WALNUT CREEK WATERSHED RESTORATION AND WATER QUALITY MONITORING PROJECT: FINAL REPORT

Iowa Geological Survey
Technical Information Series No. 49

Iowa Department of Natural Resources
Jeffrey R. Vonk, Director
April 2006

COVER

The Walnut Creek watershed is being restored to native tallgrass prairie from a landscape once dominated by row-crop agriculture. Currently, more than 3,000 acres of prairie have been planted by the U.S. Fish and Wildlife Service at the Neal Smith National Wildlife Refuge in Jasper County.

Photo courtesy of Neal Smith National Wildlife Refuge.

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Iowa Geological Survey
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Prepared by

K.E. Schilling[1], T. Hubbard[2], J. Luzier[2], J. Spooner[3]

[1]Iowa Department of Natural Resources, Geological Survey
109 Trowbridge Hall, Iowa City, IA 52242-1319

[2]University of Iowa Hygienic Laboratory
University of Iowa; Oakdale Campus, Iowa City, IA 52242

[3]NCSU Water Quality Group, North Carolina State University,
Raleigh, NC 27695-7637

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EXECUTIVE SUMMARY

The Walnut Creek Watershed Restoration and Water-Quality Monitoring Project was established in 1995 as a NPS monitoring program in conjunction with watershed habitat restoration and agricultural management changes implemented by the U.S. Fish and Wildlife Service (USFWS) at the Neal Smith National Wildlife Refuge and Prairie Learning Center (Refuge) in Jasper County Iowa. A large portion of the Walnut Creek watershed is being restored from row crop agriculture to native prairie and savanna. The objectives of the project were to: 1) perform comprehensive, long-term NPS monitoring in the Walnut and Squaw Creek watersheds; 2) quantitatively document over time reduction in NPS pollution and associated environmental improvements resulting from watershed habitat restoration and land management changes; and 3) use the monitoring data to increase our understanding of what implementation measures are successful and will be helpful in similar areas, and expand public awareness of the need for NPS pollution prevention measures in the State of Iowa.

The Walnut Creek Monitoring project utilized a paired-watershed as well as upstream/downstream comparisons for analysis and tracking of trends. The Walnut Creek watershed is paired with the Squaw Creek watershed and a common basin divide is shared. Based on their similar basin characteristics, the watersheds are well suited to such a design. In addition, several subbasins are monitored in both watersheds to allow comparisons of differential implementation over time, and for analyzing their incremental contributions to the overall basin response. Four basic components comprised the project: 1) tracking of land cover and land management changes within the basins, 2) stream gaging for discharge and suspended sediment at two locations on Walnut Creek and one on Squaw Creek, 3) surface water quality monitoring in the Walnut and Squaw Creek watersheds, and 4) biomonitoring for aquatic macroinvertebrates and fish in Walnut and Squaw Creeks.

In 1990, land use in both Walnut and Squaw Creek watersheds was dominated by row crops of corn and soybeans, with 69.4 percent row crop in Walnut Creek and 71.4 percent in Squaw Creek. From 1990 to 2005, major changes in land cover occurred in both watersheds. Squaw Creek showed an increasing trend of row crop land use whereas row crop in Walnut Creek significantly decreased. In Squaw Creek, the 9.2 percent increase in row crop area from 1990 to 2005 was likely due to the passage of the Freedom to Farm Act in 1996 that appeared to have substantially increased row crop production. Lands previously categorized as grasslands enrolled in the Conservation Reserve Program (CRP) were converted to row crop production. This trend was particularly evident in subbasins SQW4 and SQW5 where the row crop percentage increased by 26 and 29 percent.

In Walnut Creek watershed, row crop land use decreased from 69.4 to 54.5 percent between 1992 to 2005 as a result of prairie restoration by the USFWS at the Neal Smith refuge. From 1992 to 2005, an average of approximately 222 acres of prairie were planted each year. As of 2005, 3,023 acres of land in Walnut Creek watershed were planted in native prairie, representing 23.5 percent of the watershed. In the subbasins, restored prairie accounted for 14.3 to 45.9 percent of the land area.

In Squaw Creek, nitrogen applications increased 12.8% over 1990 N applications whereas nitrogen applications in the Walnut Creek watershed decreased 21.4%. Pesticide applications in Walnut Creek watershed were reduced by nearly 28 percent compared to levels in 1990.

Hydrology and Suspended Sediment

Annual precipitation ranged from 25.4 to 41.57 inches at WNT2, 18.6 to 30.2 inches at WNT1 and 14.96 to 35.84 inches at SQW2. Stream discharge varied considerably from year to year with annual discharge varying more than 4-fold between water years 1996 to 2005. Annual discharge in Walnut Creek ranged from 4.31 inches in Water Year 2002 to 16.61 inches in Water Year 1998, whereas discharge in Squaw Creek SQW2) varied from 3.36 to 16.91 inches in the same two years. Average annual discharge for WNT2, SQW2 and WNT1 was similar for all gage sites, ranging between 8.62 to 8.93 inches. Streamflow in both watersheds was controlled in large part by seasonal precipitation patterns and soil moisture conditions, with greatest streamflow typically occurring during rainy periods when antecedent soil moisture conditions are high. During most years, this period included May and June when nearly one-half of the annual total streamflow typically occurred. Streamflow events were often characterized by flashy conditions typical of flow in incised channels. Few consistent patterns were evident in the statistical trends over time.

Suspended sediment concentrations and loads varied widely during the 10-year monitoring period. Total annual sediment export ranged from 3,706 to 18,367 tons in Walnut Creek and from 893 to 20,456 tons in Squaw Creek, with higher average annual loss higher in Walnut Creek (8,384 tons) than Squaw Creek (8,044 tons). Sediment transport through Walnut and Squaw creek watersheds was very flashy, evidenced by most of the annual suspended sediment load occurring during intermittent high flow events. While single day discharge events typically accounted for six to eight percent of the annual discharge, single day suspended sediment loads accounted for 25 to 37 percent of annual sediment total. A 20-day period in any given water year accounted for as much as 98 percent of the annual sediment total. This pattern of rapid conveyance of discharge and sediment loads is typical of incised channels. Greatest sediment transport typically occurred in May and June of each year, when on average these months accounted for 59.2 and 68.2% of the total annual load in Walnut and Squaw Creek watersheds, respectively. Annual sediment loss was slightly higher in Squaw Creek compared to Walnut Creek, averaging 0.69 and 0.65 tons/acre, respectively, with annual sediment yield significantly related to annual discharge.

Suspended sediment concentrations were similar in Walnut Creek and Squaw Creek, with average and median values of 104.1 and 46.0 mg/l at WNT2 and 90.1 and 42.7 mg/l at SQW2, respectively. Suspended sediment concentrations most commonly ranged between 20-50 mg/l, with concentrations within this range approximately 35 to 39 percent of the time. Trends in daily sediment concentrations and loads were mixed and reflected the variable nature of sediment transport. One regression model indicated a decreasing trend in sediment concentrations and loads over time was observed at WNT2 whereas another model indicated an increase over time.

Water Quality Monitoring

Nitrate concentrations have ranged between <0.5 to 14 mg/l at the Walnut Creek outlet (WNT2) and 2.1 to 15 mg/l at the downstream Squaw Creek outlet (SQW2). Mean nitrate concentrations were 1.7 mg/l higher at SQW2 than WNT2, and highest at the upstream monitoring sites in both watersheds, averaging 11.2 mg/l at WNT1 and 12.4 mg/l at SQW1.

Monthly nitrate concentrations exhibited clear seasonality, with higher concentrations occurring during May, June and July. Both Walnut and Squaw Creek watersheds have shown a similar temporal pattern of detection, with higher concentrations observed in the spring and early summer months coinciding with periods of application, greater precipitation and higher stream flow.

Total export of nitrate from Walnut Creek (WNT2) was lower than Squaw Creek (SQW2) averaging 22.0 and 26.1 kg/ha, respectively. The average flow-weighted concentration of nitrate was 8.6 mg/l in Squaw Creek and 10.4 mg/l in upper Walnut Creek but was 4.9 mg/l in lower Walnut Creek.

During the 10-year project, nitrate concentrations significantly decreased in Walnut Creek watershed, both at the watershed outlet and in monitored subbasins. At the Walnut Creek outlet (WNT2), the trend analysis indicated that nitrate concentrations decreased 0.119 mg/l/year or 1.2 mg/l over 10 years when the Squaw Control watershed was utilized as a covariate. Nitrate concentrations decreased 3.4, 1.2 and 2.7 mg/l at WNT3, WNT5 and WNT6 subbasins, respectively. Nitrate concentrations increased 1.9 mg/l over 10 years in the downstream Squaw station SQW2 and 1.1 mg/l over 10 years in the upstream Squaw station SQW1. All subbasins in the Squaw Creek increased in nitrate concentrations, with subbasins SQW4 and SQW5 having quite dramatic increases. Over the 10-year monitoring program, nitrate in surface water in SQW4 and SQW5 subbasins increased 11.6 and 8.0 mg/l, respectively.

Atrazine and DEA were the most commonly detected herbicides in both watersheds with detection frequencies greater than 70 percent. Acetochlor was occasionally detected (up to 27 percent) whereas alachlor and metolachlor were rarely detectable (less than 5%). Cyanazine detections were also rare during the last five years of the project. Concentrations of atrazine often exceeded 1 ug/L during high streamflows in late spring/early summer; however, overall median concentrations of atrazine and DEA were less than 0.3 ug/l. May and June accounted for approximately 80 percent of the export load of atrazine, and the period of April through July accounted for 96 percent of the annual atrazine load. Statistical changes in herbicide concentrations over time were mixed, since both decreasing and increasing trends were observed. Sites WNT3 and SQW2 had decreasing trends in atrazine concentration with respect to time whereas sites WNT5, WNT6, and SQW5 had increasing trends in DEA concentration with respect to time. Other sites had no herbicide trends over time.

Fecal coliform bacteria were detected frequently above the EPA water quality standard of 200 count/100 ml in both watersheds. Elevated detections were occasionally observed at all monitored watersheds with highest fecal coliform counts occurring at any time between May and October during high stream flow periods associated with rainfall runoff. No changes in fecal coliform concentrations were observed during the 10-year monitoring project at downstream Walnut Creek (WNT2). Increases in fecal coliform concentrations were noted in two Walnut subbasins. Similarly, subbasin changes in Squaw Creek watershed did not result in changes in downstream Squaw Creek levels measured at SQW2.

Phosphorus (P) monitoring began in Water Year 2001 and thus five years of monitoring data are available for analysis. Annually, median P concentrations were also fairly consistent, ranging between 0.14 to 0.2 mg/l at SQW2 and 0.17 to 0.2 mg/l at WNT2 for water years 2001 to 2005. The range in annual median P concentrations varied between 0.06 to 0.2 mg/l at all sites. Phosphorus did not change in any of the main stem streams in either Walnut Creek or Squaw Creek. The only statistically significant trend in phosphorus was an increase in the SQW3 subbasin and a decreasing trend in SQW5. Lack of phosphorus concentration trends in five years of monitoring in the watersheds was not unexpected given the episodic transport and variability in P concentrations detected in water.

Biological Monitoring

Quantitative collections from Squaw Creek and Walnut Creek had poor macroinvertebrate colonization during the project. Taxa richness metrics for Walnut Creek initially showed consistent improvement until 2001 after which metrics have steadily declined to lower levels than project inception. The metric measures of community balance showed similar positive trends with values decreasing until 2002, after which values have increased to levels at or higher than project inception levels. However, many of the positive changes in the macroinvertebrate community appeared to be driven by the habitat modification (addition of coarse substrate for a bridge crossing) that occurred at the Walnut Creek sampling site. Metric means were calculated for both streams. Data did not show consistent trends in either watershed. Except for 2001 when large differences were evident, patterns of the four quantitative metrics have been similar between Walnut Creek and Squaw Creek.

Thirty-one species of fish from eight families were collected from Walnut Creek and twenty-two species of fish from six families were collected from Squaw Creek since 1995. The fish community in both streams was dominated by minnows and most of the minnow species collected are considered abundant to common in Iowa streams. Walnut Creek FIBIs ranged from 15 in 1995 to 40 in 1996 and 2002 whereas FIBI scores for Squaw Creek ranged from 21 in 2000 to 38 in 1997. FIBI scores for Walnut or Squaw Creek did not show any visual improvement or decline since 1995. Most FIBIs calculated for Walnut Creek and Squaw Creek were considered fair.

Lessons Learned

The following are some of the lessons learned from the Walnut Creek monitoring project:

- Prairie restoration can be an effective BMP to reduce nitrate concentrations and loads in agricultural watersheds. A nitrate reduction of 0.7 to 3.4 mg/l/10 years was measured in Walnut Creek watershed.
- As demonstrated by other studies, row crop land cover is significantly related to stream nitrate concentrations. Converting row crop to native prairie at the Neal Smith NWR reduced the amount of row crop in the various watershed areas and reduced stream nitrate, whereas converting CRP grass back to row crop in Squaw Creek increased the amount of row crop and greatly increased stream nitrate.
- The rate of nitrate concentration reduction measured in streams will be dependent upon the rate of groundwater flow to transport nitrate water to streams. In the Walnut Creek watershed, slow groundwater flow velocities suggest that nitrate reductions from upland prairie restoration plots will take many decades to be measured in streams. Land use changes

occurring in the floodplain are more likely to have an impact on short-term water quality than those associated with upland settings. Tile drainage accelerates the movement of subsurface water through soils and can possibly accelerate detection of concentration changes through time.

- Headwater regions of watersheds exert a proportionally large effect on watershed NPS export. In Walnut Creek watershed, statistical analyses and synoptic surveys indicate that much of the downstream concentrations of NPS pollutants in Walnut and Squaw Creek watersheds can be explained by upstream contributions. Once the pollutant is discharged into the incised stream network from row crop dominated headwater regions, concentrations remain elevated in the stream. Prairie restoration placed in the core of a watershed served to dilute concentrations from upstream sources.
- It was easier to detect changes occurring in NPS pollutants over time in smaller subbasins than the larger project watersheds. When areas of land use change were isolated at the subbasin scale, substantially greater water quality changes were observed.
- An event-based sampling protocol rather than a set sampling schedule would have been more appropriate to detect changes in herbicides, fecal coliform and phosphorus concentrations over time. A set sampling schedule was useful to characterize concentration ranges and long-term variability, but was not effective in capturing changes in NPS pollutants delivered primarily with runoff.
- Biological monitoring of benthic macroinvertebrates and fish was not sufficiently sensitive to detect any changes in water quality occurring in Walnut Creek watershed. Difficulties included obtaining sufficient colonization in the flashy incised streams, and accounting for the effects of downstream fish populations on measured populations. Biological monitoring may be more appropriate to assess water quality patterns across spatial scales rather than temporal scales less than 10 years.
- Suspended sediment concentrations and loads are difficult to characterize in incised streams that transport most of their sediment loads during infrequent high flows. Event-based monitoring is needed to supplement fixed monitoring to fully characterize sediment transport in these incised streams.
- Characterizing sediment reductions in watershed projects using sediment erosion models does not accurately reflect reduced sediment export. Sediment sources vary in watersheds and streambank erosion can contribute significantly to watershed sediment loads.
- Reducing upland sheet and rill erosion in watersheds without reducing water discharge from these areas will likely shift sediment sources from upland sources to instream sources such as streambanks and streambed.
- A lag time of decades is likely needed to measure changes in sediment export in order to overcome variable climate and historical sediment storage. Paired watershed studies assist in detecting change but consideration must be given to account for differences in sediment sources and delivery to streams.
- Long-term monitoring is needed to capture changes in water quality due to implementation (or abandonment) of conservation practices. If benefits of conservation practices on water quality are to be fully assessed, a combination of intensive monitoring and modeling is recommended.

INTRODUCTION

Background

Nonpoint source (NPS) pollution is a major cause of surface water impairment in the United States. In the Upper Mississippi River basin, more than 1,200 stream segments and lakes appear on the U.S. Environmental Protection Agency (USEPA) listing of impaired waterways (USEPA, 2003). Export of NPS pollution from the Midwestern region of the United States is receiving increasing attention due to concerns regarding excessive nutrient enrichment and eutrophication in streams (Turner and Rabalais, 1994; Vitousek, et al., 1997; Dodds and Welch, 2000; USEPA, 2000) and development of hypoxic conditions in the Gulf of Mexico (Rabalais et al., 1996; 2002; Goolsby et al., 1999; Burkart and James, 1999). Nitrate-nitrogen (nitrate) export from the State of Iowa, located in the middle of the U.S. corn belt region, has been identified as a major contributor to Mississippi River pollutant loads (Goolsby et al., 1999). Average annual export of nitrate from surface water in Iowa was estimated to range from approximately 204,000 to 222,000 megagram (metric tons or Mg), or about 25% of the nitrate that the Mississippi river delivers to the Gulf of Mexico, despite Iowa occupying less than 5% of its drainage basin (Libra, 1998).

Agriculture is the major nonpoint source impacting Iowa's surface waters (IDNR, 2000). Recent assessments indicated agriculture was the primary source of impairment of 93 percent of Iowa's streams and the source of impairment of the majority of lakes and wetlands (IDNR, 2000). Sediment and nutrients have been most frequently identified as the agricultural pollutants causing the greatest water quality impacts, with pesticides and bacteria also identified as important sources (IDNR, 2000).

The amount of agricultural land in a watershed is well understood to be a good predictor of NPS pollution in streams (Hill, 1978; Mason et al., 1990; Jorden et al., 1997b; Schilling and Libra, 2000). In Iowa, average annual nitrate concentrations in rivers can be approximated by simply multiplying a watershed's row crop percentage by 0.1 (Schilling and Libra, 2000). Furthermore, agricultural land use strongly affects the hydrology of watersheds (Schilling and Wolter, 2005). The percentage of row crop land in a watershed largely governs the partitioning of total streamflow into baseflow and stormflow (runoff) components by delivering more total discharge and baseflow to streams per unit area (Schilling and Wolter, 2005). Baseflow, in particular, is significantly related to row crop intensity in Iowa (Schilling, 2005; Schilling and Libra, 2003). Nitrate is primarily delivered to Iowa streams through groundwater discharge as baseflow and tile drainage (Hallberg, 1987; Schilling, 2002a; Schilling and Zhang, 2004).

Considerable research has demonstrated that agricultural conservation practices utilizing perennial cover reduce NPS pollution in streams. Along stream corridors, perennial riparian buffers have been shown to influence the amount, timing and pathways of water and pollutants that move through them (e.g., Peterjohn and Correll, 1983; Jorden et al., 1993; Hill, 1996; Bharati et al., 2002; Lee et al., 2003; Schultz et al., 2005). In field studies, Randall et al. (1997) found that nitrate concentrations in drainage water from alfalfa and perennial grasses were 35 times lower than drainage water from corn and soybean fields. Brye et al. (2000) compared the hydrologic budgets of restored prairie and cultivated corn ecosystems and found that prairie maintained greater soil water content in the soil profile, larger evapotranspiration (ET), and significantly less drainage. Leaching losses of nitrogen and phosphorus were also higher from managed corn systems compared to restored tallgrass prairie (Brye et al., 2001; 2002). On a watershed scale, recent modeling studies have suggested that a conversion of substantial portions of the landscape to perennial cover offers promise for improving water quality

(Nassauer et al, 2002; Coiner et al., 2001; Vache et al., 2002). Vache et al. (2002) predicted that targeted agricultural conservation practices (buffers, wetlands, grassed waterways, filter strips, and field borders) could potentially reduce nutrient loadings by 54-75% and sediment loadings by 37-67%. Dinnes et al. (2002) suggested that diversifying plant rotations in watersheds could better utilize water during vulnerable leaching periods occurring in the spring and fall.

