managing their operation in certain ways. Unfortunately, most farmers receive limited or no compensation for these positive externalities they produce. If we wish to encourage more of these positive externalities, policies need to be established and implemented that compensate producers for the benefits society receives from specific farming practices. This research examined how production practices that provide environmental benefits would affect water quality (nutrient and sediment loss), fisheries populations, and farm income in two distinct study areas in Minnesota.

We developed a computer simulation model to examine the relationship between agricultural practices, water quality, fish communities, and net farm income within two small watersheds. Our analyses focused on a coolwater stream, the Wells Creek watershed in southeastern Minnesota, and a warmwater stream, a sub-watershed of the Chippewa River in western Minnesota. We used the Agricultural Drainage and Pesticide Transport (ADAPT) model in relation to land use to calculate in-stream suspended sediment concentrations using estimates of sediment delivery, runoff, baseflow and stream bank erosion, and quantified the effects of suspended sediment exposure on fish communities. Our focus was to estimate how agricultural Best Management Practices (BMPs – conservation tillage and nutrient management implemented on all cropland, and 100 foot grass buffers along streams) would affect stream fish communities and net farm income, with reference to current conditions. We found a decrease in "lethal" concentrations of suspended sediment on fish in Wells Creek watershed with an increase in conservation tillage, riparian buffers, and permanent vegetative cover. However, land use change in the Chippewa River watershed did not significantly decrease the effects of suspended sediment on the fish community. This difference between watersheds is likely due to differential tolerance to suspended sediment between coolwater and warmwater fish communities and differences in topography, runoff and bank erosion between the two streams.

Introduction

Agricultural practices have altered stream ecosystems across the United States by increasing sediment and nutrient loads, increasing stream temperature, and altering channel morphology, hydrological regime, and composition and abundance of riparian vegetation (Berkman and Rabeni 1987, Schlosser 1991, Poff and Allan 1995, Waters 1995, Wohl and Carline 1996, Allan et al. 1997, Wang et al. 1997, Harding et al. 1998, Schleiger 2000). In the Midwest, row-crop agriculture is currently the leading source of water pollution, a contributing factor to 70% of streams considered impaired in the 1996 National Water Quality Inventory (USEPA 1996). In general, large areas of forest or wetlands in a watershed were associated with healthy stream ecosystems, whereas high amounts of land under cultivation were associated with degraded conditions (Wang et al. 1997).

For streams included in the 1996 National Water Quality Inventory, the most common agricultural pollutant was sediment, which was a contributing factor for 50% of impaired streams. Annual nitrogen and phosphorous losses in surface runoff (the majority of which is transported by eroded sediment) also are greater in cultivated areas than areas with natural vegetation (Timmons and Holt 1977). Although some sedimentation is natural, excessive sedimentation has negative effects on biota (Waters 1995).

Changes in the structure and function of streams as a result of land use practices often result in reduced diversity of fish, a less complex size structure, a higher relative abundance of herbivorous or detritivorous fish, or greater temporal variability in fish abundance (Berkman and Rabeni 1987, Schlosser 1991). Such changes in fish assemblage structure are often a result of reduced habitat heterogeneity, such as the loss of deep pools and decreased complexity of stream substrate, resulting from increased fine sediment loads (Saunders and Smith 1965, Berkman and Rabeni 1987, Paragamian 1989, Schlosser 1991, Richards and Host 1993, Wood and Armitage 1997). However, increases in suspended sediment concentrations can also result in shifts in fish assemblages through both sublethal and lethal effects, such as impaired respiration, reduced feeding rates and growth, reduced tolerance to disease or toxicants, or increased physiological stress (Waters 1995, Newcombe and Jensen 1996).

Hydrological variability in agricultural streams can lead to fish assemblages with more species tolerant to silt, compared to fish assemblages in stable systems (Poff and Allan 1995). Allan et al. (1997)

2

found that runoff volume and increased hydrological variability from storm events increases when cultivated land increases in a watershed, whereas runoff volume decreases with an increase in forest cover. Conversion of permanent vegetation to cultivated areas with bare soil greatly increases runoff and total sediment loss (Olness et al. 1975, Allan et al. 1997, Larson et al. 1997).   However, Best Management Practices (BMPs) such as conservation tillage, crop rotation, winter cover crops, or grassed or forested riparian buffers, can reduce runoff and soil losses dramatically (Larson et al. 1997).

Although it is widely accepted that suspended sediment has negative impacts on fish, and that the severity of the effects increase with increasing sediment concentrations and duration of exposure, few studies have attempted to make quantitative predictions of the effects of suspended sediment on fish assemblages (Newcombe and Jensen 1996).   Expansion of field-scale agricultural models has allowed scientists to predict sediment and nutrient loadings to streams by incorporating within-field hydrological processes and watershed agricultural practices (Gowda et al. 1999a, Westra et al. 2002).

The first goal of this study was to combine estimates of sediment loads from a watershed-scale agricultural model, Agricultural Drainage and Pesticide Transport (ADAPT) (Chung et al. 1992, and modified by Gowda 1996) with information on the effects of sediment on fish assemblages (Newcombe and Jensen 1996) to examine the effects of land use on fish in agricultural landscapes. Secondly, we used the biophysical information from simulated production activities, combined with estimates of production costs and returns, to compare output between the baseline and a BMP scenario. Our objectives were to: 1) use ADAPT to estimate total sediment load in two watersheds in Minnesota (Wells Creek -- a coolwater stream, and a sub-watershed of the Chippewa River -- a warmwater stream); 2) quantify the effects of suspended sediment concentrations and duration of exposure for fish assemblages for both watersheds; 3) compare the effects of suspended sediment on fish between current land use and the BMP scenario; 4) compare the effects on net farm income in both watersheds between current practices and the BMP scenario for each watershed. Our hypothesis was that net farm income and lethal and sublethal effects (as defined by Newcombe and Jensen 1996) of suspended sediment on fish would decrease as land use shifted from conventional row-crop agriculture to implementation of BMPs throughout the watershed.

3

Wells Creek Watershed

Wells Creek is located in the Driftless Area Ecoregion (Omernik and Gallant 1988) in Goodhue County of southeastern Minnesota. The headwaters originate in a relatively flat agricultural area, then flows through a valley bordered by steep bluffs with groundwater input and eventually drains into the Mississippi River. The stream historically supported a coolwater fish assemblage, with low species diversity and naturally reproducing trout populations. We classified Wells Creek as a coolwater stream because it supported trout and in 1997 only exceeded 22ºC about 12 % of the period from late July to early September (Blann et al. 2002). A Minnesota Department of Natural Resources (MNDNR) survey (Anonymous 1999) identified nine fish species in Wells Creek (Table 1). White sucker *(Catostomus commersoni)* and creek chub *(Semotilus atromaculatus)* were identified as the most common species; both are tolerant of high temperature and sediment concentrations (Lyons et al. 1996). Brown trout *(Salmo trutta)* were present but in low abundance, with some natural reproduction noted. Most species in Wells Creek were considered eurythermal, except for burbot *(Lota lota)*, which has a narrow coolwater preference, and brown trout, which prefers coolwater habitats (Table 1). The overall assessment was that stream habitat conditions were fair, with minimal amounts of cover for adult fish. Bank erosion was severe in many sections of the stream, with some eroding banks nearly 40 feet high (Anonymous 1999).