One perennial cover option available to Iowa is reintroduction of grasses to the agricultural landscape (Schilling, 2001; Jackson and Jackson, 2002). Iowa was once part of the tallgrass prairie ecosystem that covered 67.6 million ha in the United States, of which more than 99.6 to 99.9 percent has been lost (Sampson and Knopf, 1994). Although the plowdown of prairies occurred primarily between the 1850's to 1890's (Smith, 1992), perennial cover remained a part of the landscape through crop rotations of sod crops (oats, hay) with annual crops (corn, soybean). The balance of sod versus annual crops was about fifty-fifty through the 1950's (Jackson, 2002). However, from the mid-20th century to present, soybean production has increased dramatically as tractors and nitrogen fertilizers became available. Between 1940 and 2000, soybean production increased from 1,000,000 acres to approximately 11,000,000 acres, so that combined with minor increases in corn production, total row crop area (corn and soybeans) increased approximately 30-40% during this time (Iowa Agricultural Statistics, 2001). Similarly, nitrogen fertilizer use in Iowa significantly increased from 1965 to 1981, generally averaging between 900,000 to 1.0 million tons per year in the 1990's (IAS, 2001).

Removal of perennial vegetation from Iowa's agricultural landscape profoundly affected streamflow characteristics and nitrate concentrations over the 20th century. Baseflow and the percentage of streamflow as baseflow have significantly increased in Iowa over the second half of the 20th century, more than precipitation alone can explain (Schilling and Libra, 2003). Schilling and Libra (2003) hypothesized that one of the main reasons for increasing baseflow in Iowa over the 20th century was converting previously untilled land or other perennial cover crops to annual row crops. In conjunction with the land use change, a two- and three-fold increase in nitrate concentrations has been observed in the Cedar and Des Moines rivers in Iowa during the 1940-2000 period (IDNR, 2001). Schilling and Lutz (2004) estimated that in the Raccoon River in central Iowa, nitrate concentrations could have been increased by 44% from 1916 to 2000 just by increasing baseflow alone.

Need for Project

It is evident that 1) NPS pollution from agriculture is a major problem in Iowa and the agricultural Midwest, and 2) perennial cover in an agricultural ecosystem reduces NPS pollution loading to streams. However, the effectiveness of introduction of perennial cover into an agricultural landscape to reduce NPS pollution in a stream is relatively untested at a watershed scale. Introduction of perennial cover is one of many best management practices (BMPs) that have been implemented in Iowa to mitigate NPS pollution from agriculture. Yet monitoring NPS water-quality improvements resulting from BMPs has rarely been done because it is not an easy task. NPS pollution results from runoff across a landscape with varied land-management practices, resultant NPS impacts measured in perennial streams are typically a mix of effects from many different parcels of land and many different components of management, integrated over many time scales. Hence, it is difficult to document the relationship between improvements in water quality and changes in management practices on a watershed scale. While many projects have been implemented under Section 319 of the Clean Water Act most have had little or no monitoring associated with them. Water quality improvements are generally assumed rather than measured, or estimated using field-scale or watershed models.

Watershed studies that have had adequate monitoring have been less than successful at demonstrating an improvement (Hallberg et al., 1983; Libra et al., 1991; Rowden et al., 1995; Fields et al., 2005; Gale et al., 1993; USEPA, 1990). The Big Spring Demonstration Project in northeast Iowa collected water quality data for over a decade (Hallberg et al., 1983, Libra et al., 1991; and Rowden et al., 1995, 2001). From 1981 to 1993, average rates of nitrogen application on corn within the basin were reduced from 174 to 115 pounds/acre, a 34% reduction, with no loss in yield. Although nitrogen inputs were reduced significantly, relating these declines to changes in Big Spring groundwater remained a problem. The effects of nitrogen reductions, occurring gradually over a decade, were obscured by year-to-year variations caused by climatic variability, particularly the variability of precipitation. In the Sny Magill watershed, results from a 10-year monitoring project in northeast Iowa indicated that turbidity and suspended sediment concentrations were reduced following BMP implementation in the steeply sloping watershed. However, other important water quality indices, such as nutrients, dissolved oxygen, pesticides, benthic macroinvertebrates and fish either indicated no change or a significant increase during the study (Fields et al., 2005). Results from other long-term watershed monitoring projects suggest that implementation of livestock exclusion practices near streams and improved livestock grazing may have an increased probability of detection of stream water quality improvement in watershed monitoring projects (Meals, 2001; Line and Jennings, 2002; McNeil, et al., 2003). Schilling and Thompson (2000) discussed possible reasons for failure to observe water quality changes in long-term watershed monitoring projects and questioned the lag time needed for observing changes, the size and location of land use changes in a watershed, and the appropriateness of the monitoring design.

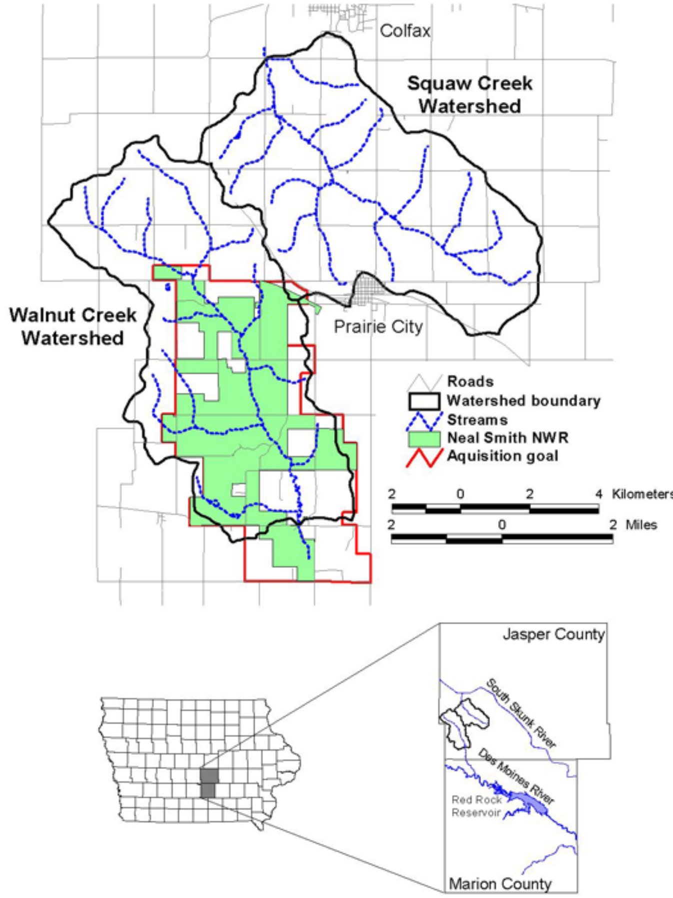


Figure 1. Location map including refuge ownership and future acquisition boundaries.

Walnut Creek Monitoring Project

The Walnut Creek Watershed Restoration and Water-Quality Monitoring Project provides a valuable opportunity to quantitatively measure, on a watershed scale, water quality improvements resulting from large-scale land use changes. The project was established in 1995 as a NPS monitoring program in conjunction with watershed habitat restoration and agricultural management changes implemented by the U.S. Fish and Wildlife Service (USFWS) at the Neal Smith National Wildlife Refuge and Prairie Learning Center (Refuge) in Jasper County Iowa (Figure 1). A large portion of the Walnut Creek watershed is being restored from row crop agriculture to native prairie and savanna. Riparian zones and wetlands are being restored in context, with riparian zones grading from prairie waterways, to savanna, to timbered stream borders (Drobney, 1994). The refuge was established in 1991 with the purchase of approximately 3,600 acres of land that had been intended as the site of a nuclear generator. Future acquisition boundaries comprise 8,654 acres of which more than 5500 acres have been purchased from willing sellers. Figure 1 shows the acquisition boundaries and ownership boundaries of refuge lands.

While it is recognized that large-scale prairie restoration is not a typical nonpoint source management practice, restoration and land management activities occurring at the Neal Smith Refuge are analogous to many traditional BMPs installed in other watersheds. Enrollment of row crop land into the USDA Conservation Reserve Program (CRP) is a widespread conservation practice used throughout Iowa to manage NPS runoff from erodible lands. Use of warm season grasses and forbs for CRP cover is encouraged by many groups (i.e., local conservation boards, Pheasants Forever, Trees Forever). Restoration of natural riparian zones and wetlands at the Neal Smith refuge is also consistent with establishment of riparian buffer systems in other watersheds. Because of the similarity of prairie restoration and riparian zone management to other common NPS conservation practices, monitoring results from Walnut Creek are transferable to other watershed projects.

In 1996 the Walnut Creek Monitoring project was approved by the U.S. Environmental Protection Agency (EPA) as a Section 319 National Monitoring Program project. These projects comprise a small subset of nonpoint source (NPS) pollution control projects funded under the Clean Water Act. The goal of the national program is to support 20-30 watershed projects nationwide that meet a minimum set of planning, implementation, monitoring, and evaluation requirements designed to lead to successful documentation of project effectiveness with regard to water quality protection or improvement. Monitoring of both land treatment and water quality to document improvement is necessary to provide decision makers with information on the effectiveness of NPS control efforts. Currently there are 22 projects, including Walnut Creek, in the national program. National Monitoring Program projects are designed for 10-year timeframes including monitoring before, during and after pollution controls are implemented.

Project Objectives

The objectives of the Walnut Creek Monitoring Project were to: 1) perform comprehensive, long-term NPS monitoring in the Walnut and Squaw Creek watersheds; 2) quantitatively document reduction in NPS pollution over time and other environmental improvements resulting from watershed habitat restoration and land management changes; and 3) use the monitoring data to increase our understanding of what implementation measures are successful and will be helpful in similar areas, and expand public awareness of the need for NPS pollution prevention measures in the State of Iowa.

MONITORING PLAN DESIGN

Study Area

Walnut and Squaw creeks are warm-water streams located in Jasper County, Iowa (Figure 1). Walnut Creek drains 30.7 mi^2 (19,500 acres) and discharges into the Des Moines River at the upper end of the Red Rock Reservoir. Only the upper part of the watershed (12,890 acres) is included in the monitoring project because of possible backwater effects from the reservoir. The Squaw Creek basin, adjacent to Walnut Creek, drains 25.2 mi^2 (16,130 acres) above its junction with the Skunk River. The watershed included in the monitoring project is 18.3 mi^2 (11,714 acres) and does not include the wide floodplain area near the intersection with the Skunk River. Basin characteristics of the

Table 1. Basin characteristics of the Walnut and Squaw creek watersheds.

BASIN CHARACTERISTICS	**Walnut Creek**	**Squaw Creek**
Total Drainage Area (sq mi)	20.142	18.305
Total Drainage Area (acres)	12,890	11,714
Slope Class:		
A (0-2%)	19.9	19.7
B (2-5%)	26.2	26.7
C (5-9%)	24.4	25.0
D (9-14%)	24.5	22.2
E (14-18%)	5.0	6.5
Basin Length (mi)	7.772	6.667
Basin Perimeter (mi)	23.342	19.947
Average Basin Slope (ft/mi)	10.963	10.981
Basin Relief (ft)	168	191
Relative Relief (ft/mi)	7.197	9.575
Main Channel Length (mi)	9.082	7.605
Total Stream Length (mi)	26.479	26.111
Main Channel Slope (ft/mi)	11.304	12.623
Main Channel Sinuosity Ratio	1.169	1.141
Stream Density (mi/sq mi)	1.315	1.426
Number of First Order Streams (FOS)	12	13
Drainage Frequency (FOS/sq mi)	0.596	0.710

Walnut Creek and Squaw Creek watersheds are very similar and make them well suited for a paired watershed design (Table 1).

The Walnut Creek and Squaw Creek watersheds are located in the Southern Iowa Drift Plain, an area characterized steeply rolling hills and well-developed drainage (Prior, 1991). The soils and geology of the two watersheds are similar (Table 2). Soils within the Walnut and Squaw Creek watersheds fall primarily within four major soil associations: Tama-Killduff-Muscatine; Downs-Tama-Shelby; Otley-Mahaska and Ladoga-Gara (Nestrud and Worster, 1979). Dominant soil taxa are indicated in Table 2; these soil taxa account for 82% of the soils found in the Walnut basin and 78% of the soils found in the Squaw basin. Tama and Muscatine soils are found primarily in upland divide areas, whereas Ackmore soils are associated with bottomlands. Killduff, Otley and Ladoga-Gara soils are found developed in slope areas. Most of the soils are silty clay loams, silt loams or clay loams formed in loess and till. Moderate to high erosion potential characterizes many of the soils and both watersheds contain equal amounts of highly erodible land (Table 2).

Loess mantled pre-Illinoian till typifies much of the geology of the Walnut and Squaw creek watersheds. Both watersheds are mantled primarily by loess in upland areas. Outcrops of pre-Illinoian till and Late Sangamon paleosols are occasionally found in hillslope areas, whereas alluvium dominates the shallow subsurface of the main channels and second order tributaries. Pre-Illinoian till underlying most of the watersheds is 20 to 100 feet thick. Bedrock occurs at an approximate elevation of 850 to 700 feet above mean sea level and is primarily Pennsylvanian Cherokee Group shale, limestone, sandstone, and coal.

Table 2. Soil characteristics in the Walnut and Squaw creek watersheds.

Soil Characteristics	Walnut Creek		Squaw Creek	
	Acres	Percent	Acres	Percent
Soil Parent Material:				
Alluvium	2043.87	15.86	2050.90	17.51
Eolian Sand			245.15	2.09
Weathered Shale	14.88	0.12		
Local Alluvium	192.79	1.50	383.34	3.27
Gray Paleosol	405.27	3.14	157.86	1.35
Loess	6155.89	47.75	6312.66	53.89
Loess and Local Alluvium	24.99	0.19	27.62	0.24
Loess-gray or gray mottles	2073.92	16.09	1245.56	10.63
Paleosol-reddish	13.27	0.10	7.96	0.07
Sandy Alluvium	168.52	1.31		
Till (pre-Illinoian)	1773.99	13.76	1255.80	10.72
Highly Erodible Land	6935.11	53.78	6226.13	53.57
Dominant Soil Taxa:				
Tama	2528.92	19.61	4018.23	34.29
Killduff	1889.72	14.66	1242.04	10.66
Muscatine	1038.25	8.05	548.54	4.68
Otley-Mahaska	1396.53	10.83	999.57	8.53
Shelby-Adair	508.47	3.94	986.67	8.42
Ackmore, Ackmore-Colo	1612.18	12.50	1309.69	11.17
Ladoga-Gara	1556.96	12.08	40.56	0.35

In the floodplains of Walnut and Squaw Creeks, Holocene alluvial deposits consist of stratified sands, silts, clays and occasional peat. A detailed cross section across the Walnut Creek floodplain in the central portion of the watershed identified six principal stratigraphic units (Schilling et al., 2004; Figure 2). Three alluvial units comprise members of the DeForest Formation (Camp Creek, Roberts Creek and Gunder members; Bettis, 1990, Bettis and Littke, 1987), a fourth alluvial unit was considered older than the oldest member of the DeForest Formation (herein termed “pre-Gunder”), and two units occupy hillslope locations (loess and pre-Illinoian till). Elsewhere in the Walnut Creek floodplain, post-settlement alluvial and colluvial materials (Camp Creek Member) deposited in the stream valley ranged from approximately two to six feet in thickness (Schilling and Wolter, 2001). The alluvial stratigraphy found in the Walnut Creek riparian corridor is similar to other third and fourth order watersheds in the Southern Iowa Drift Plain landscape region (Bettis and Littke, 1987).

Monitoring Design

The project utilizes a paired-watershed design as well as upstream/downstream comparisons for analysis and tracking of trends. Paired watershed studies offer increased statistical power to detect changes in water quality from land treatment (Loftis et al., 2001; USEPA, 1993; Clausen and Spooner, 1993; Spooner et al, 1987). The approach typically involves two monitoring periods (calibration and

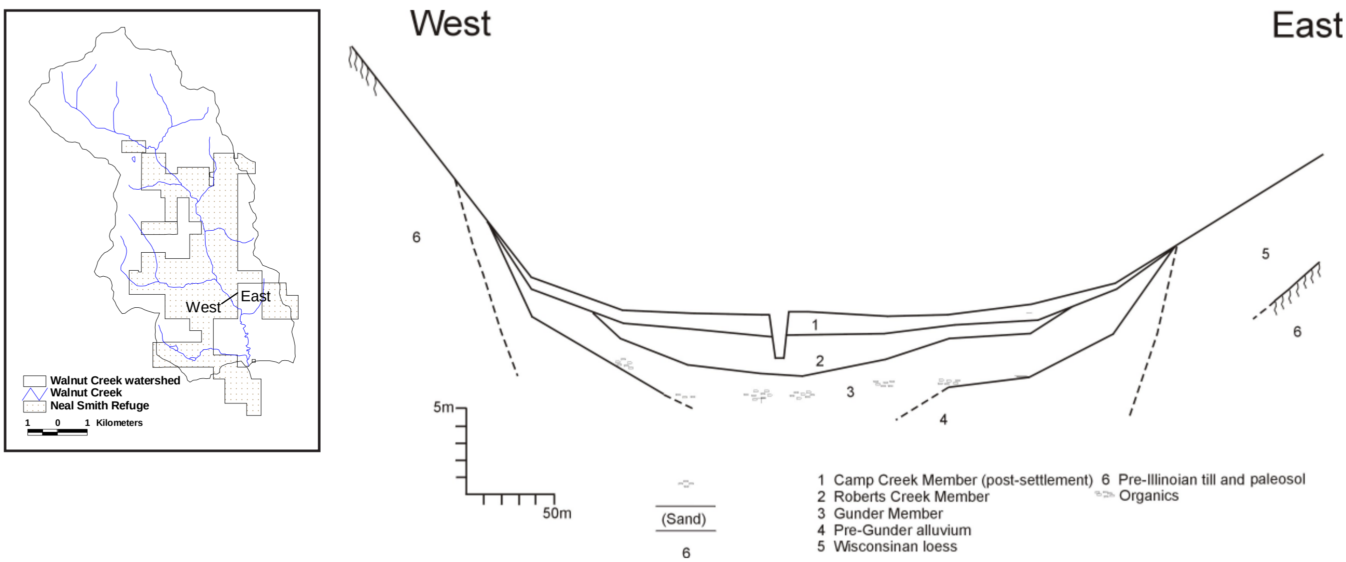


Figure 2. Cross section of alluvium in Walnut Creek floodplain.

treatment), and two watersheds (treatment and control). In this study, the Walnut Creek watershed (treatment) is paired with the Squaw Creek watershed (control). The watersheds are well suited to such a design since they share a common basin divide and have similar basin characteristics (Tables 1 and 2). In typical paired watershed studies, two similar watersheds are monitored for a calibration period and then a treatment is imposed on one of the watersheds (i.e., prairie restoration in Walnut Creek). A change in the relation of a variable of interest (e.g., nitrate) between treatment and control watersheds is then considered a treatment effect (Loftis et al., 2001).

This project differed from typical paired watershed studies because a calibration period was not utilized for two principal reasons: 1) pretreatment data collection (as reported in Schilling and Thompson, 1999) was not sufficient to derive relations between treatment and control watersheds during a pre-refuge calibration period; and 2) land treatment implemented in the Walnut Creek watershed has gradually occurred throughout the entire monitoring period. For these reasons, a gradual change model was used in the paired watershed study rather than a typical pre/post paired study (see statistics section).

Several subbasins were also monitored in both watersheds during the project to allow comparisons of differential implementation of treatments over time, and for analyzing their incremental contributions to the overall basin response. Some of these subbasins are primarily within refuge lands; others contain a high percentage of cropland. The upstream sampling point on Walnut Creek is above the refuge boundaries and allows an evaluation of upper basin effects on water quality.

Four basic components comprised the project: 1) tracking of land cover and land management changes within the basins, 2) stream gaging for discharge and suspended sediment at two locations on Walnut Creek and one on Squaw Creek, 3) surface water quality monitoring in the Walnut and Squaw Creek watersheds, and 4) biomonitoring for aquatic macroinvertebrates and fish in Walnut and Squaw Creeks. A fifth project component, groundwater quality and hydrologic monitoring, was discontinued in WY1999. Sampling stations located in Walnut and Squaw Creek basins are shown on Figure 3.

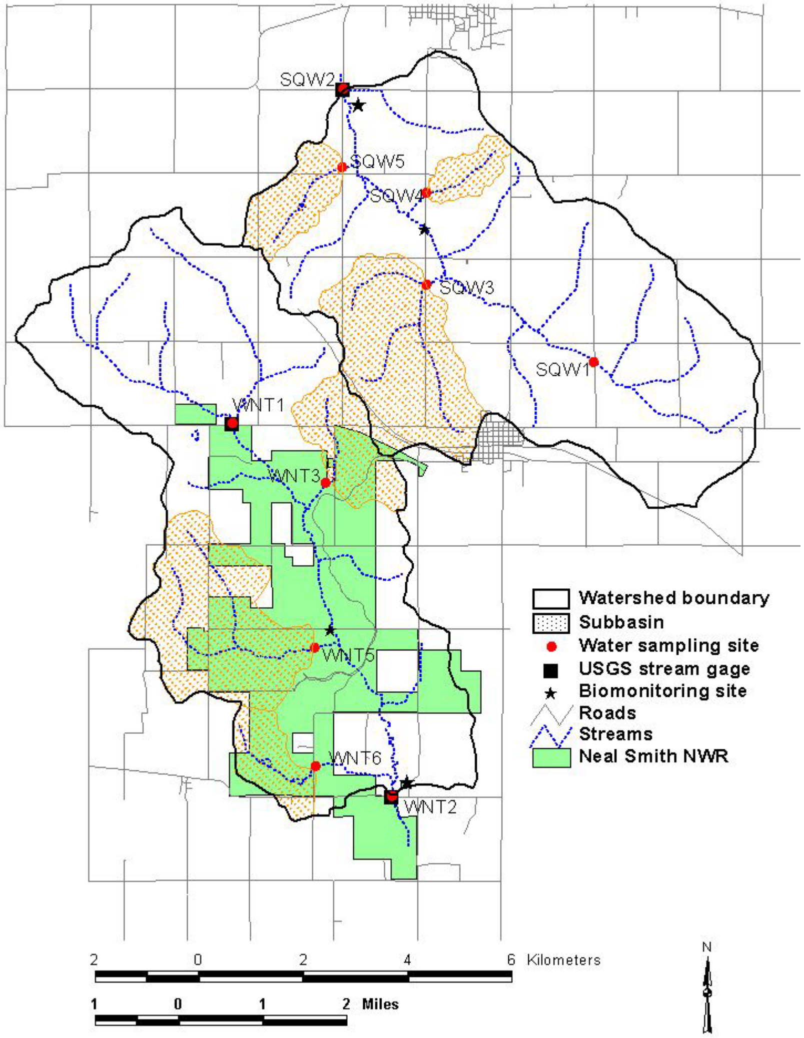


Figure 3. Sampling locations in Walnut Creek and Squaw Creek watersheds.

METHODS

Land Cover Tracking

Land cover in the Walnut and Squaw Creek basins was tracked using a combination of methods. Initially, land cover data from both watersheds was compiled using a combination of plat maps, aerial photographs and field surveys. Data from 1994 and 1995 was derived primarily from plat maps and aerial photographs, whereas 1996 through 1998 data were compiled mainly from annual field surveys. However, annual field surveys did not prove especially effective for monitoring land use changes at a watershed scale due to inconsistencies in land use designations and field boundaries. From 1998 to 2004, statewide inventories of land use completed in 2000 and 2002 were used to track land use in the Walnut and Squaw creek watersheds. Land cover data was interpreted from Landsat satellite imagery taken in 2000 and 2002. Land cover data from the satellite imagery was available at a 30-meter resolution. However, land cover tracking with this method was inconsistent at the scale of the project watersheds without substantial ground truthing. For example, imagery evaluated for the 2002 land cover map interpreted recently burned prairie plantings as forest.

In 2005, detailed land use/land cover mapping was conducted by field survey in

Walnut and Squaw creek watersheds by a retired NRCS employee (Charlie Kiepe). Using a map of common land units (CLUs) in the Walnut and Squaw Creek watersheds, a tablet PC was used with a GIS interface to enter land cover and conservation practices descriptions for each CLU into the GIS database. Conservation practices mapped included tillage practices, grass waterways, terraces, CRP grasslands, and other common USDA-funded conservation practices. The results of the field mapping project were used as the final 2005 land cover for the monitoring project. In order for land cover tracking to be consistent with the beginning of the project, the 2005 CLU boundaries were overlain on 1990 aerial photographs for the Walnut and Squaw Creek watersheds and the 1990 land cover for each CLU was entered into the GIS database.

USFWS personnel have tracked prairie planting areas and locations of cooperative farmer rental ground in the Walnut Creek watershed. GIS coverages of prairie planting sites and rental lands were made available to track annual land use changes within the refuge boundary.

USGS Stream Gaging Stations

Standard USGS gaging facilities were located at three main stem sites (WNT1, WNT2, and SQW1; Figure 1). Stage was monitored continuously with bubble-gage sensors (fluid gages) and recorded by data collection platforms (DCP) and analog recorders (Rantz et al., 1982). The DCPs digitally recorded rainfall and stream stage at 15-minute intervals. The equipment was powered by 12 volt gel-cell batteries which were recharged by solar panels or battery chargers run by external power. Reference elevations for all USGS gage stations were established by standard surveys from USGS benchmarks. Stage recording instruments were referenced to outside staff plates placed in the streambeds, or to type-A wire-weights attached to the adjacent bridges. Rainfall was recorded using standard tipping bucket rain gages.

Stream discharge was computed from the rating developed for each site (Kennedy, 1983). The stream-gaging and calibration is performed by USGS personnel, using standard methods (Rantz et al., 1982; Kennedy, 1983). Current meters and portable flumes were used periodically to measure stream discharge and refine the station ratings.

Suspended Sediment

Suspended sediment samples were collected daily by local observers and weekly by water quality monitoring personnel. The observers collected depth integrated samples at one vertical section at one point in the stream using techniques described by Guy and Norman (1970). Samples were collected daily at all three stations. During storm events, suspended sediment samples were collected with an automatic water-quality sampler installed by the USGS at the gaging stations. Sampling was initiated by the DCP when the stream rose to a pre-set stage, and terminates when the stream fell below this stage. Suspended sediment concentrations were determined by the U.S. Geological Survey Sediment Laboratory in Iowa City, Iowa, using standard filtration and evaporation methods (Guy, 1969). Discharge, rainfall, and sediment data are stored in the USGS Automatic Data Processing System (ADAPS) and published in the Iowa District Annual Water-Data Report.

Chemical Parameters

Table 3 shows the sampling sites, analytes, and frequency used for each water year. Actual sample collection has occasionally varied from this schedule in response to field conditions and precipitation patterns. Temperature, pH, conductivity, dissolved oxygen, reduction-oxidation potential (redox), and turbidity were measured in the field; all other analyses were performed by the University Hygienic Laboratory (UHL) using standard methods and an EPA-approved QA/QC plan (Thompson et al., 1995).

Table 3. Summary of sampling locations, parameters and frequency.

Sampling Location	Parameters	Frequency
WNT1, WNT2, SQW2	Stage/Discharge, Suspended Sediment	Daily
WNT1, WNT2, WNT3, WNT5, WNT6, SQW1, SQW2, SQW3, SQW4, SQW5	Fecal Coliform, Anions, Phosphorus, Common Herbicides, Temperature, Conductivity, Dissolved Oxygen, Turbidity, pH	April (2), May (4), June (4), July (2), August (2), September (2)
WNT1, WNT2, SQW1, SQW2	Fecal coliform, Anions, Phosphorus, Common Herbicides, Temperature, Conductivity, Dissolved Oxygen, Turbidity, pH	January, March, July, August, September, October, November
Biomonitoring sites (two sites in each watershed)	Biomonitoring	Annually (Aug)

Note: Number of samples collected per month indicated under frequency column.

Biomonitoring

The purpose of the biomonitoring was to document the changes in the aquatic vegetation, fish and macroinvertebrate populations of Walnut Creek as a result of the land use and management changes implemented in the watershed. Two biomonitoring sites were established in each watershed, one site was located at the watershed outlet near the gaging site and a second site was located at a midreach location (Figure 3). Details regarding the biological monitoring procedures are provided in the 2005 Biological Monitoring Report (UHL, 2005).

Statistical Methods

Statistical analyses were performed according to the guidelines of Spooner et al. (1987) and Grabow et al. (1998, 1999). To test for the gradual change in chemical concentrations over time a multiple linear regression analysis was performed. A simplified form of the equation is given by:

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2$$

where Y is either the water quality variable or log of the variable for the treatment watershed (Walnut Creek), X_1 is the same water quality variable (or log) for the control watershed (Squaw Creek), and X_2 is elapsed time, and β_0, β_1, and β_2 are regression parameters. In this equation, the estimate of β_2 indicates the magnitude of change over time. By including covariates (e.g., variable X_1), the analysis blocks out much of the hydrologic variability and the change should be isolated to the effect of treatment, which in this case is being modeled as time (X_2). In some cases, multiple covariates were used to develop the regression equation, including discharge or baseflow, upstream and control chemical concentrations and seasonality.

All data were evaluated for normality and were log10 transformed if the data did not fit a normal distribution (skewness >1). The time-series data were also examined for temporal autocorrelation. Temporal autocorrelation is the correlation of values from the samples taken on one day with samples taken from previous sample dates. Autocorrelation is common with environmental data since today's sample is not independent from yesterday's values. Typically, weekly and less frequent sampling frequencies result in a time series that an Autoregessive, lag 1, or AR(1) for the residuals from a regression over time. That means that most of the non-independence between samples dates can be explained by the correlation of each sample with previous samples. The higher the autocorrelation coefficient, the more correlation between previous sample dates, and therefore, the less independent the data points. Each sample point adds more information, but not as much as one additional degree of freedom. A low autocorrelation indicates more "flashy" data.

The Durbin-Watson test was used to examine the degree of autocorrelation in time-series data. The Durbin-Watson *d* statistic indicates positive autocorrelation and has a range from 0 to 4. Uncorrelated residuals generate a *d* statistic of 2. Positively autocorrelated residuals generate a *d* statistic between zero and two. The closer the correlation is to one (highly autocorrelated), the closer the *d* statistic is to zero. Thus it was important to obtain a *d* close to the value of 2.0.

Various procedures were used to select the most appropriate model for evaluating trends in the discharge and water quality data. For daily environmental samples, sometimes the AR(1) time series model is sufficient; other times, a moving average component needs to be added. Moving average states that one sample is only related to the previous sample and not samples before that. Moving average rarely fits daily observations, but a combination of autoregessive and moving average or ARMA models sometimes are appropriate for daily measurements. To identify an AR(1) or Autoregressive, Lag 1 time series, the Autocorrelation Function (ACF) graph starts with high values and trails off exponentially with increasing lags. An AR(1) time series also have a distinctive Partial Autorrelation Function. For AR(1) series, the values drops off to 0 after lag 1. If AR(1) is the correct model, the residuals from a AR(1) time series model will be 'white noise' or will have no significant autocorrelation. The Walnut Creek project data were found to be best described by the AR(1) model and for consistency, this model was used for all trend tests.

Corrections for autocorrelation were sometimes accomplished by the use of explanatory variables. Additional correction for autocorrelation was performed by using time series analysis. To determine if added correction for autocorrelation was required by the use of time-series analysis, the tests for autocorrelation were performed again on the residuals from the multivariate regression models.

It was also important to adjust the overall trends for seasonal patterns. It was evident from the monitoring program that some parameter data exhibited strong seasonality, with peaks occurring in May/June and late fall. Thus, seasonal adjustments between each month were made to account for the seasonality in the statistical trend analysis. Seasonality could be seen at higher lags in the ACF. A cycle or sin/cosine option was not used because the cycles were not of uniform width and the bimodal peaks were not the same magnitude. Essentially, the overall trend data were 'corrected' for the average mean value of all the samples taken in a given month over time. This allowed for more refined comparisons over time. By adding a 'month' grouping or class variable to the statistical models, tests could be made to adjust for changes between months, but retain nearly the entire degrees of freedom and account for the variations due to seasonality in the statistical models. The months where the overall trend had the greatest magnitude could then be assessed.

An added feature of the seasonal analysis was the ability to calculate the least square means (LSMEANS) for each month (e.g., the average value adjusted for each month evaluated on a comparable basis). It should be noted that this was not the average values for each month since it accounts for differing sample frequencies and adjusts for any trends that may have occurred over time. PROC AUTOREG in SAS 9.1 was utilized to run the time series regressions.

Other specific statistical methods and analyses conducted on hydrologic and chemical data are presented in their respective sections.

Hydrograph Separation and Chemical Loads

Hydrograph separation into baseflow and runoff components was performed on streamflow data collected at the three USGS gaging sites using an automated method developed by Sloto and Crouse (1996). A local-minimum method was utilized, which essentially connects the lowest points on the hydrograph and provides estimates of daily baseflow discharge between local minimums by linear interpolation (Sloto and Crouse, 1996). Daily runoff discharge was determined at each stream gauge site by subtracting daily baseflow discharge from daily streamflow discharge.

The USGS program ESTIMATOR was used to estimate daily loads of solutes at the three stream gaging sites. The ESTIMATOR program utilizes a Minimum Variance Unbiased Estimator to implement a seven-parameter regression model based on the relationship between log-flow and log-concentration (Cohn et al, 1989, 1992; Gilroy et al., 1990). Daily chemical load data were tabulated and summarized by month and water year. Load data were normalized on a unit area basis by dividing the total annual load at each gauging site by the watershed area above the gage. In the case of Walnut Creek watershed, the load per unit area between the two gage sites was determined by subtracting the load estimated at WNT1 from WNT2. Flow-weighted concentrations were calculated by dividing the daily constituent load by daily discharge.

LAND RESTORATION IMPELMENTATION

Cropland Management Plan

A Cropland Management Plan was prepared by the USWFS in 1993 to guide the rapid conversion of traditional row crop areas to native, local ecotype habitat (USFWS, 1993). The goal has been to restore the land as rapidly as possible, although the rate at which refuge development is occurring has varied with political, ecological and operational needs of the refuge. As refuge development takes place, various tracts of ground currently in crops are removed from row crop production and converted to native habitat.

Land currently owned by the Refuge but still farmed is rented to area cooperative farmers on a cash-rent basis. At the end of each crop year, a determination is made of which tracts to remove from row crop production. Farmers are notified of this decision and required to discontinue the farming practices on that particular tract. Criteria for selection is based on what type of ground is needed for prairie/savanna reconstruction. Refuge cropland is managed by conventional crop rotation of corn and soybeans. No-till production methods are mandatory whereas other management methods are more prescriptive, including soil conservation practices, nutrient management through soil testing, yield goals and nutrient credit records.

Herbicide And Fertilizer Management

It is the ongoing intent of the Refuge to move towards a reduced chemical dependency for the cooperating farmers on refuge ground. All chemicals and application rates are approved prior to application to minimize adverse impacts on non-target plants and animals. Use of chemicals not on the “pre-approved” list may be requested only after demonstrating that the intended use is consistent with an Integrated Pest Management Plan and crop scouting indicates a favorable cost/benefit ratio. All cooperative farmers are required to enter into a contract for crop scouting services for pest management. The following list of procedures for herbicide and fertilizer management are followed on Refuge-owned land (USFWS, 1993):

Table 4. Summary of land cover in Walnut and Squaw creek watersheds (1990 and 2005).

Watershed and Subwatershed	Basin Size (acres)	Year	Row Crop	Prairie	Grass (1)	Woods	Artificial (2)	Other (3)
Walnut Creek	12,891.0	1990	69.4		20.75	5.09	4.54	0.22
(WNT2)		2005	54.54	25.42	11.06	4.08	4.90	
WNT1	4,312.5	1990	75.32		18.50		5.72	0.46
		2005	83.22		9.70		7.08	
WNT3	731.3	1990	71.34		15.59	2.30	10.77	
		2005	43.88	35.71	7.67	1.85	10.89	
WNT5	1,964.6	1990	77.54		16.22	2.21	4.03	
		2005	45.84	45.90	3.96	0.21	4.09	
WNT6	497.8	1990	74.81		10.77	10.62	2.03	1.77
		2005	71.79	14.31	1.92	10.62	1.35	0.01
Squaw Creek	11,622.0	1990	71.41		21.73	1.53	5.12	0.21
(SQW2)		2005	80.62		12.28	1.40	3.46	2.24
SQW1	2,876.0	1990	85.61		9.41	0.03	4.95	
		2005	89.10		5.91	0.03	3.2	1.76
SQW3	1,859.3	1990	67.24		22.13	1.62	9.00	0.01
		2005	72.46		15.74	1.41	7.72	2.67
SQW4	292.1	1990	34.64		64.26		1.10	
		2005	60.63		38.27		0.03	1.07
SQW5	585.7	1990	53.71		42.55		3.49	0.25
		2005	82.20		14.06		0.67	3.07

(1) includes cool season grasslands, pasture, CRP, alfalfa
(2) includes farmsteads, railroads, roads and urban areas
(3) includes cemeteries, golf course, ponds

1. No fall application of fertilizer is allowed.
2. No anyhydrous ammonia has been allowed since 1993; only liquid fertilizer is permitted. Care in application is exercised to avoid runoff into wetlands or riparian areas.
3. A maximum of 100 pounds of liquid nitrogen per acre is allowed on conventional rotation corn ground.
4. Post emergent and banding application of fertilizer is required because this process increases the potential for immediate plant uptake and decreases leaching.
5. Post emergent herbicide is required and pre-emergent herbicide is not allowed. This decreases chances for leaching, encourages herbicides for target species and prevents broad spectrum use.
6. As land use and vegetation type changes occur during restoration, use of pesticides has decreased; however, there are some long-term needs for certain pesticides to manage specific problem areas. These are addressed on a case-by-case basis as they become known.