The Wells Creek watershed is dominated by agriculture with 61% of the total watershed area under cultivation, 10% was in grassland or managed pasture primarily for dairy cattle, and 26% of the watershed was forested, mainly on steep slopes and riparian areas (Figure 1). Data from the US Census of Agriculture (USDA 1999) for Goodhue County indicated the majority of cultivated land is under a corn-soybean rotation, followed by small grain-hay rotations, and some "corn on corn" land (which generally goes into small grain after the second year of corn) (Figure 1). It was estimated that approximately 55% of cultivated land was under some type of conservation tillage, based on the past 11 years of data from the Conservation Tillage Information Center (CTIC 1999) for Goodhue County.

Chippewa River Sub-Watershed

Our study focused on a sub-watershed of the Chippewa River drainage, located primarily in Chippewa County, with a small section in Swift County in western Minnesota. The Chippewa River is

classified as a warmwater river, with a diverse fish assemblage and a temperature range of 23ºC to 26ºC in August (Anonymous 1998a). The river drains relatively flat cropland throughout the drainage, which was primarily prairie and wet prairie prior to European settlement, and empties into the Minnesota River near Montevideo. A 1998 MNDNR survey (Anonymous 1998a) identified 19 fish species in the river, with silver redhorse (*Maxostoma anisurum)* and carp *(Cyprinus carpio*) the most common species (Table 2). Walleye *(Stizostedion vitreum*), northern pike *(Esox lucius)* and channel catfish *(Ictalurus punctatus)* were present, although in low abundance.

The Chippewa River watershed is within the Western Cornbelt Plains Ecoregion (Omernik and Gallant 1988) and primarily is in row-crop agriculture, with 81% of the land area in cultivation, 8% in grassland or pasture primarily for beef cattle, and 5% forested (Figure 2). Almost all of the cultivated land is under a corn-soybean rotation, with some land in sugarbeets, and approximately 1,300 acres in small grains with hay (USDA 1999). Data for Chippewa County from CTIC (1999) indicated approximately 30% of cropland is under some form of residue management system.

Methods

ADAPT Modeling

ADAPT is a field-scale water table management model that combines GLEAMS (Leonard et al. 1987) and DRAINMOD (Skaggs 1978). ADAPT can model crop fields that have tile drainage, a dominant feature of fields in the Chippewa River sub-watershed and an increasingly prevalent feature of some areas of the Wells Creek sub-watershed. Land cover, agricultural management practices (crops, rotation and tillage), slope and soil information were overlaid with Geographic Information System (GIS) to create data input files for the ADAPT model that reflected the spatial distribution of current production practices in the watershed. ADAPT has been calibrated to the data collected in several watersheds in Minnesota (Davis et al. 2000, Dalzell 2000, Johansson 2000, Westra et al. 2002). Davis et al. (2000) calibrated and validated the ADAPT model for tile drainage and associated nitrate nitrogen losses using long-term monitoring data measured on three experimental plots in southern Minnesota. Dalzell (2000) used observed data from six gauged tributary watersheds of the Lower Minnesota River Watershed to calibrate ADAPT to local conditions (land cover, slope, tillage practices, soils information and weather

5

data). This calibrated model was used to simulate monthly flow, sediment and nitrate nitrogen losses from ungauged watershed of the same watershed. Dalzell concluded the watershed methodology applied to ADAPT modeling was best suited for watershed dominated by agricultural activities. Johansson (2000) and Westra et al. (2002) used the ADAPT model calibrated to different watersheds to examine various biophysical and economic effects of policies to reduce nonpoint pollution in the Minnesota River Watershed. ADAPT has provided estimates of monthly stream flows using NRCS soils data, land use, and tillage information for a small agricultural watershed in Ohio (Gowda et al. 1999a).

Field-edge sediment losses and runoff were estimated for each current farming system using the ADAPT model. ADAPT provides edge-of-field estimates for nutrient and soil losses from the different systems, based on soil type, application rates and management techniques, and daily weather data. ADAPT has four components: hydrology, erosion, nutrient, and pesticide transport. The hydrologic component consists of snowmelt, surface runoff, macropore flow, evapotranspiration, infiltration, subsurface drainage, and deep seepage. The remaining components provide estimates of soil loss, nutrient surface and subsurface losses, and pesticide transport through the soil horizon. Weather data were rainfall, temperature, wind speed, relative humidity, and solar radiation. Simulation of all agricultural activities and land uses occurred over a 50-year period (1950 - 1999). Daily temperature and precipitation data were obtained from weather stations in both watersheds from the Historical Data Retrieval and Climate Summaries webpage maintained by the University of Minnesota, Department of Soil, Water and Climate. For Wells Creek, weather information was gathered around Belvidere in Goodhue County, while Big Bend in Chippewa County provided data for the Chippewa subwatershed.

Using a methodology described by Westra et al. (2002), we estimated the sediment delivered to the mouth of each stream with a modification of ADAPT developed by Gowda (1996) to aggregate field-edge estimates across each watershed. For this approach, each watershed was divided into sub-watersheds (termed Transformed Hydrological Response Units or THRUs; Gowda et al. 1999b), an ADAPT simulation was performed for each THRU, a hydrograph was developed for each sub-watershed, the hydrographs were combined, and then routed to the outlet of each watershed.

Estimates were developed for the baseline land use in 1999 and the BMP scenario by simulating different proportions of each land use or farming practice in each watershed. In each watershed, 20

producers or land managers were surveyed on the telephone to identify field locations, crop rotations or livestock systems, production practices, tillage and nutrient practices. These producers were identified by staff from USDA NRCS, Soil and Water Conservation Districts for the two counties, and the Land Stewardship Project office in Chippewa as being representative of the range of conventional and alternative production practices currently being used in the watersheds. Land use in the watershed was assumed to correspond to that of the county in which the watershed was located (i.e., Goodhue for Wells Creek and Chippewa for Chippewa) (Minnesota Land Use and Cover). Information on the crop acreage and livestock numbers from the latest Census of Agriculture (USDA 1999) was combined with the land use data to reflect the predominance and location of various production practices in the watershed.   For each of the agricultural systems analyzed (cropping and livestock, traditional pastured and intensive grazing systems), specific hydrology and erosion files were created. Information in the hydrology and erosion files pertaining to soils was obtained for the predominant STATSGO map units from the MUUF (Map Unit Use File) soils database for the corresponding soils in these watersheds in Minnesota.

Estimated loss of sediments and nutrients in surface runoff and through the tile drainage systems (where tile drainage was appropriate for the soils) was obtained for each system on the soil associations on which they occurred. The Wells Creek sub-watershed was disaggregated into four soil associations while the Chippewa River sub-watershed was disaggregated into five soil associations. These associations (groupings of similar soils) represented over 95 % of the land within the watershed.

Each land manager's representation in a soil association was determined in the following manner. In the survey, land managers indicated the location (county, township and section) of the fields in which they farmed. Each location was identified with one or more soil associations. The total acreage was allocated to specific soil associations based on the proportion of land in each soil association. The resulting acreage allocated to each land manager in a specific soil association was summed. A land manager's proportional representation was the ratio of acreage in a soil association to the total acreage of all land managers in a soil association. Calculating each soil association in this manner established the proportional representation of each land manager in each soil association.

Each soil association was divided into land "close" (within 300 feet) to bodies of water and land "distant" (greater than 300 feet) from water. The proportion of sediment loss that actually reached the

7

mouth of the sub-watershed depended on the delivery ratio associated with the location of the farming activity within the soil association. Soil associations with tile drainage had a delivery ratio for surface water of 1.00 for sediment; indicating 100% of the sediment estimated to reach the tile drain reached the outlet of the watershed. Undrained soil types had surface water delivery ratios for sediment as follows: Wells Creek close to waterway was 0.20 and distant from waterway was 0.10, Chippewa River close to waterway was 0.10 and distant from waterway was 0.05.