Land Use

In 1990, land use in both Walnut and Squaw Creek watersheds was dominated by row crops of corn and soybeans (Table 4). Similar row crop percentages were evident in both watersheds, with 69.4 percent row crop in Walnut Creek and 71.4 percent in Squaw Creek. Grassland cover, including cool season grasslands, pasture and CRP comprised 20.75 and 21.73 percent of the watershed area, respectively (Table 4). Walnut Creek watershed contained more woodland than Squaw Creek (5.09 versus 1.53 percent). In the subbasins, land use in Walnut Creek subbasins (WNT1, WNT3, WNT5 and WNT6) ranged between 71.3 and 77.6 percent row crop. Row crop land

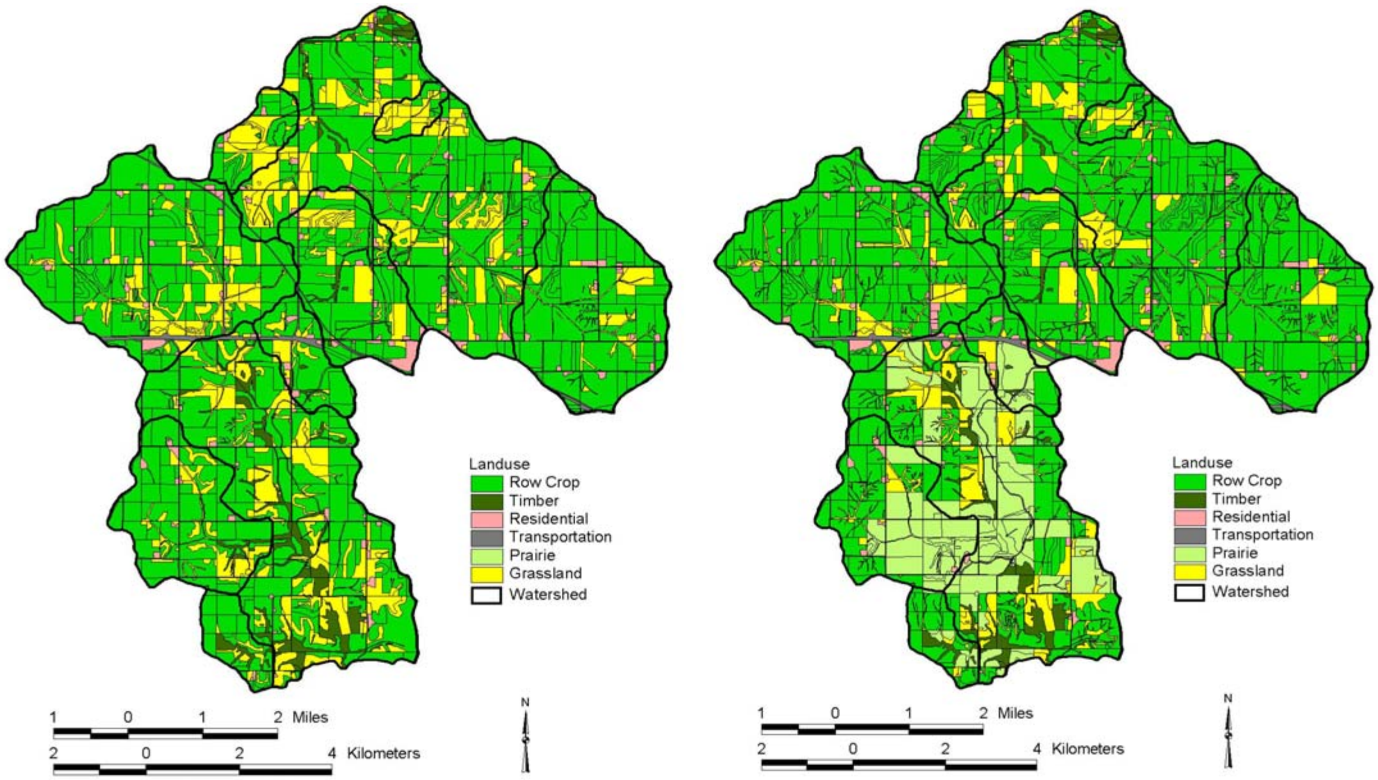


Figure 4. Land cover in 1990 and 2005 in the Walnut Creek and Squaw Creek watersheds.

use in Squaw Creek subbasins in 1990 ranged from 34.6 percent in SQW4 to 85.6 percent in SQW1 (Table 4).

From 1990 to 2005, major changes in land cover occurred in both Walnut and Squaw creek watersheds and the percentage of land in row crops in the watershed diverged from one another (Figure 4). From similar row crop percentages in 1990 (2.4% difference between Walnut and Squaw), by 2005, Squaw Creek watershed contained 26.1 percent more row crop than Walnut Creek. Squaw Creek showed an increasing trend of row crop land use whereas row crop in Walnut Creek significantly decreased (Figure 5). Two major changes in land management account for these diverging patterns. In Squaw Creek, the 9.2 percent increase in row crop area from 1990 to 2005 was likely due the passage of the Freedom to Farm Act in 1996 that appeared to have substantially increased row crop production. Lands previously categorized as grasslands enrolled in the Conservation Reserve Program (CRP) were converted to row crop production, primarily occurring between 1996 and 2000 (Schilling et al., 2002). This trend was particularly evident in subbasins SQW4 and SQW5 where row crop percentage increased by 26 and 29 percent, respectively, with a corresponding decrease in grass land cover (Table 4). Land use in Walnut Creek was not immune to this trend in areas less affected by refuge activities. In upper Walnut Creek watershed (WNT1), row crop acreage increased by eight percent between 1990 and 2005.

In Walnut Creek watershed, row crop land use decreased from 69.4 to 54.5 percent between 1992 to 2005 as a result of prairie restoration by the USFWS at the Neal Smith refuge (Table 4). From 1992 to 2005, an average of approximately 222 acres of prairie were planted each year, with areas planted in 1994 and 1995 exceeding 400 acres (Figure 6).

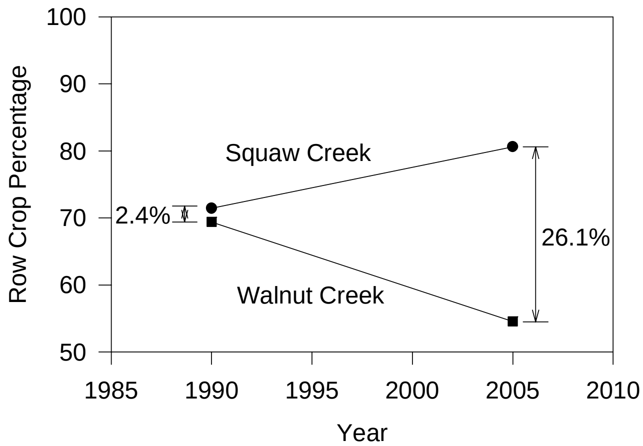


Figure 5. Change in row crop land cover in the Walnut Creek and Squaw Creek watersheds from 1990 to 2005.

As of 2005, 3,023 acres of land in Walnut Creek watershed were planted in native prairie, representing 23.5 percent of the watershed. This percentage is slightly less than the prairie percentage noted by the land use tracking using CLU boundaries (25.4 percent, Table 4). The difference is likely due to assigning blocks of CLUs to prairie when they might be a combination of prairie and non-prairie land cover. The truest assessment of the amount of prairie established in the Walnut Creek watershed is provided by the land use tracking by the Neal Smith refuge that accounted for 3,023 total acres. However, the percentage of prairie in Walnut Creek watershed in 2005 does not equal the reduction of row crop percentage from 1990 to 2005 because other land covers in addition to row crop land were converted to prairie by the Neal Smith refuge. Comparing locations of prairie plantings in 2005 to their location and land cover in 1990, GIS analyses suggest that prairie was planted into formerly row crop ground approximately 73.2 percent of the time. Prairie plantings were also placed in land covers previously grassland (21 percent) and even timber (3.8 percent). Hence, approximately 2,213 acres of row crop ground in 1990 were converted to prairie by 2005, representing 17.2 percent of the watershed.

In the subbasins, restored prairie accounted for 14.3 to 45.9 percent of the land area (Table 4). The greatest percentage of prairie conversion occurred in subbasin WNT5 where prairie accounted for 45.9 percent of the watershed area.

The amount of land owned by the refuge but farmed on a cash-rent basis totaled 479 acres in 2005, or 3.7 percent of the watershed. The remaining land within the refuge boundary in the watershed consists of cool season grass or woods and comprises approximately 1,261 acres (9.8 percent). As of 2005, the USFWS controlled approximately 36.95% (4,763 acres) of the Walnut Creek watershed above the WNT2 gaging station.

Nitrogen Application Reductions

Land use changes have significantly reduced nitrogen loadings in Walnut Creek watershed. With the conversion of row crop areas to native prairie and mandatory reduced nitrogen applications on refuge-owned croplands, reduced nitrogen loadings in Walnut Creek watershed have invariably occurred. However, accurately quantifying nitrogen reductions is problematic. A major confounding factor is the changing land use picture in the control watershed (Squaw Creek), where row crop land use has increased nearly 9.2 percent since 1990. Interpreting nitrogen loading reductions in the treatment watershed is difficult with a moving baseline condition indicated by the control watershed. Earlier estimations of nitrogen load reductions in Walnut Creek watershed were based on land use conditions in 1992 as the baseline condition (Schilling and Thompson, 1999). However, baseline land use conditions in 1992 do not represent conditions in 2005 since row crop land use increased substantially post-1996 with passage of the Freedom to Farm Act.

Nevertheless, nitrogen load reductions in Walnut Creek watershed were estimated using some of the same assumptions used previously (i.e., Schilling and Thompson, 1999; Schilling et al., 2002) and some hypothetical scenarios of land use conditions. First, a control condition in

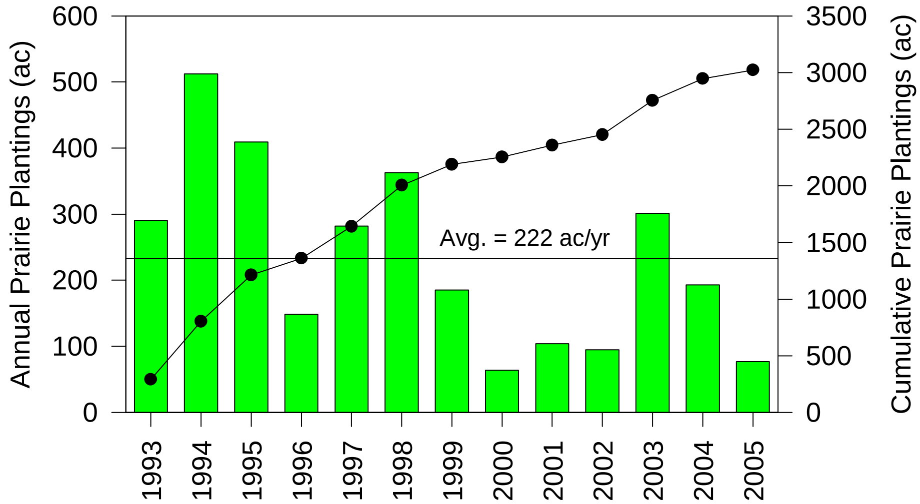


Figure 6. Annual and cumulative prairie plantings in Walnut Creek watershed.

Squaw Creek watershed was established. In 1990, 71.4% of the land in Squaw Creek watershed consisted of row crop. Schilling and Thompson (1999) showed that corn is the predominant row crop approximately 57% of the time, corresponding to a frequency of nearly two out of every three years in corn rotation. This percentage was confirmed by land use tracking in 2005 by CLU; corn again consisted of 57 percent of the total row crop in the Squaw Creek watershed. Typical nitrogen application in farmland around Prairie City was estimated to be 150 lbs/acre (Schilling and Thompson, 1999). Thus the amount of nitrogen applied to Squaw Creek watershed in 1990 was estimated by the following equation:

(11,622 acres) x (71.41% RC) x (57% corn) x (150 lbsN/ac) = 709,885 lbs N (1)

Using the same equation for 2005 land use conditions and substituting 80.62% row crop for the 71.41% value, suggests that nitrogen applications in 2005 were 801,105 lbs, or a 12.8% increase over 1990 N application. In terms of unit loads, nitrogen application increased from 61.1 lbs/acre in 1990 to 68.9 lbs/acre in 2005.

Using the same approach for the Walnut Creek watershed, the Walnut Creek baseline condition in 1990 was estimated to be:

(12,981 acres) x (69.40% RC) x (57% corn) x (150 lbsN/ac) = 764,913 lbs N (2)

In 2005, the percentage of row crop was reduced to 54.54 percent, thus reducing the nitrogen applications in the Walnut Creek watershed to 601,129 lbs N, or a 21.4% decrease over 1990 N application levels. Nitrogen application rates decreased from 66.4 lbs/acre in 1990 to 46.6 lbs/acre in 2005.

The amount of nitrogen application reduction provided by prairie restoration in Walnut Creek watershed was also estimated by considering the amount of row crop acres converted to prairie. From above, 2,213 acres out of total prairie plantings of 3,023 acres were established on 1990 row crop land. Using the same assumptions as above, prairie restoration from 1992 to 2005 reduced N applications as follows:

(2,213 acres) x (57% corn) x (150 lbs N) = 189,220 lbs N (3)

The amount of nitrogen reduction provided by prairie restoration by the Neal Smith refuge is higher than the difference calculated from comparing the 1990 and 2005 land covers (163,784 lbs N). In addition, 479 acres of row crop land were owned by the refuge in 2005 but rented to area farmers on a cash-rent basis.

In these areas, N applications were reduced from 150 lbs/acre to 100 lbs/acre. Assuming a 50% corn rotation in these areas mandated by the refuge, rental farmlands in 2005 reduced N applications by an additional 11,975 lbs N (479 ac x 50% corn x 50 lbs N). However, nitrogen application reductions provided by prairie conversion and reduced N application rates at the Neal Smith refuge are offset by the otherwise increasing trend in row crop land use in non-refuge portions of the watershed. For example, in the WNT1 subbasin, the 7.9 percent increase in row crop land cover from 1990 to 2005 would have contributed an additional 32,800 lbs of nitrogen per year in the watershed. Subtracting the additional N from WNT1 from the reductions provided by prairie restoration at the Neal Smith refuge (189,220 lbs) brings the estimated nitrogen application reductions resulting from the refuge in general agreement with the estimated total application reduction for the watershed (156,408 lbs N).

It is evident that a confounding factor in the analysis of nitrogen reductions is the otherwise increasing trend of row crop land cover in Squaw Creek and on non-refuge lands in Walnut Creek. Hence it is difficult to develop a true paired comparison of nitrogen application loads between Walnut and Squaw creek watersheds since N rates increased 12.8 percent in Squaw Creek watershed and decreased 21.4 percent in Walnut Creek. If the nitrogen application rates between Walnut and Squaw were compared in 1990, the rate of N application in Squaw Creek watershed was 92 percent of Walnut Creek watershed (61.1 compared to 66.4 lbs N/acre). In 2005, the ratio of nitrogen applications in Squaw Creek compared to Walnut Creek was 1.48 (68.9 compared to 46.6 lbs N/acre, respectively). Thus, one measure of the comparison in nitrogen loading reductions in Walnut Creek watershed compared to Squaw Creek would suggest that N applications have been reduced by 56 percent in Walnut Creek relative to Squaw Creek watershed (negative eight percent difference in 1990 plus 48 percent difference in 2005).

A second measure of quantifying reduced nitrogen applications in Walnut Creek watershed relative to Squaw Creek used the paired Squaw Creek land cover from 1990 and 2005 to adjust a hypothetical Walnut Creek baseline land cover. Provided that row crop land use in the Walnut Creek watershed increased by the same percentage as Squaw Creek from 1990 to 2005 (9.2%), nitrogen application loads in Walnut Creek watershed in 2005 would have been approximately 866,314 lbs N (assuming the row crop percentage is equal to 78.6% in equation 2). Comparing this hypothetical N load to the current condition of 601,129 lbs N (54.5% row crop in 2005) suggests that prairie restoration in Walnut Creek watershed may have reduced nitrogen applications by 30.6% from what applications might have been if the refuge had not been established. While speculative, the analysis nonetheless highlights the substantial effect of the Neal Smith refuge on nitrogen fertilizer loads in the Walnut Creek watershed

Pesticide Application Reductions

Pesticide use in the Walnut Creek watershed was significantly reduced following purchase of refuge-owned lands and adoption of the Cropland Management Plan. In the plan, pre-emergent pesticides are not allowed on refuge-owned lands and post-emergent pesticides must be approved before their use. For the pre-emergent pesticides, including common Iowa herbicides atrazine, cyanazine, metolachlor, alachlor, metribuzin, and acetochlor, this mandate resulted in the complete elimination of pre-emergent pesticide use on refuge lands by 1993. Because these pesticides are typically associated with controlling weeds and grasses in corn crops, pesticide load reductions are closely tied to the amount of corn acres in the refuge.

Estimating reductions in pesticide applications in Walnut Creek watershed suffers from many of the same difficulties that were apparent in the nitrogen reduction estimations.

Baseline land cover conditions have changed in the treatment and control watersheds since 1990. The simplest method to estimate pesticide reductions considers the amount of row crop ground purchased by the Neal Smith refuge in 1992. In 1990, row crops in Walnut Creek watershed comprised 8,946 acres where pesticide applications were made. In 1992, 2,213 acres of former row crop ground were obtained by the USFWS for the Neal Smith refuge. On this land, pre-emergent pesticide use was subsequently eliminated; implying that pesticides were reduced 24.7 percent based on land use data alone. Since 2000, Neal Smith refuge has acquired 262 acres of additional farmland. With these acquisitions, the amount of current and former row crop land under the Cropland Management Plan of the refuge increased to 2,476 acres, or 27.7 percent of the 1990 row crop acreage. Thus in 2005, pesticide applications in Walnut Creek watershed may be considered reduced by nearly 28 percent compared to levels in 1990. This estimated reduction is the same as estimated in previous project reports (Schilling and Thompson, 1999; Schilling et al., 2002).

HYDROLOGY

Precipitation

Annual precipitation at the three USGS gages showed variability over the 10-year monitoring period (Figure 7). Annual precipitation ranged from 25.4 to 41.57 inches at WNT2, 18.6 to 30.2 inches at WNT1 and 14.96 to 35.84 inches at SQW2. Average annual precipitation was highest at WNT2 (29.58 inches) compared to WNT1 (24.61 inches) and SQW2 (27.59 inches). Water Year 1998 was the wettest year recorded during the project, with above-normal annual precipitation at WNT2 and SQW2 relative to the average for central Iowa region of the State Climatologist Office (average of 33.43 inches) (Table 5). The driest year was Water Year 2000 when annual precipitation was more than

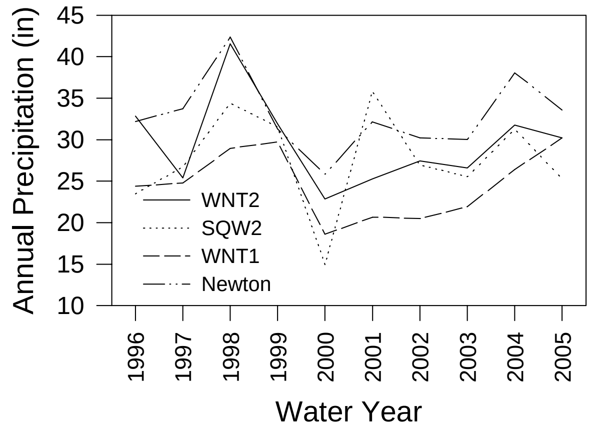


Figure 7. Annual precipitation totals measured at project U.S.G.S. gages and Newton weather station.

10 inches below normal. Overall, annual rainfall totals for the 10-year period of water years 1996 to 2005 were less than the long-term average annual value of 33.43 inches.

It should be noted that precipitation data for WNT1 and WNT2 for water year 2005 was rejected and replaced with data downloaded from a newly operational NOAA weather station located near the Prairie Learning Center at the Neal Smith refuge (http://www.ncdc.noaa.gov/crn/hourly). The USGS rainfall data for WNT1 and WNT2 were anomalously low in Water Year 2005 (16.6 and 12.3, respectively) compared to SQW2 and nearby weather stations. Thus daily rainfall data for both WNT1 and WNT2 sites for Water Year 2005 was assumed to be equal to the precipitation record from the NOAA station, centrally located in the Walnut Creek watershed, with an annual total similar to nearby weather stations (30.2 inches).

Overall, while the patterns of annual precipitation were similar, differences in annual totals were substantial. For example, maximum annual precipitation measured at WNT2 occurred in water year 1998 (41.57 inches) whereas measurements at WNT1 and SQW2 in 1998 were considerably less (24.77 and 26.69 inches, respectively). In contrast, peak annual precipitation at SQW2 was recorded in

Table 5. Summary of annual precipitation totals (in inches) and departure from long-term average for central Iowa (33.43 inches).