Nutrient files were created from information gathered from the producer surveys. These files had information about all the field operations performed for that farming system by the farmer. Data about nutrient application rates and methods (including manure, where applicable), planting, all tillage operations, and harvesting were furnished for all crops by the producer and included in the input files.

For beef cattle or dairy cattle, either traditional pasture or intensive grazing systems, the producers provided information about the stocking or grazing rates for the animals. The information was used to create nutrient input files in which the manure production for a ten-day period was surface applied (as if it were being applied with a manure spreader) every ten days for the traditional pasture systems. For the intensive grazing systems, the manure produced by the animals while they were in the paddock (for 12 hours to several days) was surface applied (as if by a manure spreader).

For systems currently using conservation tillage, no change in tillage was simulated for the BMP scenario. However, for a system under conventional tillage, a change to conservation tillage was simulated. This typically entailed switching to a less aggressive tillage system. One or two fewer trips across the field occurred, or a less aggressive tillage implement was substituted for the most aggressive implement. For most systems, the same complement of equipment was used. All systems currently being practiced by producers were changed for the BMP scenario to reflect a reduction in phosphorus and nitrogen application rate. These changes reflected appropriate nutrient credits for the previous crops or manure applications (according to University of Minnesota Extension Service recommendations).

Buffer strips, wetlands, and government acreage set aside programs like the Conservation Reserve Program (CRP) are modeled in ADAPT as grassland with no animals or forested land (whichever was appropriate for the program). This approach is likely to create conservative (lower bound) estimates of the potential benefits (reductions in erosion and nutrient loss) of these types of conservation

8

practices and the scenarios that contained them. Model estimates of sediment and nutrient losses from

pasture systems and Managed Intensive Grazing systems were compared to data on soil loss collected

from field-scale monitoring in the nearby Sand Creek watershed and within the Chippewa River

Watershed on similar soils. Model estimates for these systems were found to be consistent with

comparable systems in the monitored areas (C. Iremonger, Department of Soil, Water, and Climate,

University of Minnesota, personal communication).

We defined base flow for both streams as the flow that was exceeded 90% of the time from daily

stream gauge data (Greg Payne, Minnesota Pollution Control Agency, personal communication). Base

flow for the Chippewa River was 0.5 m^3/sec and 1.12 m^3/sec for Wells Creek. We assumed the

proportion of in-stream sediment concentrations due to stream bank erosion as 20% in Wells Creek

(based on estimates for the Whitewater River watershed, a similar watershed in southeast Minnesota,

Anonymous 1998b) and 40% in the Chippewa (Joe Magner, Minnesota Pollution Control Agency, and

Dave Mulla, Department of Water, Soil, and Climate, University of Minnesota, personal communication).

We chose to keep bank erosion estimates constant for both scenarios to separate the effects of land use

on in-stream sediment concentrations from those due to stream bank stabilization. However, stream

bank erosion likely would decrease with BMPs that included increased riparian buffers and permanent

cover along streams. Therefore the estimated benefits from the BMP scenario are conservative.

Adjustment of Crop Yields by Region and Tillage

When surveyed, each producer provided an estimate of their typical or five-year average crop

yield for corn and soybean on the land they farmed. To reflect potential variation in yield by soil, these

typical yields for each crop were adjusted in the following manner. First, the proportion of land farmed by

a producer in each soil association was established. Second, the productivity of the predominant soils

was determined with crop equivalency ratings (CER) as determined by soil scientists and agronomists in

Minnesota (Anderson, Robert, and Rust 1992). The CER for each predominant soil in each soil

association was adjusted based on proportion of the total land in that soil association the soil constituted.

The area-adjusted CER for each soil were summed to obtain the yield-adjusting CER for the soil

association. Last, for a particular producer, the yields in each soil association were adjusted by the CER

for that soil association and the proportion of all land the producer had in any soil association. In this way the weighted average yield of the producer, as adjusted by the CER for each soil association, equaled the typical yield for crops the producer provided on the survey.

Yields were also adjusted for systems that transitioned to conservation tillage. To estimate how yields might change if producers changed from conventional to conservation tillage practices, the following protocol was used. Crop yields from ADAPT for conventional systems were used as the base. Next, these systems were simulated in ADAPT with conservation tillage practices. Essentially, the SCS runoff curve number was adjusted in an ADAPT input file to reflect a cropping system changing to conservation tillage. To determine the marginal change in output for a particular system, crop yields from a given system under conservation tillage were compared to those of the same system with conventional tillage (all other things equal). Though some systems had corn and soybean yields decline by 2% at most, in other parts of the watershed, there were no changes in crop yields.

Estimating Production Costs and Returns

Costs of production for each system for each producer simulated were calculated. These costs of production assumed one acre of production in the given system. For example, on a corn-soybean rotation one-half an acre was planted to corn and the other one-half was planted to soybeans. To estimate costs of production for each system, information from producer surveys was combined with data from the records of the West Central Farm Business Management Association and the Southeastern Farm Business Management Association.

For each crop in each system, costs for fertilizer, agrichemicals and equipment were calculated from the survey responses. Fertilizer cost for each crop in a given system was the product of input level and input price (averaged over 1995-1998, USDA) summed across all inputs. Agrichemical cost for each system was calculated in the same manner (pounds of active ingredient applied multiplied by the pesticide price) (Gunsolus et al. 1999). Cost of machinery for a crop in a given system was the product of the number of uses of the equipment (per acre) and the total cost of machinery (per acre) as obtained from Lazarus (1999). Machinery costs from Lazarus were not adjusted to reflect potential differences in

hours of annual use. It was assumed the costs estimated by Lazarus corresponded to those typical for a farming operation of similar size and scale in the watershed.

To determine all other costs of production, crop enterprise budgets were used. The data for these budgets were obtained from producers in the counties of the sub-watershed who provided records to the Southeast and West Central Farm Business Management Association for analysis. (Note that these producers were not necessarily the same producers surveyed for this research project). The remaining costs for a particular crop enterprise were calculated as the weighted-average of the owned and rented crop budget based on the proportion of land that farmer rented or owned. Consider a producer who rented one-half of the cropland for corn and soybeans. The remaining production costs (other than chemicals and machinery) for each enterprise were calculated as the sum of one-half the crop production costs on rented land and one-half the crop production costs on owned land.

Production costs were adjusted to reflect changes in the current set of production activities. For example, to calculate production costs for a change in nutrient application rate, a new application rate (such as 15 pounds of phosphorus) was substituted for the original application rate and costs were adjusted accordingly. A similar method was used to adjust production costs to reflect changes in tillage (fewer operations or different equipment) or nitrogen application rates.

Fish Effects

For days when sediment loading was predicted by the ADAPT simulations, we calculated in-stream sediment concentrations (mg/L) based on estimates of daily sediment load, daily runoff, base flow for each stream, and the proportion of in-stream sediment due to stream bank erosion. We used estimated daily suspended sediment concentrations to evaluate the effects of sediment on fish assemblages in each stream, by calculating the total number of days for each year that sediment concentrations would be lethal or sublethal to fish in that stream. For our calculations, we referenced a meta-analysis of fish responses to suspended sediment in streams that quantitatively related the biological response of fishes to suspended sediment concentrations and duration of exposure (Newcombe and Jensen 1996). Negative effects to fish increase as suspended sediment concentrations and exposure time increase, and threshold values of lethal and sublethal effects depend on the fish

11

assemblage. Newcombe and Jensen (1996) characterize sublethal effects as reduction in feeding rates or feeding success, physiological stress such as coughing and increased respiration rate, moderate habitat degradation, and impaired homing. Lethal effects are described as reduced growth rate, delayed hatching, reduced fish density, increased predation, severe habitat degradation, and mortality. We used the thresholds of Newcombe and Jensen (1996) to calculate the total number of days that sediment concentrations and duration of exposure met or exceeded the sublethal or lethal levels for the fish assemblage in each watershed. The fish assemblages in the analysis included juvenile and adult salmonids, which we used to represent Wells Creek, and adult freshwater non-salmonids, mainly composed of warmwater species, which represented the Chippewa River. Because Wells Creek supported a trout population prior to increases in suspended sediment and temperature resulting from land use, the biological response of the salmonid community to sediment concentrations was more appropriate to use in this stream than the response of warmwater species.

Results

Runoff and Sediment Load

Of a total of 18,263 days over the modeling period, sediment was potentially delivered to the stream mouths between 1614 and 2590 days, depending on the watershed and land use scenario. The total number of days with sediment loading was higher in the Chippewa River watershed than for Wells Creek for both scenarios (baseline and BMP), with sediment loading occurring between 2024 and 2590 days in the Chippewa, compared to a range of 1614 to 1729 days in Wells Creek. The average duration of runoff events was longer in the Chippewa than Wells Creek, which resulted in longer average exposure times to suspended sediment for fish in the Chippewa watershed. Water runoff in both watersheds decreased slightly under the BMP scenario (3% reduction from baseline in Wells Creek and 1% reduction from baseline in the Chippewa). However, the mean in-stream sediment concentration was higher in Wells Creek (range 293 mg/L to 1476 mg/L). The Chippewa watershed had less variation in mean sediment concentration among scenarios, with values from 377 mg/L to 585 mg/L.

In Wells Creek watershed, estimated sediment loading decreased by 31% with BMPs from the estimated baseline load of 39,615 tons per year. Nitrogen loss decreased by 37% from baseline load of

3,001 tons annually and phosphorus loss declined by 52% from the estimated baseline load of 7,547 tons annually under the BMP scenario in Wells Creek. In the Chippewa River subwatershed, approximately 2,000 tons of sediment reaches the main stem of the Chippewa River annually. Under the BMP scenario, estimated sediment loading decreased by 25% from the baseline. Compared to the baseline, nitrogen loss declined by 17% (from an estimated 13,966 tons annually) and phosphorus loss decreased by 40% (from an estimated 5,108 tons annually) under the BMP scenario.

Sediment Effects on Fish

Mean sediment concentrations for both streams and both scenarios were above the threshold for sublethal effects to fish, but were not lethal for an exposure of one day or less. However, mean sediment concentrations could have been classified as lethal for greater than one day of exposure. The mean annual number of days with sublethal or lethal sediment concentrations to fish was slightly higher in the Chippewa River than Wells Creek. The mean number of days per year with lethal sediment concentrations to fish ranged from 10.2 to 11.6 in the Chippewa, depending on the scenario, compared to 0.2 to 7.6 days in Wells Creek. Mean sublethal events in the Chippewa ranged from 31.1 to 40.8 days per year, compared to 25.8 to 32.4 in Wells Creek. These values were not significantly different in the Chippewa River ($p>0.99$ for mean annual days with lethal sediment concentrations, $p>0.060$ for sublethal), but represented a slight decrease in lethal and sublethal events with BMPs.

There were significant differences between mean values among scenarios for lethal events ($p<0.0001$) in Wells Creek. The baseline had significantly higher mean annual days with lethal sediment concentrations than the BMP scenario. Under the BMP scenario, lethal fish effects were estimated to decline by almost 60%. However, counterintuitively, there was an increase from the baseline (although not statistically significant) in the number of sublethal events in the Wells Creek with the BMP scenario.

Land Use and Net Farm Income

Under the BMP scenario, 1,366 acres of cropland were converted to 100 foot grass buffer strips along all permanent waterbodies in Wells Creek (Figure 3). All cropland remaining in production in Wells Creek used conservation tillage and nutrient management practices. For the watershed, total revenue

decreased by $435,461 while estimated total costs declined by $417,574 (Table 3). As a result, net farm income declined by less than $18,000 (1%) annually from the baseline with BMPs.

In the Chippewa River subwatershed, which has fewer waterbodies, 879 acres of grass buffer strips were created from land currently in row-crops (Figure 4). Under the BMP scenario, total revenue decreased by approximately $301,995, but estimated total costs declined by $274,405 (Table 4). As a result, net farm income declined by less than $28,000 (3%) annually with BMPs being implemented throughout the Chippewa River subwatershed.

Discussion

Effects of land use practices on fish assemblages, as well as on patterns of sediment and runoff, were dependent on physical attributes of each watershed. The mean number of days with lethal and sublethal sediment concentrations was higher in the Chippewa River than in Wells Creek, which is likely a function of the combined influences of differences among fish assemblages, land use practices, topography, and soils between the two watersheds. In general, the fish assemblage in the Chippewa watershed was more sensitive to sediment concentrations than the coolwater assemblage in Wells Creek for exposure longer than one day. Sediment concentrations were often lower in the Chippewa River watershed than those in Wells Creek, but were delivered for a longer duration.

The concentration of suspended sediment in a stream on any given day is a product of several factors, including land use practice, soil type, vegetative cover, topography, precipitation, and time of year (Wood and Armitage 1997). Whereas physical differences such as soil type, topography and precipitation amounts and timing likely contributed to differences in suspended sediment concentrations between watersheds, we varied land use in our analysis to examine how implementing BMPs would affect suspended sediment concentrations and fish assemblages in watersheds with different physical attributes. We determined that although land using BMPs has an important role in controlling the amount of sediment that reaches a stream, interactions between land uses, biological attributes of the fish assemblage in a stream, and physical properties of a watershed ultimately determine the degree of change within a watershed needed to have a measurable effect. Thus, environmental benefits society

hopes will result from farmers using BMPs may be mitigated by physical and physiological characteristics and processes of the watershed.

Both sediment loading and runoff can be influenced by agricultural activities, as well as by physical attributes of a river's drainage watershed (Wood and Armitage 1997). Sediment loading is high in cultivated watersheds (Allan et al. 1997) and the amount of row crop agriculture in a watershed is correlated with stream habitat quality and biotic integrity (Wang et al. 1997). A decrease in the amount of cultivated land can decrease sediment loading by reducing the availability of easily erodible sediment and the volume of runoff that can transport soil particles (Kuhnle et al. 1996). BMPs that leave crop residue on the soil surface can significantly reduce soil loss relative to conventional cropping systems, although surface runoff may not significantly decrease (Daryoush 1990, Ghidey and Alberts 1998).

Increased amounts of riparian forests and grasslands in a watershed generally provide a decrease in runoff and sediment loading, with the vegetation stabilizing streambanks, decreasing water temperatures by shading the stream, and intercepting surface runoff and subsurface flow (Timmons and Holt 1977, Lowrance et al. 1984, Peterjohn and Correll 1984, Barton et al. 1985, Delong and Brusven 1991, Richards and Host 1993, Tufford et al. 1998, Castelle and Johnson 2000, Stauffer et al. 2000).

The role of land use and ground cover on stream sediment concentrations was illustrated by the comparison of mean suspended sediment concentration and days with lethal effects to fish in Wells Creek. Baseline conditions in Wells Creek resulted in a relatively high number of days with lethal sediment concentrations to fish. Adoption of BMPs led to a measurable decrease in lethal events for trout. However, decreases were less evident in the Chippewa River, likely due to other factors controlling runoff patterns, differences in the tolerance of warmwater fish species to suspended sediment compared to salmonids, and the interaction of these factors with land use effects.