Water Year	WNT2	Departure from Avg.	WNT1	Departure from Avg.	SQW2	Departure from Avg.	Newton	Departure from Avg.
1996	32.83	-0.60	24.38	-9.05	23.49	-9.94	32.20	-1.23
1997	25.40	-8.03	24.77	-8.66	26.69	-6.74	33.72	0.29
1998	41.57	8.14	28.95	-4.48	34.40	0.97	42.38	8.95
1999	31.94	-1.49	29.74	-3.69	31.52	-1.91	31.29	-2.14
2000	22.87	-10.56	18.60	-14.83	14.96	-18.47	25.82	-7.61
2001	25.26	-8.17	20.67	-12.76	35.84	2.41	32.17	-1.26
2002	27.43	-6.00	20.48	-12.95	26.96	-6.47	30.21	-3.22
2003	26.59	-6.84	21.91	-11.52	25.51	-7.92	30.02	-3.41
2004	31.75	-1.68	26.39	-7.04	31.29	-2.14	38.05	4.62
2005	30.20	-3.23	30.20	-3.23	25.29	-8.14	33.57	0.14
Average	29.58	-3.85	24.61	-8.82	27.59	-5.84	32.94	-0.49

Water Year 2001 (35.84 inches) but annual totals at WNT2 and WNT1 in 2001 were much less (25.26 and 20.67 inches, respectively). Annual precipitation totals recorded at WNT1 were consistently lower than totals recorded at WNT2.

For comparison purposes, annual precipitation totals from Newton were downloaded from the Iowa Environmental Mesonet (http://mesonet.agron.iastate.edu/request/coop/fe.phtml). The rainfall data from Newton compared favorably with the WNT2 data, especially in water year 1998, and overall trends between WNT2 and Newton were similar (Table 5; Figure 7). However, annual Newton precipitation totals were typically higher than totals from the three project gages.

Patterns in total monthly precipitation were evaluated for the three project gages and Newton (Figure 8). Overall peak monthly measurements often reached or exceeded six inches at most sites, with monthly data from June 1996 exceeding 8 inches at all sites. Monthly precipitation in WNT1 and WNT2 tracked closely together, although like the annual data, WNT1 totals were lower than WNT2 totals. This was particularly evident in data from water year 1998, when monthly totals at WNT1 were more than six inches lower than WNT2 (Figure 8). Less consistency was noted between the monthly records at SQW2 and Newton. While overall patterns follow similar trends, monthly precipitation at SQW2 often trailed behind monthly totals recorded at Newton. A noteworthy exception occurred in Water Year 2001 when the October precipitation total recorded at SQW2 was 8.95 inches and the recorded values at Newton, WNT1 and WNT2 ranged from 0.66 to 0.95 inches. Thus, much of the apparent differences among measurement sites in annual precipitation in Water Year 2001 can be traced to the rather spurious measurements made at SQW2 in October 2001.

To gauge comparability among the precipitation records, correlations among the stations were assessed. Annual precipitation at WNT2 was significantly correlated with WNT1 (correlation coefficient of 0.72; $p = 0.02$) and Newton (0.82, $p = 0.004$) but not correlated with SQW2 (0.49, $p = 0.15$). Annual precipitation at WNT1 was significantly correlated with Newton (0.65, 0.043), but not correlated with SQW2 (0.42, 0.224). However, SQW2 was significantly correlated with Newton (0.68, $p = 0.03$). Correlations of total monthly precipitation records among the sites were similar. With greater degrees of freedom ($n = 120$ months), correlation coefficients and p-values improved (all p-values were <0.001).

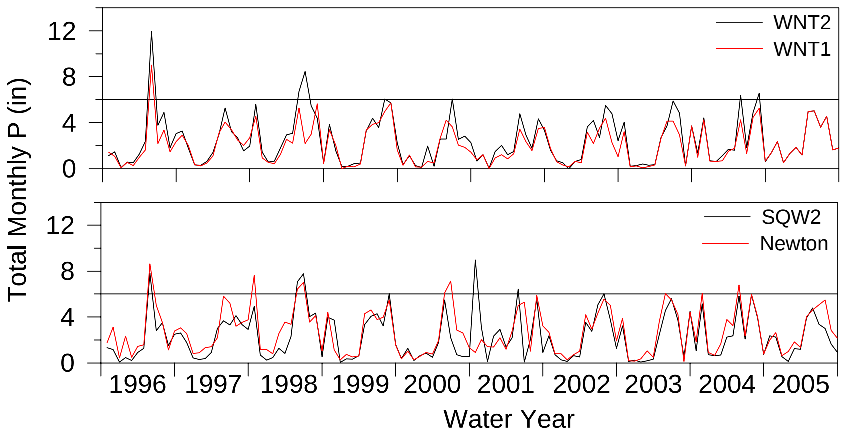


Figure 8. Variations in monthly precipitation.

Monthly precipitation at WNT2 was significantly correlated with WNT1 (0.91), Newton (0.89) and SQW2 (0.79). WNT1 precipitation was correlated with Newton (0.88) and SQW2 (0.79). SQW2 was additionally correlated with Newton (0.79). Overall correlation coefficients of Walnut Creek sites (WNT2 and WNT1) were higher when using Newton data compared to SQW2 data.

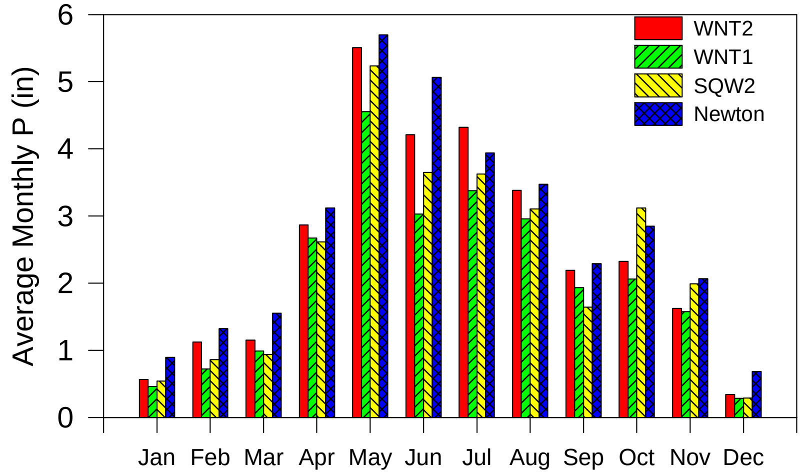
Seasonally, highest monthly precipitation totals occurred in May when total monthly precipitation exceeded five inches (except WNT1) and accounted for about 18 percent of the annual precipitation total (Figure 9). The months of April, July, and August averaged between 2.6 and 5.1 inches and accounted for between 10 and 15 percent of the annual total.

Discharge

Stream discharge varied considerably from year to year (Figure 10) with annual discharge varying more than four-fold between water years 1996 to 2005 (Table 6). Annual discharge in Walnut Creek (WNT2) ranged from 4.31 inches in Water Year 2002 to 16.61 inches in Water Year 1998, whereas discharge in Squaw Creek SQW2) varied from 3.36 to 16.91 inches in the same two years. Average annual discharge for WNT2, SQW2 and WNT1 was similar for all gage sites, ranging between 8.62 to 8.93 inches (Table 6). Annual patterns of discharge tracked similarly in Walnut and Squaw creek watersheds with exception for maximum daily discharge (Figure 11). Maximum daily discharge varied among sites and water years, with peak discharge at Walnut Creek ranging from 125 to 573 cfs in water years 2002 and 1996, respectively. In Squaw Creek, maximum daily discharge from several water years exceeded the maximum from Walnut Creek, with annual maximum values ranging between 122 and 847 cfs (Table 6). Maximum daily discharge values in Squaw Creek recorded in water years 1998 and 2000 were 326 and 386 cfs higher than maximum daily discharge in Walnut Creek in the same years (Figure 11).

Within any given year, the timing of peak monthly discharge was often similar but there were differences in peak magnitude (Figure 12). Total monthly discharge exceeded 1000 cfs on 10 occasions at WNT2 and 8 occasions at SQW2. Greatest total monthly discharge occurred in May 1998 at both gage sites when discharge exceeded 2500 cfs. Peak monthly discharge was less than 500 cfs in Water Year 2002 (Figure 12). Many water years showed evidence for two discharge peaks per water year, with one peak indicative of snow melt in February or March and a second peak occurring in May or June of each year. In years without a late winter or early spring peak, snowmelt probably occurred slowly or little snow pack was present.

Figure 9. Average monthly precipitation during the project.



Annual discharge followed precipitation patterns with greater precipitation resulting in greater streamflow (Figure 13). The relation of discharge at WNT2 to precipitation measured at WNT2 was highly significant (r2 = 0.81, p = 0.00). In contrast, the relation of annual precipitation (P) measured at SQW2 to discharge at SQW2 was not significant (r2 = 0.23, p >0.1). The relation of discharge at SQW2 with precipitation measured at Newton was considerably better (r2 = 0.74) and suggests that precipitation records from SQW2 are suspect. From this evaluation, it is recommended that consideration of precipitation in Squaw Creek should utilize records from Newton as the source of precipitation data rather than values obtained from the SQW2 gage.

Discharge in both watersheds was very flashy responding rapidly to precipitation events (Figure 14). For example, stream stage at WNT2 responded rapidly during a series of rainfall events in Water Year 2003. A total of 15.3 inches of precipitation occurred throughout a monitoring period from March 25 to July 1, 2003. Daily precipitation exceeded one inch four times (May 4, May 7, May 9 and June 26), with 5.8 inches of rainfall occurring over a 10-day period from April 30-May 9. The largest single rainfall event (1.6 inches) occurred on June 26. The WNT2 stream hydrograph during the study period was dominated by a series of sharp runoff peaks (Figure 14). The 10-day period of prolonged rainfall in late April/early May produced a series of ever-increasing runoff peaks, with each major rainfall event resulting in a pronounced rise in stream stage. Increasing soil moisture levels in the watershed during this rainy period are evidenced by runoff (Q) comprising 21.5% of the P during the April 30 event, and increasing to 43.7%, 58.7% and 72.5% of P during the next three successive rainfall events. Following a dry two-week period from June 10-24, watershed soil moisture levels evidently decreased since Q was only 18.4% of P during the large rainfall event on June 26. The rapidity of stream stage rise is shown by the runoff event occurring during the night of May 8-9, 2003 when stream stage increased from 793.3 feet to nearly 800 feet during an eight-hour period (Figure 15). Walnut Creek stage reached a maximum elevation of 799.8 feet, which is approximately 0.6 feet below the top of the streambank.

Annual baseflow in Squaw Creek was higher than Walnut Creek with baseflow ranging from 2.23 to 11.79 in Squaw Creek and 2.55 to 9.69 inches in Walnut Creek (Table 6). Average annual baseflow was highest at WNT1 (5.65 inches) compared to WNT2 (5.07) and SQW2 (5.35 inches). The fraction of total discharge as baseflow (baseflow percentage) was also highest at upstream Walnut Creek (WNT1) with an average annual baseflow of

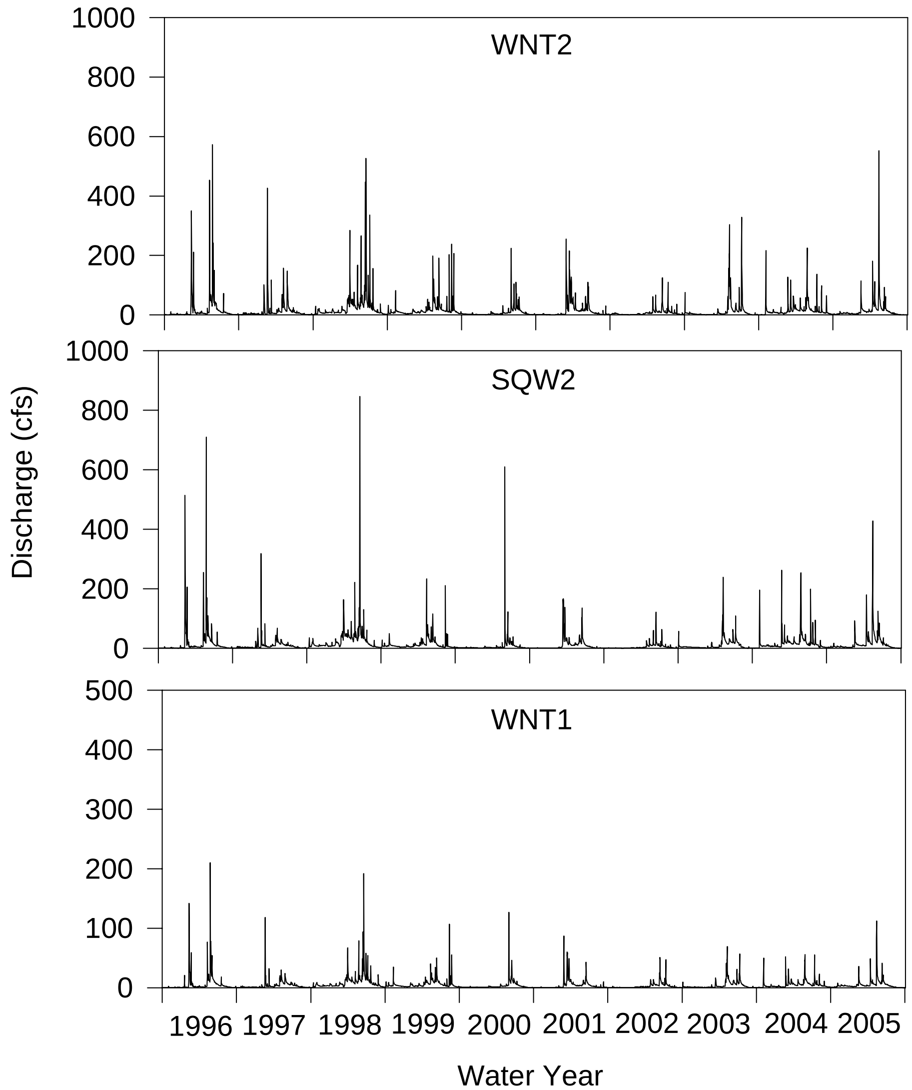


Figure 10. Daily discharge at WNT2, SQW2 and WNT1 at U.S.G.S. gaging stations.

64.7 percent. Annual baseflow at downstream Walnut Creek and Squaw Creek were less (57.3 and 62.4 percent, respectively). Annual baseflow fraction followed similar patterns with exception of Water Year 2000, when baseflow at SQW2 was 54.3 percent compared to baseflow at WNT2 and WNT1 of 69.0 and 70.7 percent, respectively. This likely reflects the greater contribution of stormflow in Water Year 2000 at Squaw Creek when a large peak discharge event occurred in Squaw Creek and not in Walnut Creek.

Subtracting the contribution of discharge and baseflow at WNT1 from WNT2 provides an indication of the streamflow contribution from lower Walnut Creek containing most of the prairie restoration. Baseflow discharge was nearly four inches higher in the upper portion of the watershed (WNT1; 8.75 inches) compared to the lower portion (WNT2-WNT1; 4.78 inches) and the percentage of streamflow as baseflow was much less in lower Walnut Creek (Table 6). On the other hand, total discharge was slightly higher in lower Walnut

Table 6. Summary of annual discharge and baseflow values measured at WNT2, SQW2, WNT1 and WNT2-1 (WNT2-1 estimated by subtracting WNT1 from WNT2).

Water Year	Discharge (cfs)	Baseflow (cfs)	Discharge (in)	Baseflow (in)	Baseflow Percentage	Mean Annual Discharge (cfs)	Median Annual Discharge (cfs)	Maximum Daily Discharge (cfs)
WNT2								
1996	6400	2847	10.60	4.71	44.5%	17.5	4.2	573
1997	4487	2064	7.43	3.42	46.0%	12.3	4.6	426
1998	10027	5851	16.61	9.69	58.4%	27.5	12.0	526
1999	6260	3916	10.37	6.48	62.5%	17.2	11.0	238
2000	2795	1930	4.63	3.20	69.0%	7.6	3.2	224
2001	5381	3231	8.91	5.35	60.0%	14.7	3.3	255
2002	2601	1539	4.31	2.55	59.2%	7.1	4.4	125
2003	4892	2862	8.10	4.74	58.5%	13.4	3.7	329
2004	5897	3540	9.77	5.86	60.0%	16.1	9.9	225
2005	5167	2825	8.56	4.68	54.7%	14.2	5.9	552
Average	5391	3060	8.93	5.07	57.3%	14.8	6.2	347
SQW2								
1996	5945	2611	10.84	4.76	43.9%	16.2	3.8	710
1997	3197	2054	5.83	3.74	64.3%	8.8	4.1	318
1998	9279	6468	16.91	11.79	69.7%	25.4	13.0	847
1999	4781	3262	8.71	5.95	68.2%	13.1	7.8	233
2000	2517	1368	4.59	2.49	54.3%	6.9	2.4	610
2001	3878	2790	7.07	5.08	71.9%	10.6	1.6	167
2002	1843	1222	3.36	2.23	66.3%	5.0	1.9	122
2003	3808	2723	6.94	4.96	71.5%	10.4	3.8	239
2004	6466	3807	11.79	6.94	58.9%	17.7	11.0	262
2005	5569	3032	10.15	5.53	54.4%	15.3	6.8	428
Average	4728	2934	8.62	5.35	62.4%	12.9	5.6	394
WNT1								
1996	2090	1051	10.35	5.20	50.3%	5.7	1.2	210
1997	1299	790	6.43	3.91	60.8%	3.6	1.4	118
1998	3374	2252	16.70	11.15	66.8%	9.2	5.3	192
1999	2168	1474	10.73	7.30	68.0%	5.9	3.5	107
2000	1067	755	5.28	3.74	70.7%	2.9	1.1	127
2001	1558	1106	7.71	5.47	70.9%	4.3	0.9	87
2002	977	631	4.84	3.13	64.6%	2.7	1.1	51
2003	1478	1011	7.32	5.01	68.4%	4.1	1.2	69
2004	1870	1267	9.26	6.27	67.7%	5.1	3.3	55
2005	1802	1067	8.92	5.28	59.2%	4.9	3.0	112
Average	1768	1140	8.75	5.65	64.7%	4.8	2.2	113
WNT2-WNT1								
1996	4310	1796	10.73	4.47	41.7%			
1997	3188	1274	7.93	3.17	40.0%			
1998	6653	3599	16.56	8.96	54.1%			
1999	4092	2442	10.18	6.08	59.7%			
2000	1727	1175	4.30	2.92	68.0%			
2001	3823	2126	9.51	5.29	55.6%			
2002	1624	907	4.04	2.26	55.9%			
2003	3414	1851	8.50	4.61	54.2%			
2004	4028	2273	10.02	5.66	56.4%			
2005	3365	1759	8.37	4.38	52.3%			
Average	3622	1920	9.01	4.78	53.8%			

Creek watershed compared to the upper portion, probably due to a greater proportion of the land in hillslope in the southern two-thirds of the watershed.

Seasonally, greatest monthly discharge occurred in May each year when more than two inches of streamflow occurred (Figure 16). Discharge in June typically exceeded one inch

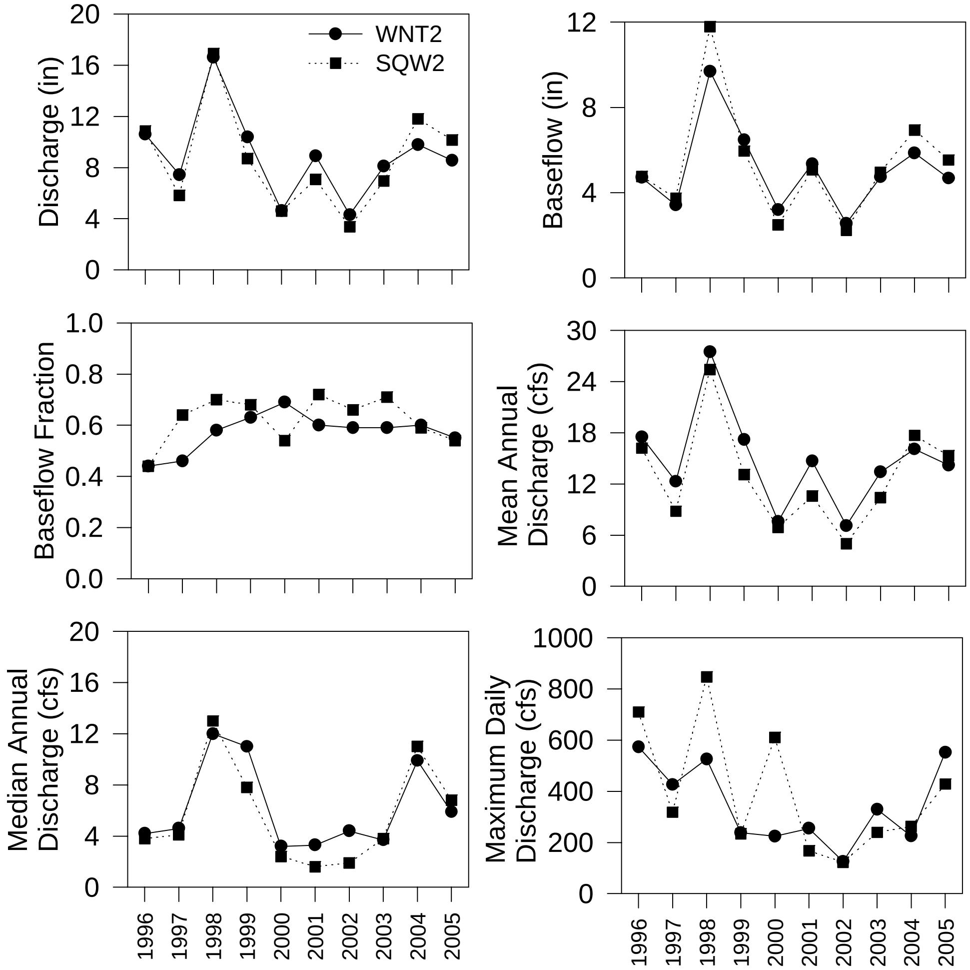


Figure 11. Comparison of annual stream discharge characteristics at WNT2 and SQW2.

and together, May and June accounted for more than 45 percent of annual streamflow (Figure 17). The months of February, March, April and July accounted for similar percentages of total streamflow, averaging near 10 percent for each month (Figure 17). Little streamflow typically occurred in the remaining months of the year (August through January) accounting for less than five percent of annual streamflow by month.

The pattern of monthly discharge is in contrast to monthly precipitation patterns that showed precipitation increasing from January to May and months of April, July and August accounting for 10-15 percent of annual totals (see Figure 9). Precipitation occurring in August to October each year did not typically result in increased discharge in these months. The relation of rainfall to runoff in Walnut and Squaw creek watersheds was typically highest in September (Figure 18) following the summer growing season when soil moisture levels in the watersheds were depleted. The rainfall-runoff relation was above 10 in August through October and typically less than 5 in the remaining months of the year.

The pattern of total monthly streamflow and baseflow shown by Figure 19 suggests greater contribution of June streamflow at SQW2 and WNT1 compared to WNT2. While total

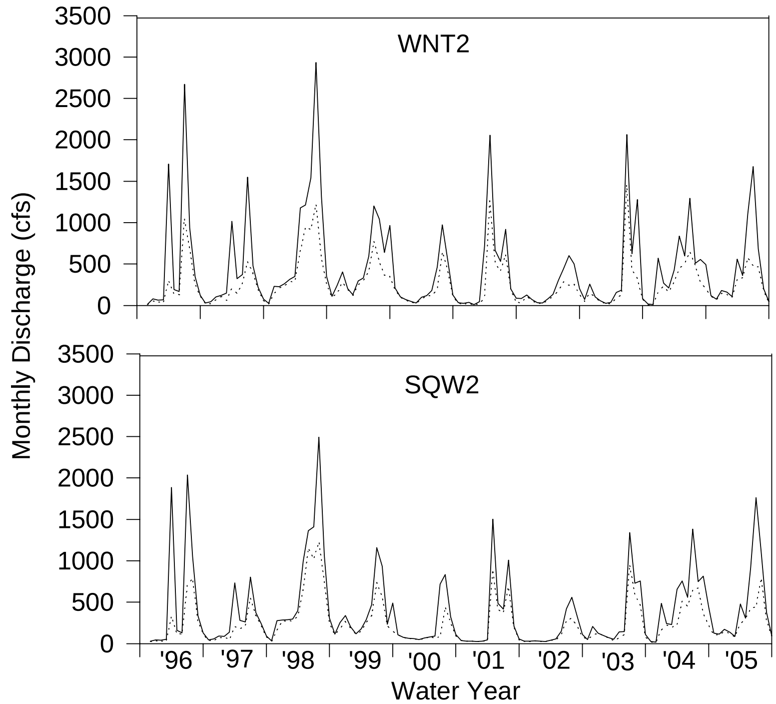


Figure 12. Time series of total monthly discharge at WNT2 and SQW2. (Solid line = total streamflow; dashed line = baseflow).

monthly streamflow at WNT2 rapidly decreases from May to June, a gradual decrease was noted at WNT1 and SQW2. Similarly, baseflow at WNT2 decreased from May to June, but baseflow at WNT1 and SQW2 increased from May to June. It is hypothesized that the increased baseflow was responsible for the gradual decrease in streamflow for the months of May and June and that this may be a signal that tile drainage from the highly agricultural areas was maintaining higher baseflow levels in the streams. In lower Walnut Creek (WNT2-1) where many tile have been removed, total baseflow rapidly decreased from May to June (Figure 19). A slight increase in streamflow and baseflow was observed from September to November at all gage sites.

Baseflow fraction was lowest in February and May each year and highest in late fall through January when baseflow percentage was near 80 percent (Figure 20). February is typified by greater runoff derived from snowmelt whereas greatest monthly rainfall occurs in May. Although the pattern of monthly baseflow was similar to discharge, the monthly fraction of annual baseflow increased steadily from January to May (Figure 20). Baseflow occurring in May and June accounted for approximately 40 percent of the annual baseflow total.

Trends

The daily discharge and baseflow values were highly skewed and were log10 transformed prior to regression analyses. Daily discharge and baseflow values were also highly autocorrelated, with autocorrelation coefficients ranging from 0.79 to 0.98. The autocorrelation structure of residuals from ordinary least squares regressions was examined. The Durbin-Watson statistic on the residuals without correcting for autocorrelation ranged from 0.03 to 0.43 (extremely low and indicate

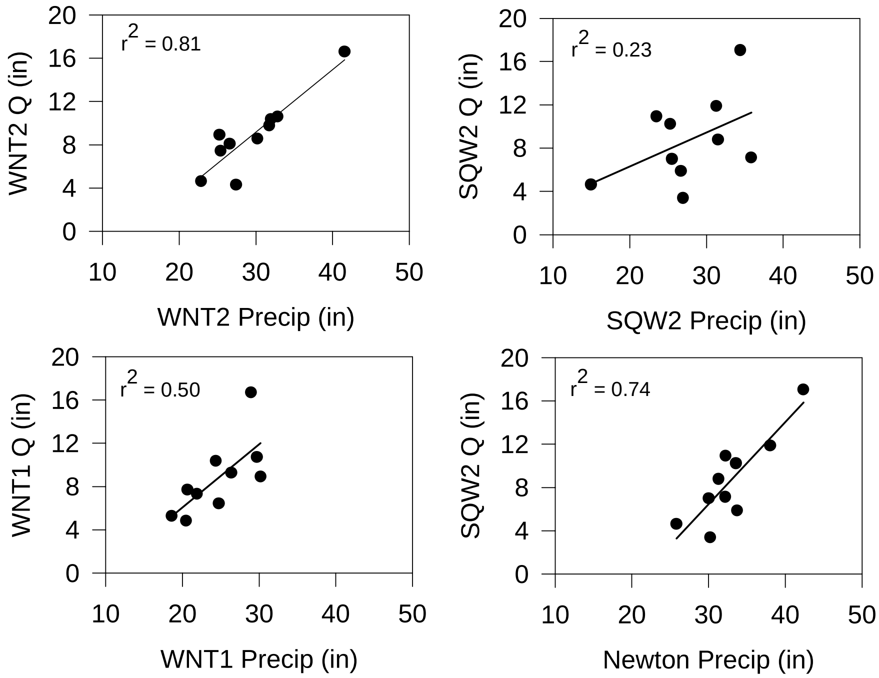


Figure 13. Relation of annual discharge at gaging sites to annual precipitation.

autocorrelation of residuals). When the regression included autocorrelation lag 1, the DW statistic ranged from 0.7 to 1.93 (values closer to 2.0). This indicates independence of the residuals and the appropriate assumption of a AR(1) time series model.

A ‘date’ variable was included to test for linear trends over the 10-year project monitoring period. The Yule Walker regression estimates were utilized that incorporated the time series error model. From preliminary data exploration, there was evidence that the regression slopes (or changes over time) were not the same for each month. Therefore, a model was utilized that included interaction terms to allow for each month to have its own intercept and slope. The significance of trends of December values was used as the benchmark to determine trend significance (since indicator variables were utilized for January through November). The significance of the interaction term indicated if it was statistically different from December.

Discharge and baseflow values were estimated for 1995 as well as 2005, using the least square means of the model covariates at each month. The estimated log values were transformed to the original scale and the percent change calculated (a negative value indicates a decrease).

Not unexpectedly, all discharge and baseflow covariates were highly correlated and statistically significant (at $P<.0001$ level). Downstream Walnut Creek (WNT2) was highly correlated with WNT2 baseflow (0.93), upstream discharge and baseflow at WNT1 (0.95 and 0.87, respectively) and discharge and baseflow at SQW2 (0.92 and 0.86, respectively). Likewise, WNT2 baseflow was highly correlated with upstream baseflow at WNT1 (0.94) and baseflow at SQW2 (0.92).

Trend tests were conducted for mean daily discharge and baseflow at WNT2 using either no covariates (simple change over time) or WNT1 discharge or baseflow as a covariate. All months were allowed to have a different slope and intercept, and all tests were adjusted for autocorrelation. Trend tests for discharge and baseflow at WNT1 and SQW2 were simply adjusted for autocorrelation and evaluated over time.

With no adjustment for upstream discharge, results indicate no statistically significant trends

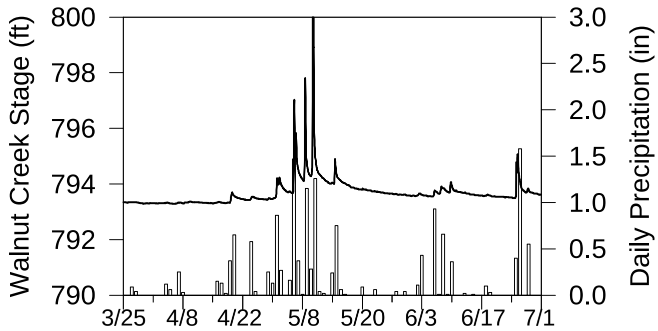


Figure 14. Response of Walnut Creek stage to precipitation in Water Year 2003.

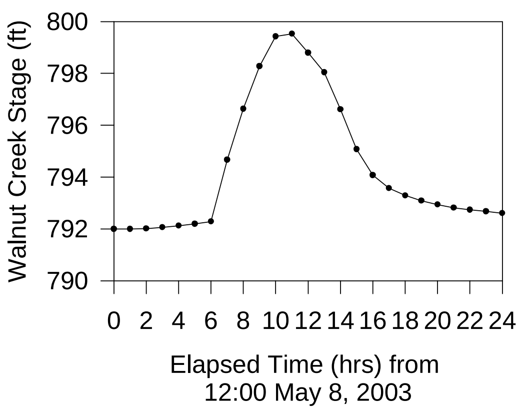


Figure 15. Hydrograph of Walnut Creek discharge measured on May 8, 2003.

in monthly discharge at WNT2 from 1995 to 2005 and a minor decrease in February and March baseflow (Table 7). Although not statistically significant, discharge decreased during the June through September period (p-values ranged from 0.41 to 0.25), and baseflow decreased at WNT2 during all months. With adjustment made for upstream daily discharge at WNT1, trends in daily discharge at WNT2 were mixed. Some months showed a non-significant decreasing trend, whereas the months of August through October showed an increasing discharge trend ranging from 14 to 58 percent (Table 7). A decreasing trend in December discharge was noted (30 percent; p = 0.03). A similar 17 to 56 percent increasing trend in WNT2 baseflow during the months of August through November was observed when adjustments were made for WNT1.

Trends in discharge and baseflow were also evaluated for upstream WNT1 and control watershed SQW2 (Table 8). Discharge at WNT1 decreased approximately 50 percent over the 10 year monitoring period for the months of June, July, August and September. Upstream baseflow also decreased in most months, with the decrease during the July through October period being statistically significant. In Squaw Creek, there was no statistically significant change in mean daily discharge at downstream SQW2 and marginal indication of increasing trends in baseflow for February, March and November (Table 8).

Discussion

Discharge in both Walnut and Squaw creek watersheds was highly variable over the 10-year monitoring period, varying by day, month and year. Streamflow in both watersheds was controlled in large part by seasonal precipitation

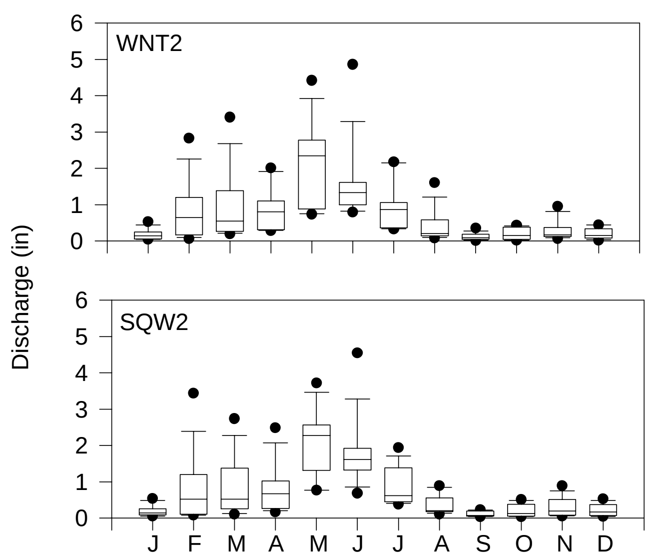


Figure 16. Box plots of total discharge by month at WNT2 and SQW2 gaging stations. Box plots illustrate the 25th, 50th and 75th percentiles; the whiskers indicate the 10th and 90th percentiles; and the circles represent data outliers.

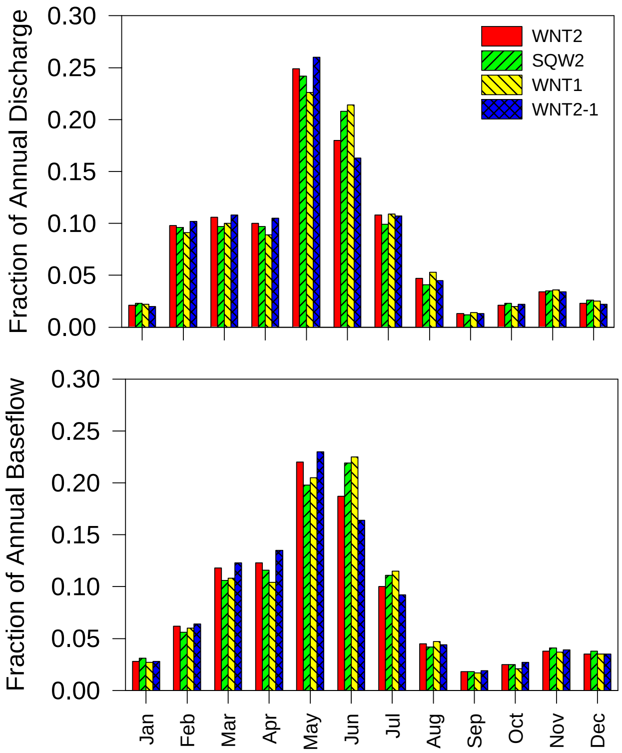


Figure 17. Fraction of annual discharge and baseflow at WNT2 and SQW2 by month.

patterns and soil moisture conditions, with greatest streamflow typically occurring during rainy periods when antecedent soil moisture conditions are high. During most years, this period included May and June when nearly one-half of the annual total streamflow typically occurred. Streamflow events were often characterized by flashy conditions typical of flow in incised channels. Stream stage rapidly increased in response to precipitation, and in one example from Walnut Creek, stream stage rose and fell nearly eight feet in the span of 12 hours (Figure 15). High flow events were occasionally observed in February and March due to snowmelt, whereas during the remainder of the year, streamflow was dominated by baseflow.

There was evidence to suggest that streamflow patterns were different among the watershed areas monitored during the project. Maximum daily discharge was often higher in Squaw Creek than Walnut Creek, exceeding 600 cfs on three occasions in Squaw Creek compared to all maximum events less than 573 cfs in Walnut Creek. This suggests that storm events may result in flashier conditions in Squaw Creek. Considering the high percentage of row crop land in the Squaw Creek watershed compared to the restored prairie areas in Walnut Creek, flashier discharge conditions might be expected. Additionally, total annual streamflow was higher in Walnut Creek, but Squaw Creek had higher annual baseflow and a higher percentage of streamflow as baseflow. Highest annual baseflow and baseflow percentage was associated with upstream Walnut Creek (WNT1), whereas lower baseflow contributions were estimated for lower Walnut Creek watershed (WNT2-1). The similar nature of elevated baseflow in the highly

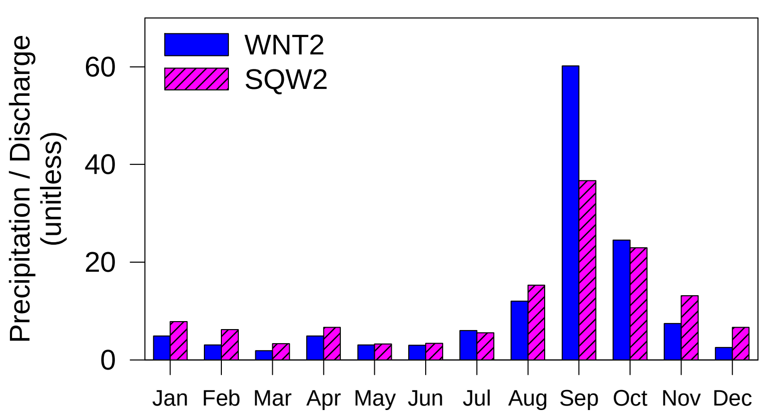


Figure 18. Ratio of precipitation to discharge at WNT2 and SQW2 by month.