Differences in the amount of cultivated land and natural vegetation in each watershed may interact with watershed physical properties, and influence patterns of runoff and sediment flow. Agriculture is currently the leading source of water pollution in the United States, and the most common agricultural pollutant is sediment (USEPA 1996). Both watersheds in our study primarily are cultivated and can have soil erosion five to ten times greater than land with natural vegetative cover (Delong and Brusven 1991). However, the Chippewa watershed has 20% more land area in row crops than Wells

Creek, with 81% of the watershed in cultivation. Wang et al. (1997) found that a decrease in habitat quality and index of biotic integrity (IBI) scores for fish assemblages were only measurable in streams draining watersheds with greater than 50% agricultural land use. The greater proportion of land in the Chippewa watershed currently under cultivation may have a disproportionately greater effect on sediment loading to the Chippewa River than sediment loading in Wells Creek. Conversely, the greater proportion of land in the Wells Creek watershed that is in forest or grassland may act as a sediment sink on days with relatively low runoff, thereby decreasing the total number of days with sediment runoff to the stream, and decreasing the periods of consecutive days with sediment loading.

A major influence of agricultural land use on stream ecosystems is the decrease in riparian buffers (Schlosser 1991). Riparian buffers are a major source of nutrients and organic matter for rivers, and are greatly decreased by flood control structures, removal of woody debris, diversion of side streams, and clearing riparian vegetation (Gregory et al. 1991, Schlosser 1991, Ward and Stanford 1995). Riparian vegetation intercepts and controls erosion and improves stream water quality through nutrient uptake and removal before runoff reaches the stream (Lowrance et al. 1984). Although both watersheds are primarily cultivated, the Chippewa River has more channelized reaches (Joe Magner, Minnesota Pollution Control Agency, personal communication), and riparian grasslands and forests are less abundant than along Wells Creek. This change in structure in the Chippewa River has likely compounded the effects of suspended sediment concentrations by reducing in-stream habitat heterogeneity, thereby decreasing the diversity of the fish assemblage and complexity of the size structure of fish populations (Berkman and Rabeni 1987, Schlosser 1991). In addition, land adjacent to a stream that is in conventional cultivation is usually much more erosive than vegetated stream banks (Wilkin and Hebel 1982). Therefore, the relative lack of riparian vegetation along the Chippewa River under current conditions likely contributes to the bulk of the estimated 40% contribution of stream bank erosion to in-stream suspended sediment. Although we held bank erosion constant across all scenarios for our simulations, an increase in riparian buffers in the BMP scenario likely would decrease bank erosion. Thus, our estimates of sediment concentrations were likely too high.

The topography of the Chippewa affected runoff and sediment delivery to the stream. Because the Chippewa is relatively flat compared to Wells Creek, with most land in row crops having slopes not

exceeding 2%, the proportion of sediment delivered to the stream from upland areas in the watershed was not as great as sediment delivery in Wells Creek. Therefore, changes in land use closer to the river will likely have a disproportionate effect on in-stream suspended sediment (Wilkin and Hebel 1982).

In Wells Creek watershed, the steeply sloped, well drained soils allowed runoff to rapidly reach the stream, resulting in a pattern of peaks of high sediment concentrations that quickly subside. In contrast, the relatively flat, moderately to very poorly drained soils of the Chippewa watershed take more time to drain. Paradoxically, this leads to a greater number of days and more protracted periods with runoff, although sediment concentrations are generally lower. Because many of the soils in the Chippewa watershed have little slope and are moderately to very poorly drained, tile drainage systems are becoming more common on agriculturally managed lands. Thus, the relatively flat topography influences runoff patterns in the watershed, which may explain why the Chippewa River watershed with less annual rainfall than Wells Creek had more consecutive days with measured runoff.

The results indicated that beneficial effects on fish assemblage can be achieved with relatively minimal adverse effect on agricultural production and net farm income. In Wells Creek, slightly more than 5% of row-crop acreage was converted to grass riparian buffers under the BMP scenario. Changing land management practices reduced net farm income by less than 1% ($18,000) per year from baseline levels. Another way to look at this is that the $18,000 annual compensation producers in Wells Creek would need to receive to be as well off under the BMP scenario for retiring 1,366 acres of cropland to benefit fish assemblage (among others) in Wells Creek is about $13 per acre.

In the Chippewa River subwatershed, the story is similar, though slightly more costly. A little more than 2% of cultivated cropland (879 acres) in the baseline needed to be removed from production and planted to grass buffer strips under BMP scenario. This reduced net farm income in this subwatershed by 3% ($27,590) annually from baseline levels. If producers needed to receive $27,590 annually for lost income under this scenario, society could compensate farmers $31 for each acre of land retired from production. This would allow farmers to be as well off under the BMP scenario as they are with current production practices. Relative to the current payments producers receive under the Conservation Reserve Program, compensation at this level would be a great bargain for society.

Acknowledgments

Table 1. Stream fish population assessment in 1999 for Wells Creek, Goodhue County (Anonymous 1999). Tolerance and temperature preference are from Lyons (1992).

Species	Common Name	Abundance	Tolerance	Temperature Preference
Catostomus commersoni	white sucker	common	tolerant	Eurythermal
Lota lota	burbot	present	intermediate	stenothermal
Lepomis macrochirus	bluegill	present	intermediate	Eurythermal
Lepomis cyanellus	green sunfish	present	tolerant	eurythermal
Rhinichthys atratulus	blacknose dace	occasional	tolerant	eurythermal
Etheostoma flabellare	fantail darter	occasional	intermediate	eurythermal
Semotilus atromaculatus	creek chub	occasional	tolerant	eurythermal
Luxilus cornutus	common shiner	occasional	intermediate	eurythermal
Salmo trutta	brown trout	low	intolerant	stenothermal

Table 2. Stream fish population assessment in 1998 for the Chippewa River, Chippewa and Swift counties (Anonymous 1998a). Tolerance is from Lyons (1992).

Species	Common name	Tolerance	% of total
Moxostoma anisurum	silver redhorse	intermediate	23.5
Cyprinus carpio	common carp	tolerant	15.4
Luxilus cornutus	common shiner	intermediate	8.7
Notropis heterodon	blackchin shiner	intolerant	7.4
Moxostoma macrolepidotum	shorthead redhorse	intermediate	6.0
Stizostedion vitreum	walleye	intermediate	6.0
Notropis hudsonius	spottail shiner	intolerant	5.4
Ictiobus bubalus	smallmouth buffalo	intermediate	4.7
Ictalurus punctatus	channel catfish	intermediate	4.7
Esox lucius	northern pike	intermediate	4.0
Notropis dorsalis	bigmouth shiner	intermediate	3.4
Lepomis humilis	orangespotted sunfish	intermediate	2.0
Percina shumardi	river darter	intermediate	2.0
Pimephales promelas	fathead minnow	tolerant	1.3
Semotilus atromaculatus	cheek chub	tolerant	1.3
Carpoides cyprinus	quillback	intermediate	1.3
Catostomus commersoni	white sucker	tolerant	1.3
Notropis stramineus	sand shiner	intermediate	0.7
Ictalurus melas	black bullhead	intermediate	0.7

Table 3. Estimated total revenue, total costs, and net farm income for Baseline and Best Management Practices (BMP) Scenarios in Wells Creek.