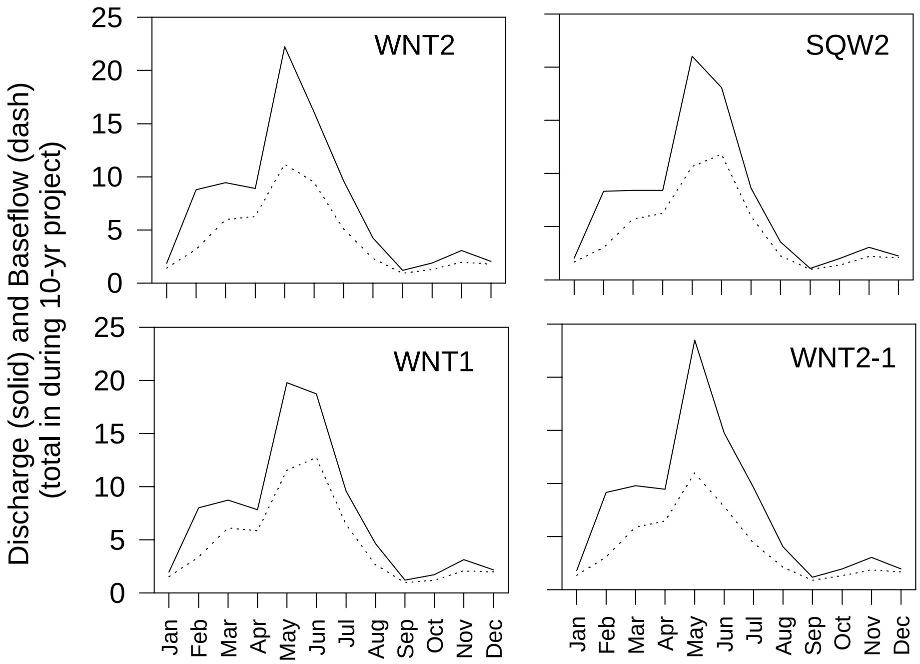


Figure 19. Average discharge and baseflow at gaging sites by month.

row-cropped regions of Squaw Creek and upstream Walnut Creek suggest that tile drainage may be contributing to apparent baseflow in streams. Tile drainage contributions may be responsible for elevated baseflow measured in June at WNT1 and SQW2. In contrast, lower baseflow in lower Walnut Creek is consistent with the hypothesized effects of prairie restoration to reduce drainage of water

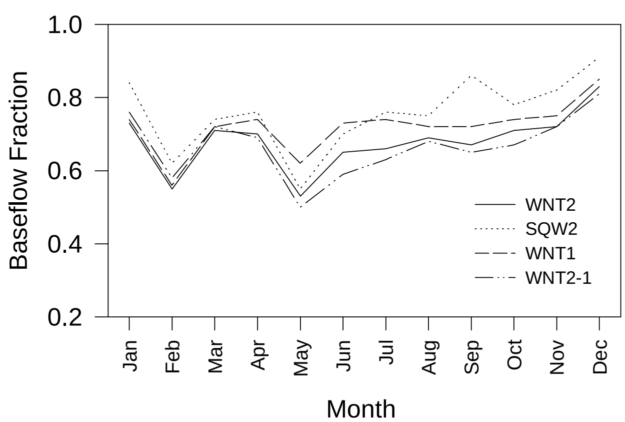


Figure 20. Fraction of total monthly discharge as baseflow.

from the soil profile and the removal of drainage tiles by the refuge.

Despite differences in discharge and baseflow patterns among the watershed areas, there were few consistent patterns evident in the statistical trends over time. Although the trend in daily discharge and baseflow over time at WNT2 was not significant, the trends were nonetheless decreasing, which suggests less water draining off the Walnut Creek watershed over time. However, downstream discharge at WNT2 was highly correlated with upstream discharge, so that by adjusting for upstream discharge, trends at WNT2 were essentially reversed. Discharge and baseflow appeared to have increased at WNT2 over time if upstream flow from WNT1 is considered in the regression model. Considering that flow and baseflow at WNT1 showed statistically significant decreases over time, it would appear that much of the apparent decrease in discharge and baseflow at WNT2 was due to changes occurring above WNT1. Reasons for reduced discharge and baseflow in the area above WNT1 are not known conclusively at this time, but it is possible that exceedingly dry

Table 7. Trend test for mean daily discharge at WNT2. Each month was allowed a different slope and intercept. Trends were adjusted for autocorrelation and covariates (as indicated).

WNT2 Covariates: None

	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate	Sig.
Discharge	January	4	3.4	-14	0.7855	ns
(cfs)	February	7.5	5.8	-23	0.6787	ns
	March	11.1	7	-37	0.4804	ns
	April	10.5	9.5	-9	0.9139	ns
	May	12.9	12.1	-7	0.9455	ns
	June	17.5	8	-54	0.2475	ns
	July	11.4	5.6	-51	0.2963	ns
	August	5.4	3.1	-43	0.4104	ns
	September	3.7	1.9	-50	0.2842	ns
	October	2.3	2.4	4	0.9065	ns
	November	3.2	3.6	14	0.7234	ns
	December	3.3	3.2	-3	0.9563	ns
Baseflow	January	5.1	2.4	-52	0.3183	ns
(cfs)	February	4.2	3.4	-19	0.0269	*
	March	4.3	3.6	-16	0.0373	*
	April	4.4	3.5	-21	0.1043	ns
	May	5	3.5	-31	0.3243	ns
	June	5.1	3.4	-33	0.3978	ns
	July	4.8	3.1	-36	0.5015	ns
	August	4.7	2.9	-38	0.6104	ns
	September	4.4	2.5	-44	0.9185	ns
	October	3.9	2.7	-32	0.2176	ns
	November	4.2	2.8	-33	0.1365	ns
	December	4.8	2.6	-45	0.1363	ns

WNT2 Covariates: Log upstream discharge (WNT1)

	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate	Sig.
Discharge	January	3.1	1.7	-46	0.1817	ns
(cfs)	February	6.2	5.9	-6	0.1947	ns
	March	9.8	8.8	-10	0.2822	ns
	April	11.2	11.4	2	0.1107	ns
	May	24.8	23.2	-6	0.2194	ns
	June	26	22.2	-15	0.4008	ns
	July	12.7	11.4	-10	0.2958	ns
	August	3.8	4.4	14	0.0384	*
	September	1.1	1.7	58	0.0006	**
	October	1.6	1.8	16	0.0253	*
	November	3.7	3	-19	0.4597	ns
	December	3.1	2.2	-30	0.0325	*
Baseflow	January	2.2	1.9	-16	0.369	ns
(cfs)	February	4.2	4.2	-1	0.487	ns
	March	6.6	6.7	3	0.391	ns
	April	7.7	8.2	7	0.2647	ns
	May	13.4	13.5	1	0.5056	ns
	June	15.6	15.7	1	0.5101	ns
	July	8.3	8.9	7	0.2823	ns
	August	3.2	3.9	24	0.0365	*
	September	1	1.6	56	0.0001	**
	October	1.3	1.9	47	0.0001	**
	November	2.4	2.8	17	0.0104	*
	December	2.4	2.2	-9	0.4095	ns

ns = not significant
+ = significant at P<0.1
* = significant at P<0.05
** = significant at P<0.01

Table 8. Trend test for mean daily discharge at WNT1 and SQW2. Each month was allowed a different slope and intercept. Trends were adjusted for autocorrelation and covariates (as indicated).

WNT1 Covariates: None	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate	Sig.
Discharge	January	0.9	1.6	67	0.615	ns
(cfs)	February	2.7	1.8	-34	0.1497	ns
	March	3.7	2.5	-33	0.1939	ns
	April	3.6	3.5	-3	0.5564	ns
	May	4.7	4.6	-2	0.5661	ns
	June	6.5	3.3	-49	0.0926	*
	July	4.3	2.3	-46	0.1073	~ns
	August	2.3	1.1	-51	0.0709	+
	September	1.5	0.5	-65	0.015	*
	October	0.8	0.8	9	0.6553	ns
	November	1	1.4	33	0.9605	ns
	December	0.9	1.2	36	0.4618	ns
Baseflow	January	1.3	1	-25	0.6781	ns
(cfs)	February	1.3	1.5	11	0.1118	ns
	March	1.7	1.9	12	0.1421	ns
	April	2.2	2	-8	0.5798	ns
	May	3.1	2.1	-30	0.59	ns
	June	3.3	1.9	-43	0.1865	ns
	July	2.9	1.3	-55	0.0244	*
	August	2.3	0.7	-68	0.0003	**
	September	1.8	0.4	-78	<.0001	**
	October	1.1	0.6	-49	0.032	*
	November	1.1	0.8	-28	0.5433	ns
	December	1.1	0.9	-20	0.2525	ns

SQW2 Covariates: None	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate	Sig.
Discharge	January	3	4.2	41	0.918	ns
(cfs)	February	6.7	4.6	-32	0.1103	ns
	March	9	6.7	-26	0.1998	ns
	April	8.7	9.2	6	0.5538	ns
	May	10.7	12.4	16	0.6721	ns
	June	12.8	9.1	-29	0.2033	ns
	July	7.9	6.5	-17	0.3109	ns
	August	4.5	4.3	-4	0.4417	ns
	September	3.4	2.9	-14	0.3264	ns
	October	2.4	3.3	40	0.9243	ns
	November	2.7	4	50	0.9573	ns
	December	2.7	4	47	0.3546	ns
Baseflow	January	4.5	3.8	-15	0.4861	ns
(cfs)	February	3.6	4.4	20	0.0462	*
	March	3.6	4.4	22	0.07	+
	April	3.8	4.4	15	0.183	ns
	May	4.2	4.3	4	0.4672	ns
	June	4.3	4.4	3	0.5123	ns
	July	4.2	4.1	-3	0.7586	ns
	August	4.1	3.8	-9	0.97	ns
	September	3.9	3.7	-4	0.7703	ns
	October	3.7	3.9	6	0.288	ns
	November	3.8	4.1	9	0.0784	+
	December	4.2	3.9	-8	0.8016	ns

ns = not significant
+ = significant at P<0.1
* = significant at P<0.05
** = significant at P<0.01

Table 9. Summary of annual sediment loads and concentrations at WNT2, SQW2 and WNT1.

Water Year	Annual Sediment Load (tons)	Daily Mean Sediment Load (tons)	Maximum Daily Sed. Load (tons)	Sediment Export (tons/acre)	Annual Mean Sediment Conc. (mg/l)	Maximum Sediment Conc. (mg/l)	Annual Median Sed. Conc. (mg/l)
WNT2							
1996	14305	39.1	3980	1.11	127.8	2800	54.0
1997	9399	25.7	1713	0.73	127.0	3020	59.5
1998	18367	50.3	4600	1.42	153.1	3120	58.0
1999	8782	24.1	1989	0.68	103.3	2632	33.5
2000	2903	7.9	1380	0.23	82.7	1570	44.5
2001	6357	17.4	1290	0.49	89.7	1950	44.0
2002	3706	10.2	583	0.29	88.6	1430	31.0
2003	8889	24.4	1530	0.69	99.3	1560	48.0
2004	4329	11.8	606	0.34	78.3	1400	35.0
2005	6803	18.6	2740	0.53	91.8	2620	53.0
Average	8384	23.0	2041	0.65	104.1	2210	46.0
SQW2							
1996	14898	40.7	6880	1.27	93.5	3270	22.0
1997	5001	13.7	1657	0.43	106.2	2648	43.2
1998	20456	56.0	11400	1.75	119.7	3250	56.0
1999	8006	21.9	2293	0.68	128.6	2822	59.6
2000	9361	25.6	7540	0.80	69.5	2520	39.0
2001	5942	16.3	775	0.51	92.8	1420	48.0
2002	893	2.4	189	0.08	47.5	507	29.0
2003	4572	12.5	1620	0.39	78.2	1590	44.0
2004	5641	15.4	1100	0.48	84.9	2660	36.0
2005	5669	15.5	1910	0.48	80.3	1180	50.0
Average	8044	22.0	3536	0.69	90.1	2187	42.7
WNT1							
1996	2534	6.9	1080	0.59	99.6	1890	53.5
1997	1961	5.4	874	0.45	118.6	1928	66.5
1998	2757	7.6	654	0.64	127.3	2130	80.0
1999	1688	4.6	334	0.39	83.2	1256	50.5
2000	678	1.9	459	0.16	49.0	949	33.0
2001	916	2.5	140	0.21	61.7	736	42.0
2002	249	0.7	47	0.06	34.2	339	24.0
2003	533	1.5	111	0.12	41.9	480	29.0
2004	476	1.3	66	0.11	41.1	467	32.5
2005	810	2.2	229	0.19	54.8	699	43.0
Average	1260	3.4	399	0.29	71.2	1087	45.4

conditions in 2005 may have contributed to the decreasing trend. For the first time during the 10-year project, flow at WNT1 dried up, as there was essentially no streamflow measured at WNT1 during the month of September 2005. No trends in discharge over time were evident in Squaw Creek, but increasing baseflow for February, March and November may imply increased drainage from converted grasslands to row crop in the watershed. However, this pattern was not evident in upper Walnut Creek watershed despite an eight percent increase in row crop occurring in this area.

Overall, it is suspected that detecting trends over time in discharge and baseflow requires a much greater timeframe than this 10-year study. The variability in precipitation and discharge across the seasons and years make detecting changes from land conservation difficult without sufficient time to account for

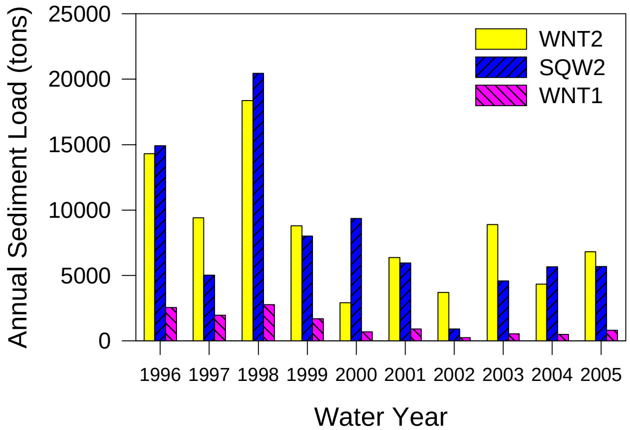


Figure 21. Total annual sediment load measured at USGS gaging sites.

changing climate conditions. Elsewhere, trends in baseflow patterns across Iowa were noted over a 60-year period (Schilling and Libra, 2003), and more than 30 years was needed to detect changes in runoff and baseflow in much smaller western Iowa catchments from improved conservation (Kramer et al., 1999). While detecting gradual changes in streamflow over time may not be possible in a limited timeframe, perhaps area- or time-weighted methods may be appropriate. In this case, less baseflow appears to be emanating from lower Walnut Creek compared to upper Walnut Creek and Squaw Creek.

SUSPENDED SEDIMENT

Suspended Sediment Loads

Suspended sediment concentrations and loads varied widely during the 10-year monitoring period (Table 9). Total annual sediment export ranged from 3,706 to 18,367 tons in Walnut Creek and from 893 to 20,456 tons in Squaw Creek, with higher average annual loss higher in Walnut Creek (8,384 tons) than Squaw Creek (8,044 tons). Greatest annual loss of sediment in both Walnut and Squaw creek watersheds occurred in water years 1998 and 1996 when annual sediment export exceeded 10,000 tons (Figure 21). The least amount of annual sediment export occurred in Walnut Creek in Water Year 2000 whereas annual sediment losses were lowest in Squaw Creek in Water Year 2002.

Maximum daily sediment loads exceeded 6,000 tons on three occasions in Squaw Creek, whereas the maximum daily load in Walnut Creek peaked greater than 3,000 tons on two occasions (Figure 22). The maximum daily sediment load measured in either watershed occurred in Squaw Creek in Water Year 1998 (June 18, 1998) when 11,400 tons of sediment was lost in a single day. The maximum discharge recorded in either watershed occurred on this day as well (847 cfs at SQW2). The maximum daily sediment load in Walnut Creek was measured on March 30, 1998 (4,600 tons). Sediment losses generally followed discharge patterns (Figure 23), but exhibited more episodic behavior with cumulative losses showing a pronounced annual stepped pattern. Sediment losses were concentrated during brief events followed by long periods of little sediment transport. On the other hand, cumulative discharge was more gradual in nature (Figure 23).

Sediment transport through Walnut and Squaw creek watersheds was very flashy, evidenced by most of the annual suspended sediment load occurring during intermittent high flow events. In Walnut Creek watershed, the maximum one-day total of sediment load averaged 24.9 percent and ranged from 15.7 to 47.5 percent of the annual sediment load (Table 10). In Squaw Creek, the one-day total was a higher proportion of the annual total, averaging 36.7 percent and ranging from 13.0 to 80.5 percent. In Water Year 2000, a single event on May 31 delivered 47.5, 80.5 and 67.7 percent of the annual sediment load for the year at WNT2, SQW2 and WNT1, respectively. Approximately two-thirds of the annual suspended sediment load was exported from the watersheds in five-days, whereas nearly 90 percent of the annual sediment load was lost from the watersheds in 20 days (Table 10). In many water years, much of the

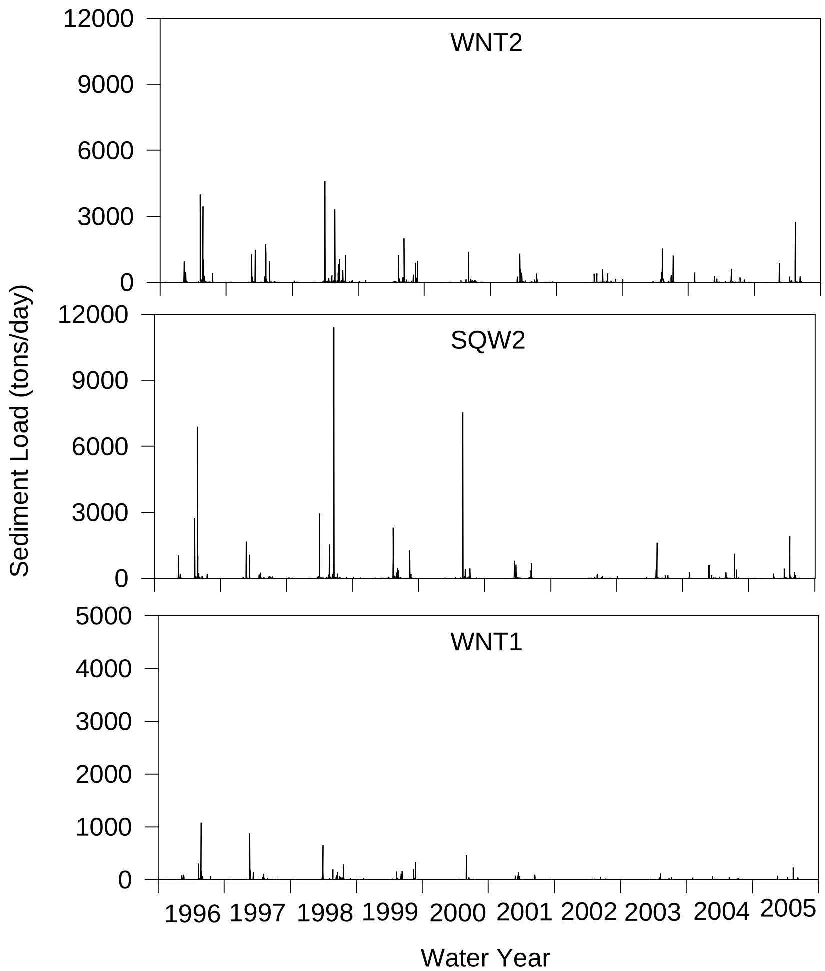


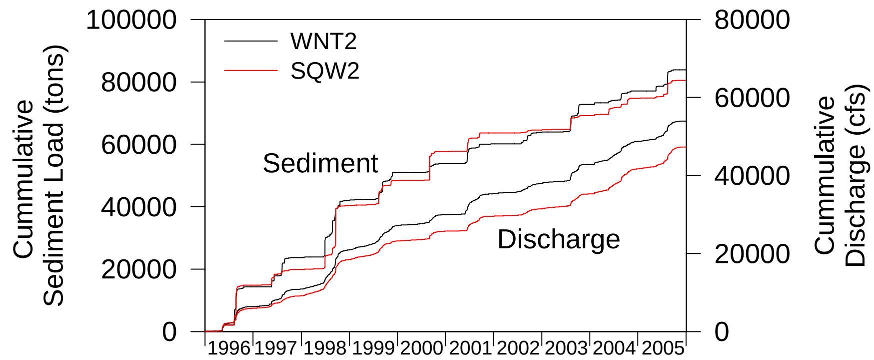
Figure 22. Time series of daily sediment loads measured at USGS gaging sites.

sediment lost occurred during a high flow period typically ranging from three to five days. For example, in Water Year 2001, a four day period occurring from March 12 to 16 comprised the maximum four daily sediment loads measured during the entire year and represented 44 and 42 percent of the annual sediment load for the year at WNT2 and SQW2, respectively. Thus, much of the five day total may be considered a combination of two to three events per year.