Wells Creek Economic Results								
	Baseline			Best Management Practices			Baseline - BMPs		
	Total Revenue	Total Cost	Net Income	Total Revenue	Total Cost	Net Income	Total Revenue	Total Cost	Net Income
Crops	7,002,559	6,670,542	332,017	6,567,098	6,252,968	314,130	435,461	417,574	17,887
Grain-Alfalfa Hay Conservation Tillage	842,551	716,844	125,707	1,298,230	1,104,537	193,693	(455,680)	(387,693)	(67,986)
Grain-Alfalfa Hay Conventional Tillage	541,151	463,987	77,165	-	-	-	541,151	463,987	77,165
Corn-Corn Conservation Tillage	960,203	942,802	17,400	1,164,652	1,143,547	21,105	(204,450)	(200,745)	(3,705)
Corn-Corn Conventional Tillage	278,982	274,680	4,302	-	-	-	278,982	274,680	4,302
Corn-Soybean Conservation Tillage	2,026,564	1,977,516	49,048	4,104,215	4,004,883	99,332	(2,077,651)	(2,027,367)	(50,284)
Corn-Soybean Conventional Tillage	2,353,109	2,294,713	58,395	-	-	-	2,353,109	2,294,713	58,395
Livestock	9,253,000	7,421,461	1,831,539	9,253,000	7,421,461	1,831,539	-	-	-
Pasture - dairy	7,549,328	6,031,237	1,518,092	7,549,328	6,031,237	1,518,092	-	-	-
Intensive Grazing - dairy	411,240	328,544	82,696	411,240	328,544	82,696	-	-	-
Pasture - beef	538,069	436,744	101,325	538,069	436,744	101,325	-	-	-
Intensive Grazing - beef	67,408	54,714	12,694	67,408	54,714	12,694	-	-	-
Hogs	686,954	570,222	116,733	686,954	570,222	116,733	-	-	-
Total	16,255,559	14,092,003	2,163,556	15,820,098	13,674,428	2,145,669	435,461	417,574	17,887

Table 4. Estimated total revenue, total costs, and net farm income for Baseline and Best Management Practices (BMP) Scenarios in Chippewa.

Chippewa Economic Results								
	Baseline			Best Management Practices			Baseline - BMPs		
	Total Revenue	Total Cost	Net Income	Total Revenue	Total Cost	Net Income	Total Revenue	Total Cost	Net Income
Crops	9,573,506	8,760,672	812,835	9,271,511	8,486,267	785,245	301,995	274,405	27,590
Grain-Alfalfa Hay Conservation Tillage	144,081	110,710	33,370	330,961	254,308	76,654	(186,881)	(143,597)	(43,283)
Grain-Alfalfa Hay Conventional Tillage	197,604	152,888	44,716	-	-	-	197,604	152,888	44,716
Corn-Soybean Conservation Tillage	2,392,697	2,201,448	191,249	6,984,517	6,426,242	558,275	(4,591,820)	(4,224,794)	(367,026)
Corn-Soybean Conventional Tillage	4,829,816	4,440,727	389,088	-	-	-	4,829,816	4,440,727	389,088
Corn-Sugar Beet Conventional Tillage	2,009,309	1,854,899	154,411	1,956,033	1,805,717	150,316	53,276	49,182	4,094
Livestock	858,201	752,650	105,551	858,201	752,650	105,551	-	-	-
Pasture - dairy	275,141	211,810	63,331	275,141	211,810	63,331	-	-	-
Intensive Grazing - dairy	14,988	11,538	3,450	14,988	11,538	3,450	-	-	-
Pasture - beef	193,696	193,696	-	193,696	193,696	-	-	-	-
Intensive Grazing - beef	23,900	23,900	-	23,900	23,900	-	-	-	-
Hogs	350,477	311,707	38,770	350,477	311,707	38,770	-	-	-
Total	10,431,707	9,513,322	918,385	10,129,712	9,238,917	890,796	301,995	274,405	27,590

Figure 1. Map of Wells Creek watershed under baseline conditions.

WELLS CREEK WATERSHED
BASELINE: CURRENT LAND USE AND COVER

LAND USE AND COVER

Acres	Description
24,533	Cultivated Land (61%)
	1,047 in-program acres
	23,486 non-program acres
920	Development (2%)
4,090	Grassland (10%)
	163 in-program acres
	3,927 non-program acres
10,430	Grassland-Shrub-Tree, Forest (26%)
69	Sand, Gravel or Unclassified (0%)
78	Water (0%)
52	Wetlands (0%)
	5 in-program acres
	47 non-program acres

EXPLANATION

The land use map displays the current land use and cover in the watershed based on 1990 data. The Land Use and Cover table reports the acres and percent of each classification including conservation program information. Cultivated land, grassland and wetland acres enrolled in a conservation program are designated as in-program. The remaining acres are designated as non-program. Total watershed area is 40,172 acres.

Scale = 1:130000

Distribution of Crops with Tillage Practices on Cultivated Land

Conservation Tillage (CT)
Conventional Tillage (CN)

Acres	Crop and Tillage
3,241	Small Grain/Alfalfa Hay CT
2,061	Small Grain/Alfalfa Hay CN
2,745	Corn-Corn CT
790	Corn-Corn CN
6,812	Corn-Soybean CT
7,836	Corn-Soybean CN

Prepared for: The Land Stewardship Project, Multiple Benefits of Agriculture
Prepared by: Cis Berg, Water Resources Center
Minnesota State University, Mankato
Date: June, 2001
Software: ARC/INFO 7.1.2 and ArcView 3.1
Data Sources: Land Use and Cover data are from the International Coalition Land Use/ Land Cover (1990). Conservation program data were developed from the GAP Stewardship and BWSR Easement Parcel GIS data. Scenario data were developed by the Multiple Benefits of Agriculture Project Team.

Figure 2. Map of the Chippewa River subwatershed under baseline conditions.

CHIPPEWA RIVER STUDY AREA
BASELINE: CURRENT LAND USE AND COVER

LAND USE AND COVER

Acres		Description
35,938		Cultivated Land (81%)
		2,794 in-program acres
		33,144 non-program acres
1,573		Development (3%)
3,615		Grassland (8%)
		487 in-program acres
		3,128 non-program acres
2,667		Grassland-Shrub-Tree, Forest (6%)
5		Sand, Gravel or Unclassified (0%)
266		Water (1%)
381		Wetlands (1%)
		86 in-program acres
		295 non-program acres

EXPLANATION

The land use map displays the current land use and cover in the watershed based on 1990 data. The Land Use and Cover table reports the acres and percent of each classification including conservation program information. Cultivated land, grassland and wetland acres enrolled in a conservation program are designated as in-program. The remaining acres are designated as non-program. Total watershed area is 44,445 acres.

Distribution of Crops with Tillage Practices on Cultivated Land

Conservation Tillage (CT)
Conventional Tillage (CN)

Acres		Crop and Tillage
566		Small Grain/Alfalfa Hay CT
770		Small Grain/Alfalfa Hay CN
9,377		Corn-Soybean CT
18,741		Corn-Soybean CN
3,689		Corn-Sugar Beets CN

Prepared for: The Land Stewardship Project, Multiple Benefits of Agriculture
Prepared by: Cis Berg, Water Resources Center
    Minnesota State University, Mankato
Date: June, 2001
Software: ARC/INFO 7.1.2 and ArcView 3.1
Data Sources: Land Use and Cover data are from the International Coalition Land Use/ Land Cover (1990). Conservation program data were developed from the GAP Stewardship and BWSR Easement Parcel GIS data. Scenario data were developed by the Multiple Benefits of Agriculture Project Team.

Scale = 1:175000

1   0   1   2 Miles

Figure 3. Map of the Wells Creek watershed under the BMP Scenario.

WELLS CREEK WATERSHED
SCENARIO B: BEST MANAGEMENT PRACTICES

LAND USE AND COVER

Acres	Description
24,533	Cultivated Land (61%)
	2,413 in-program acres
	22,120 non-program acres
920	Development (2%)
4,090	Grassland (10%)
	163 in-program acres
	3,927 non-program acres
10,430	Grassland-Shrub-Tree, Forest (26%)
69	Sand, Gravel, or Unclassified (0%)
78	Water (0%)
52	Wetlands (0%)
	5 in-program acres
	47 non-program acres

Conversion of Cultivated Land to:

100 Foot Grassed Stream Buffer
1,366 acres

EXPLANATION

The land use map displays the current land use and cover in the watershed based on 1990 data modified by Scenario B parameters. The Land Use and Cover table reports the acres and percent of each classification including conservation program information. Cultivated land, grassland and wetland acres enrolled in a conservation program are designated as in-program. The remaining acres are designated as non-program. The cultivated land in-program acres are increased by implementing a 100 foot grassed stream buffer on cultivated land. Total watershed area is 40,172 acres

Scale = 1:130000

1   0   1   2 Miles

Distribution of Crops with Tillage Practices on Cultivated Land

Conservation Tillage (CT)
Conventional Tillage (CN)

Acres	Crop and Tillage
4,994	Small Grain/Alfalfa Hay CT
3,330	Corn-Corn CT
13,796	Corn-Soybean CT

Prepared for: The Land Stewardship Project, Multiple Benefits of Agriculture
Prepared by: Cis Berg, Water Resources Center
Minnesota State University, Mankato
Date: June, 2001
Software: ARC/INFO 7.1.2 and ArcView 3.1
Data Sources: Land Use and Cover data are from the International Coalition
Land Use/ Land Cover (1990). Conservation program data were developed
from the GAP Stewardship and BWSR Easement Parcel GIS data. Scenario
data were developed by the Multiple Benefits of Agriculture Project Team.

Figure 4. Map of the Chippewa River subwatershed under the BMP Scenario.
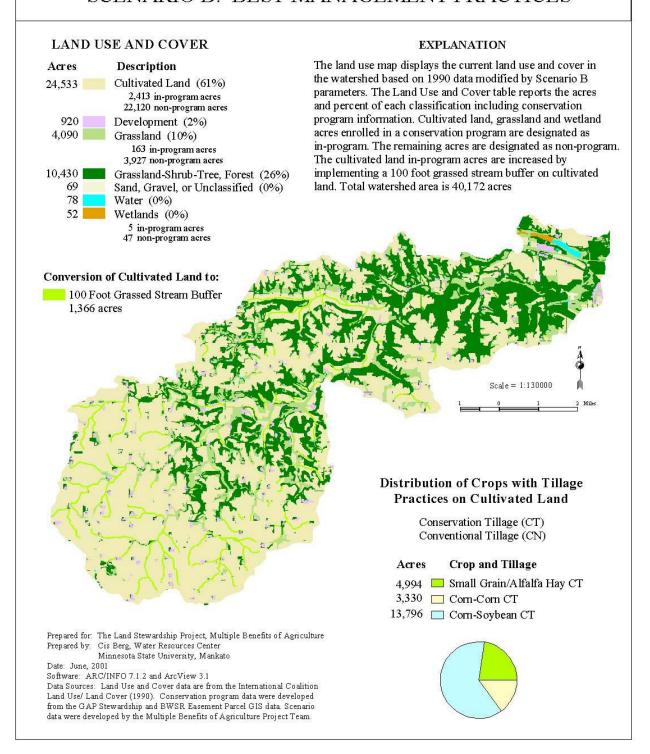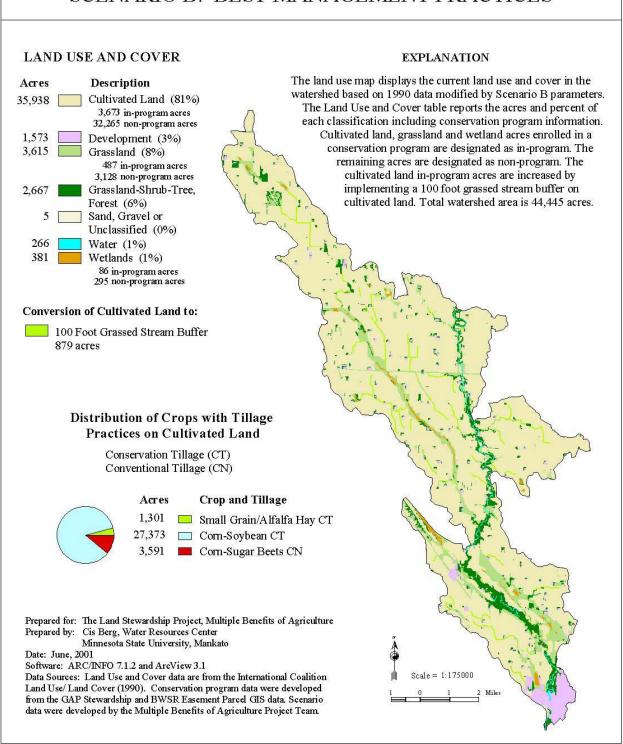

CHIPPEWA RIVER STUDY AREA
SCENARIO B: BEST MANAGEMENT PRACTICES

LAND USE AND COVER

Acres		Description
35,938		Cultivated Land (81%)
		3,673 in-program acres
		32,265 non-program acres
1,573		Development (3%)
3,615		Grassland (8%)
		487 in-program acres
		3,128 non-program acres
2,667		Grassland-Shrub-Tree, Forest (6%)
5		Sand, Gravel or Unclassified (0%)
266		Water (1%)
381		Wetlands (1%)
		86 in-program acres
		295 non-program acres

Conversion of Cultivated Land to:

100 Foot Grassed Stream Buffer
879 acres

Distribution of Crops with Tillage Practices on Cultivated Land

Conservation Tillage (CT)
Conventional Tillage (CN)

Acres		Crop and Tillage
1,301		Small Grain/Alfalfa Hay CT
27,373		Corn-Soybean CT
3,591		Corn-Sugar Beets CN

EXPLANATION

The land use map displays the current land use and cover in the watershed based on 1990 data modified by Scenario B parameters. The Land Use and Cover table reports the acres and percent of each classification including conservation program information. Cultivated land, grassland and wetland acres enrolled in a conservation program are designated as in-program. The remaining acres are designated as non-program. The cultivated land in-program acres are increased by implementing a 100 foot grassed stream buffer on cultivated land. Total watershed area is 44,445 acres.

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Scale = 1:175000

1   0   1   2 Miles

References

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

Anderson, J.L., P.C. Robert, and R.H. Rust. *Productivity Factors and Crop Equivalent Ratings for Soils of Minnesota.* Saint Paul MN: University of Minnesota Extension Service, AG-BU-2199-F, 1992.

Anonymous. 1998a. Chippewa River Population Assessment. Minnesota Department of Natural Resources, Section of Fisheries. Spicer, MN.

Anonymous. 1998b. Watershed plan - Environmental Assessment, Whitewater River watershed, MN. Natural Resources Conservation Service, St. Paul, MN.

Anonymous. 1999. Stream population assessment for Wells Creek, Goodhue County. Minnesota Department of Natural Resources, Section of Fisheries, Lake City, MN.

Barton, D. R., W. D. Taylor, and R. M. Biette. 1985. Dimensions of riparian buffer strips required to maintain trout habitat in southern Ontario streams. North American Journal of Fisheries Management 5: 364-378.

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

Blann, K., J. Frost, and B. Vondracek. In review. Relationship of riparian buffer type to water temperature in the Driftless Area of Minnesota. North American Journal of Fisheries Management.

Castelle, A. J. and A. W. Johnson. 2000. Riparian vegetation effectiveness. Technical Bulletin No. 799, National Council for Air and Stream Improvement, Inc. Research Park Triangle, NC.

Chung, S. O., A. D. Ward and C. W. Schalk. 1992. Evaluation of the hydrologic component of the ADAPT water table management model. Transaction of the American Society of Agricultural Engineers 35:571-579.

Conservation Tillage Information Center. 1999. Conservation Tillage Survey Data. July 1999 <http://www.ctic.purdue.edu/Core4/CT/CT.html/>.

Davis, D. M., P. H. Gowda, D. J. Mulla, and G. W. Randall. 2000. Modeling nitrate nitrogen leaching in response to nitrogen fertilizer rate and tile drain depth or spacing for southern Minnesota, USA. Journal of Environmental Quality 29:1568-1581.

Dalzell, B.J. 2000. Modeling and evaluation of non-point source pollution in the lower Minnesota river

watershed. University of Minnesota. Unpublished thesis.

Delong, M. D. and M. A. Brusven. 1991. Classification and spatial mapping of riparian habitat with
applications toward management of streams impacted by nonpoint source pollution. Environmental
Management 15: 565-571.

Historical Data Retrieval and Climate Summaries. Department of Soil, Water and Climate, University of
Minnesota. April 2000 <http://climate.umn.edu/doc/historical.htm>.

Ghidey, F. and E. E. Alberts. 1998. Runoff and soil losses as affected by corn and soybean tillage systems.
Journal of Soil and Water Conservation 53: 64-70.

Gowda, P. H., A. D. Ward, D. A. White, D. B. Baker, and J. G. Lyon. 1999a. Using field scale models to
predict peak flows on agricultural watersheds. Journal of the American Water Resources Association 35:
1223-1232.

Gowda, P. A. Ward, D. White, J. Lyon and E. Desmond. 1999b. The sensitivity of ADAPT model predictions
of streamflows to parameters used to define hydrologic response units. Transaction of the American
Society of Agricultural Engineers 42:381-389.

Gunsolus, J.L., R.L. Becker, B.R. Durgan, W. Lueschen and A.G. Dexter. *1999 Cultural and Chemical Weed
Control in Field Crops.* Saint Paul MN: University of Minnesota Extension Service BU-3157-S, 1999.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones , III. 1998. Stream biodiversity:
The ghost of land use past. Proceedings of the National Academy of Sciences of the United States of
America 95: 14843-14847.

Johansson, R.C. 2000. Point- nonpoint emissions trading for Minnesota river phosphorus. University of
Minnesota. Unpublished thesis.

Kuhnle, R. A., R. L. Bingner, G. R. Foster, and E. H. Grissinger. 1996. Effect of land use changes on
sediment transport in Goodwin Creek. Water Resources Research 32: 3189- 3196.

Minnesota Land Use and Cover: 1990's Census of the Land. Minnesota Planning, Land Management
Information Center. April 2000 <http://mapserver.lmic.state.mn.us/landuse>.

Larson, W. E., M. J. Lindstrom, and T. E. Schumacher. 1997. The role of severe storms in soil erosion: A
problem needing consideration. Journal of Soil and Water Conservation 52: 90-95.

Lazarus, W.F. *Farm Machinery Economic Costs for 1999: Minnesota Estimates with Adjustments for Use in*

Canada. Saint Paul MN: Department of Applied Economics, University of Minnesota, Staff Paper P99-5, May 1999.

Leonard, R. A., W. G. Knisel and D. A. Still. 1987. GLEAMS: Groundwater loading effects of agricultural management systems. Transactions of the American Society of Agricultural Engineers 40:945-952.

Lowrance, R., R. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34: 374-377.

Lyons, J. 1992. Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. General Technical Report NC-149, U.S. Forest Service, North Central Forest Experiment Division, St. Paul, MN, 51 pp.

Lyons, J., L. Wang, and T. D. Simonson. 1996. Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. North American Journal of Fisheries Management 16: 241-256.

Newcombe, C. P. and J. O. Jensen. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16: 693-727.

Olness, A., S. J. Smith, E. D. Rhoades, and R. G. Menzel. 1975. Nutrient and sediment discharge from agricultural watersheds in Oklahoma. Journal of Environmental Quality 4: 331-336.

Omernik, J. M, and A. L. Gallant. 1988. Ecoregions of the upper Midwest states. EPA/600/3- 88-037, U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Paragamian, V. L. 1989. Seasonal habitat use by walleye in a warmwater river system, as determined by radiotelemetry. North American Journal of Fisheries Management 9: 392- 401.

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

Poff, N. L. and J. D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76: 606-627.

Richards, C. and G. E. Host. 1993. Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment. Freshwater Biology 29: 285-294.

Saunders, J. W. and M. W. Smith. 1965. Changes in a stream population of trout associated with increased silt. Journal of the Fisheries Research Board of Canada 22: 395-404.

Schleiger, S. L. 2000. Use of an index of biotic integrity to detect effects of land use on stream fish

    communities in west-central Georgia. Transactions of the American Fisheries Society 129: 1118-1133.

Schlosser, I. J. 1991. Stream fish ecology: A landscape perspective. Bioscience 41: 704-712.

Skaggs, R. W. 1978. A water table management model for shallow water table soils. Report No. 134, Water

    Resources Research Institute, University of North Carolina, Raleigh, North Carolina.

Stauffer, J. C., R. M. Goldstein, and R. M. Newman. 2000. Relationship of wooded riparian zones and runoff

    potential to fish community composition in agricultural streams. Canadian Journal of Fisheries and

    Aquatic Sciences 57: 307-316.

Timmons, D. R. and R. F. Holt. 1977. Nutrient losses in surface runoff from a native prairie. Journal of

    Environmental Quality 6: 369-373.

Tufford, D. L., H. N. McKellar, and J. R. Hussey. 1998. In-stream nonpoint source nutrient prediction with

    land-use proximity and seasonality. Journal of Environmental Quality 27: 100-111.

U.S. Environmental Protection Agency. 1996. National Water Quality Inventory of 1996. Washington, D.C.

U.S. Department of Agriculture. 1999. 1997 Census of Agriculture, Volume 1, Geographic Area Series, Part

    23, Minnesota State and County Data. Washington, D.C.

Wang, L., J. Lyons, P. Kanehl, and R. Gatti. 1997. Influences of watershed land use on habitat quality and

    biotic integrity in Wisconsin streams. Fisheries 22(6): 6-12.

Ward, J. V. and J. A. Stanford. 1995. Ecological connectivity in alluvial river ecosystems and its disruption by

    flow regulation. Regulated Rivers: Research and Management 11: 105-119.

Waters, T. F. 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society

    Monograph 7, 251pp.

Wehrly, K. E., M. J. Wiley, and P. W. Seelbach. 1999. A thermal habitat classification for lower Michigan

    Rivers.2038: 1-51.

Westra, J. V., K. W. Easter, and K. D. Olson. 2002. Targeting nonpoint source pollution control: phosphorus

    in the Minnesota River watershed. Journal of the American Water Resources Association 38(2): 493-505.

Wilkin, D. C. and S. J. Hebel. 1982. Erosion, redeposition, and delivery of sediment to Midwestern streams.

    Water Resources Research 18: 1278-1282.

Wohl, N. E. and R. F. Carline. 1996. Relations among riparian grazing, sediment loads, macroinvertebrates,

and fishes in three central Pennsylvania streams. Canadian Journal of Fisheries and Aquatic Sciences

53: 260-266.

Wood, P. J. and P. D. Armitage. 1997. Biological effects of fine sediment in the lotic environment.

Environmental Management 21: 203-217.