The timing of peak monthly sediment losses was often similar at WNT2 and SQW2, but there were occasional large differences in peak magnitude (Figure 24). For example, in Water Year 1998, two sediment load peaks were evident but the second monthly peak in Squaw Creek was 2.5 times greater than measured in Walnut Creek. Similarly in Water Year 2000, monthly sediment loss exceeded 7000 tons in Squaw Creek but was less than 2000 tons in Walnut Creek (Figure 24). Many water years were characterized by two peaks in sediment export, with one peak associated with snowmelt in February and March, and a second higher peak occurring in May and June each year.

Greatest sediment transport typically occurred in May and June of each year (Figure 25), when on average these months accounted for 59.2 and 68.2% of the total annual load in Walnut and Squaw Creek watersheds, respectively (Table 11). The percentage of total annual sediment loads occurring in February and

Figure 23. Cumulative sediment load and discharge at USGS gaging sites.



March were similar, averaging 21 to 22%, whereas monthly sediment export were considerably lower in April (Figure 25). Overall, the period between February and July of each year accounted for approximately 94.6 and 96.3 percent of the total annual suspended sediment load in Walnut and Squaw Creek watersheds, respectively.

Annual sediment loss was slightly higher in Squaw Creek compared to Walnut Creek, averaging 0.69 and 0.65 tons/acre, respectively (Table 9). Sediment export ranged from 0.23 to 1.42 tons/acre in Walnut Creek watershed and from 0.08 to 1.75 tons/acre in Squaw Creek watershed. Annual sediment losses were lower in upper Walnut Creek watershed ranging from 0.06 to 0.64 tons/acre and the overall average export from WNT1 was significantly less (0.29 tons/acre) than either WNT2 or SQW2. Annual sediment yield was significantly related to annual discharge (Figure 26). The slopes of the regression lines for WNT2 and SQW2 were very similar (0.092 to 0.093), with the best-fit line of SQW2 plotting parallel but slightly higher than WNT2. Although more scatter was evident in the Squaw Creek data than Walnut Creek, the similar regression line equations suggests that annual sediment losses from both Jasper County watersheds can be approximated if annual discharge is known. For an average discharge of 8.5 inches, sediment losses would be approximately 0.61 and 0.67 tons/acre in Walnut and Squaw Creek watersheds, respectively. The slope of the regression line for WNT1 is approximately one-half of the slopes of either WNT2 or SQW2 (Figure 26). The WNT1 watershed area consists chiefly of row crop land use, but the terrain is considerably flatter than the lower portions of either Walnut or Squaw creek watersheds. Hence, sediment yield from this region was less than the overall watershed area. On a monthly basis, mean

Table 10. Percentage of annual sediment loads at WNT2, SQW2 and WNT1 from nonconsecutive 1-day, 5-day, 10-day and 20-day periods.

Water Year	WNT2 1-day	WNT2 5-day	WNT2 10-day	WNT2 20-day	SQW2 1-day	SQW2 5-day	SQW2 10-day	SQW2 20-day	WNT1 1-day	WNT1 5-day	WNT1 10-day	WNT1 20-day
1996	27.8%	68.9%	80.3%	91.7%	46.2%	80.2%	87.4%	93.7%	42.6%	67.5%	78.3%	86.3%
1997	18.2%	70.3%	81.8%	90.6%	33.1%	68.7%	80.8%	88.9%	44.6%	69.0%	77.5%	84.0%
1998	25.0%	59.9%	73.0%	82.8%	55.7%	81.4%	85.5%	89.9%	23.7%	51.1%	61.6%	72.3%
1999	22.6%	67.8%	80.3%	87.5%	28.6%	67.0%	81.5%	88.1%	19.8%	57.6%	70.2%	77.9%
2000	47.5%	63.5%	74.8%	83.4%	80.5%	94.0%	96.5%	98.0%	67.7%	81.0%	86.3%	89.6%
2001	20.3%	50.9%	71.0%	87.0%	13.0%	55.7%	81.2%	90.6%	15.3%	50.1%	75.4%	85.1%
2002	15.7%	59.3%	78.8%	88.9%	21.2%	58.2%	67.6%	77.4%	18.9%	47.8%	61.9%	72.1%
2003	17.2%	60.1%	78.7%	91.8%	35.4%	70.3%	81.6%	89.1%	20.8%	50.1%	64.8%	77.0%
2004	14.0%	50.8%	68.5%	80.7%	19.5%	55.0%	74.2%	86.4%	13.9%	42.5%	61.5%	71.7%
2005	40.3%	72.3%	83.8%	90.7%	33.7%	65.7%	78.8%	87.9%	28.3%	68.7%	76.3%	82.7%
average	24.9%	62.4%	77.1%	87.5%	36.7%	69.6%	81.5%	89.0%	29.6%	58.5%	71.4%	79.9%

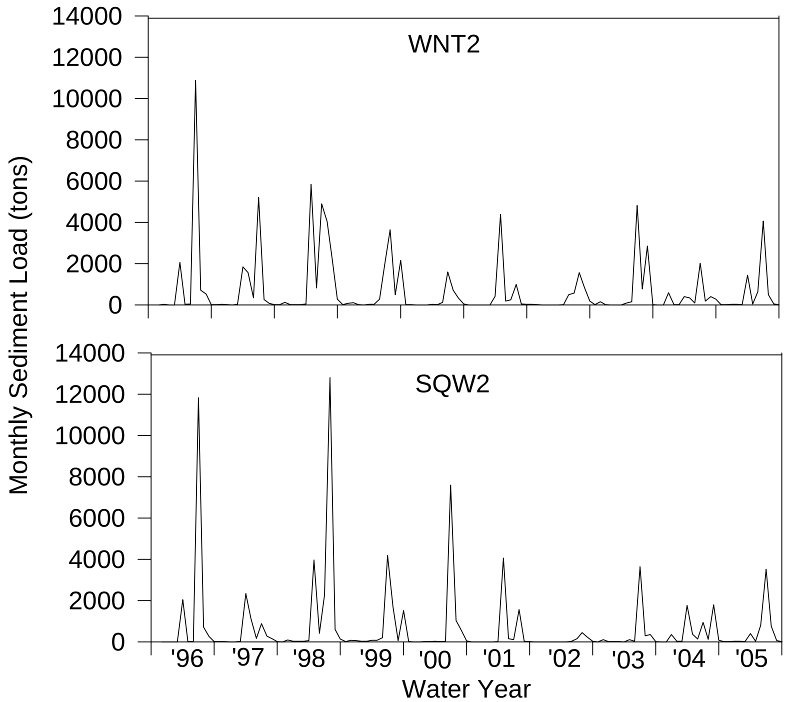


Figure 24. Time series of monthly sediment loads at WNT2 and SQW2.

sediment yield followed monthly discharge patterns, with highest yields occurring in May and June when sediment export exceeded one ton/acre (Table 11). Little sediment transport occurred during the September through January period at WNT2 and SQW2 when export was less than 0.06 tons/acre.

Suspended Sediment Concentrations

Suspended sediment concentrations were similar in Walnut Creek and Squaw Creek, with average and median values of 104.1 and 46.0 mg/l at WNT2 and 90.1 and 42.7 mg/l at SQW2, respectively (Table 9). Annual mean concentrations ranged from 78.3 to 153.1 mg/l in Walnut Creek and 47.5 to 128.6 mg/l in Squaw Creek. Maximum daily suspended sediment concentrations exceeded 3,000 mg/l during two water years in both watersheds. Median sediment concentrations were less than mean values, indicating that the means were biased by occasional peak concentrations.

Suspended sediment concentrations most commonly ranged between 20-50 mg/l, with concentrations within this range approximately 35 to 39 percent of the time (Table 12). Overall, concentrations were less than 100 mg/l 76.5 to 83 percent of the time at WNT2 and SQW2, respectively. Concentrations greater than 500 mg/l were infrequent, occurring 3.2 percent of the time in Walnut Creek and 2.8 percent in Squaw Creek.

Seasonally, highest suspended sediment concentrations occurred in May each year when mean concentrations exceeded 200 mg/l (Table 11). Mean monthly concentrations decreased from May to lowest concentrations measured during December and January. Considering daily concentration values by month (Figure 27) suggests more variability in sediment concentration in Walnut Creek compared to Squaw Creek particularly during January and February. A much stronger seasonal signal is evident in the daily sediment concentrations in Walnut Creek watershed.

The relationship of suspended sediment concentrations to discharge is stronger in Squaw Creek than Walnut Creek and weakest at WNT1 (Figure 28). The slope of the

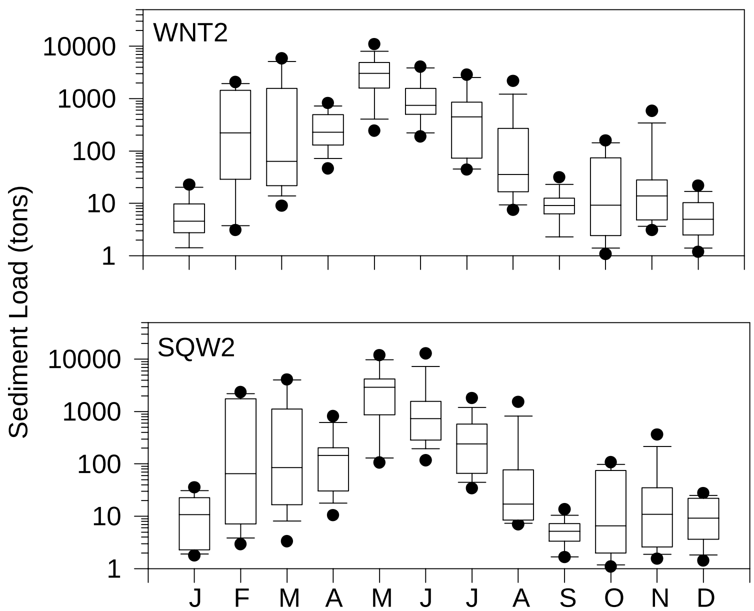


Figure 25. Box plots of monthly sediment loads.

regression line is steepest at SQW2 (0.51) compared to WNT2 (0.39). Although the relation between concentration and discharge appears similar at higher discharge values, more scatter is evident in the plot for WNT2, particularly at lower discharge values. While this may suggest that lower discharge maintains higher suspended sediment concentrations at Walnut Creek than Squaw Creek, it is likely a function of how discharge is measured in both watersheds. In Squaw Creek, discharge is measured in a free-flowing channel in an open channel control station. In Walnut Creek upstream and downstream v-notched weirs were used to measure discharge. It is probable that the weir setup at WNT2 may have caused 'stirring' of the water as water falling over the first (upstream) weir 'stirred' the sediments in the pool before exiting over the downstream weir for sediment sampling. Low flow effects were apparent in the concentration range breakdown that indicated a slightly higher proportion of water samples with sediment concentrations less than 20 mg/l at WNT2 than SQW2 (total of 0.7 percent more). Overall, the scatter in the relationship of suspended sediment concentrations to discharge indicates difficulty in developing a sediment rating curve to predict sediment losses from discharge alone.

On a seasonal basis, the relationship of suspended sediment concentrations to discharge was highly variable (Figures 29 and 30). The rating curve was best established in May, when the r^2 values were 0.69 and 0.60 in Walnut and Squaw creeks, respectively. The slopes of the regression lines were highest in May for both watersheds, with the regression slopes greater than 0.53 between the months of February to July. The r^2 values for these months ranged from 0.33 to 0.69. During the remaining months of the year, the r^2 values were considerably less, ranging from 0.00 to 0.34. A weak positive relation in daily sediment and discharge evident in Squaw Creek in October and November was not apparent in data from Walnut Creek.

Trends

Daily suspended sediment concentrations and loads values were log10 transformed prior to regression analyses. The daily sediment values were highly autocorrelated, with autocorrelation coefficients ranging from 0.75 to

Table 11. Summary of average monthly sediment loads and concentrations at WNT2, SQW2 and WNT1.

Month	WNT2 (tons)	WNT2 (% of total)	WNT2 (tons/ac)	WNT2 (mg/l)	WNT2 (stdev)	SQW2 (tons)	SQW2 (% of total)	SQW2 (tons/ac)	SQW2 (mg/l)	SQW2 (stdev)	WNT1 (tons)	WNT1 (% of total)	WNT1 (tons/ac)	WNT1 (mg/l)	WNT1 (stdev)
Jan	75	0.1%	0.0006	19.0	7.9	140	0.2%	0.0012	37.2	18.4	35	0.3%	0.0008	23.0	8.5
Feb	6284	7.5%	0.0487	37.1	30.3	6723	8.4%	0.0574	61.5	39.5	1822	14.5%	0.0423	29.7	18.5
Mar	12350	14.7%	0.0958	53.8	41.7	9751	12.1%	0.0832	69.8	58.5	1743	13.8%	0.0404	38.3	27.4
Apr	3181	3.8%	0.0247	53.5	36.0	2003	2.5%	0.0171	46.8	25.3	423	3.4%	0.0098	39.1	32.0
May	36266	43.3%	0.2813	149.6	100.3	35142	43.7%	0.3000	108.2	77.6	4360	34.6%	0.1011	82.1	58.7
Jun	13398	16.0%	0.1039	146.8	45.2	19696	24.5%	0.1681	102.3	38.4	2021	16.0%	0.0469	90.9	55.4
Jul	7869	9.4%	0.0610	99.1	24.1	4129	5.1%	0.0352	61.9	23.1	1082	8.6%	0.0251	104.5	72.4
Aug	3018	3.6%	0.0234	63.3	26.4	1839	2.3%	0.0157	36.0	18.1	801	6.4%	0.0186	75.6	47.3
Sep	108	0.1%	0.0008	48.3	24.0	56	0.1%	0.0005	30.6	9.9	55	0.4%	0.0013	60.9	40.2
Oct	423	0.5%	0.0033	39.6	16.4	304	0.4%	0.0026	33.7	17.2	91	0.7%	0.0021	50.8	23.0
Nov	793	0.9%	0.0062	27.2	11.9	531	0.7%	0.0045	31.9	18.7	137	1.1%	0.0032	23.4	13.0
Dec	74	0.1%	0.0006	25.0	13.9	124	0.2%	0.0011	35.4	20.8	31	0.2%	0.0007	22.4	11.3

0.83. The Durbin Watson (DW) statistic on the residuals without correcting for autocorrelation ranged from 0.33 to 0.49 (extremely low and indicate autocorrelation of residuals). When the regression included autocorrelation, the DW statistic ranged from 1.65 to 1.83 (values close to 2.0 indicate independence of the residuals and the appropriate assumption of an AR(1) time series model).

Like discharge, there was evidence that the slopes (or changes over time) were not the same for each month. Therefore, an overall trend model was utilized that included interaction terms and allowed for each month to have its own intercept and slope. This was done to evaluate the months where the overall trend had the greatest magnitude. The significance of trends of December values were in the 'date' slope significance value (since indicator variables were utilized for January through November). The significance of the interaction term indicated if it was statistically different from December.

Correlations among sediment concentrations at downstream Walnut Creek (WNT2) with upstream WNT1 and control watershed SWQ2 were assessed. Not unexpectedly, all covariates were highly correlated significant (at P<.0001 level). The correlation between log (sediment concentration) from WNT2 and WTN1 was high (r=0.72) as well as between SQW2 and WNT2 (r=0.72). The correlation between WNT2 sediment and discharge was also high (r=0.54). Not surprisingly, the correlation was higher with daily mean discharge than with baseflow (r=0.42). Therefore, discharge, upstream, and control watershed were meaningful covariates or explanatory variables when examining for trends at WNT2 for sediment concentrations and loads.

Because the data exhibited a strong seasonal signal a second regression time series model was created. Adjustment for seasonality was performed by the addition of trigonometric functions as explanatory variables. Terms for sine and cosine functions corresponding to the seasonal cycle of 2˜radians per cycle were added to the regression model. For the annual cycle for sediment, the following covariates were added to the trend models:

Cosine ([2 * 3.141593 * t] / 365)
Sine ([2 * 3.141593 * t] / 365),
where t was the sample number.

PROC AUTOREG in SAS 9.1 was utilized to run the time series regressions. A 'date' variable was included in all models to test for linear trends over the 10-year project monitoring period. The Yule Walker regression estimates were utilized that incorporated the time series error model. As with discharge, values were estimated for 1995 as well as 2005, adjusting at the LS Means of the model covariates at each month. These estimated log values were transformed to the original scale and the

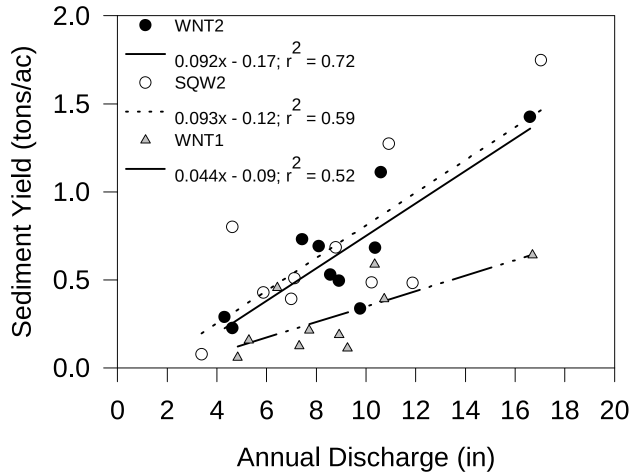


Figure 26. Relation of annual sediment yield to discharge at USGS gaging sites.

percent change calculated (a negative value indicated a decrease).

Results indicated that with no correction for seasonality, autocorrelation, upstream covariates or discharge, there was a non-significant but slight decreasing trend in sediment concentration at WNT2 and a significant decreasing trend in sediment at the upstream, WNT1 station. With adjustments made for autocorrelation and various covariates, trends in sediment concentrations and loads at WNT2 were mixed. With adjustments made for log discharge at WNT2, there was a statistically significant 20-30 percent decrease in sediment concentrations at loads during the months of May and June (Table 13). Other months also appeared to suggest a decrease, but they were not statistically significant. Sediment concentrations during the months of January and December appeared to increase over time.

Table 12. Frequency of occurrence of various suspended sediment concentrations for the 10-year monitoring period.

Concentration Range (mg/l)	Percent of Samples Within Concentration Range		
	WNT2	SQW2	WNT1
<10	2.0%	1.7%	5.5%
10-20	16.8%	16.4%	15.4%
20-50	35.0%	37.5%	39.0%
50-100	22.7%	27.4%	22.4%
100-200	12.9%	9.6%	11.4%
200-300	4.2%	2.4%	3.4%
300-400	1.9%	1.1%	1.2%
400-500	1.3%	1.0%	0.6%
500-1000	2.0%	1.7%	0.7%
>1000	1.2%	1.1%	0.3%

Adding covariates of log WNT2 discharge and log WNT1 sediment concentrations and loads, WNT2 sediment concentrations showed an increasing, though non-significant trend over 10 years, whereas WNT2 loads did not change appreciably (Table 14). Compared with conditions at SQW2, sediment concentrations and loads at WNT2 appeared to decrease over time, but except for the months of October and November, the changes were not statistically significant. Finally, combining all covariates (WNT2 discharge, WNT1 sediment and SQW2 sediment) into a single regression equation indicated there was no statistically significant trends in either concentration or load at WNT2. Concentrations and loads appeared to increase at WNT2 during many months, but the changes were non-significant. Minor decreasing concentrations and loads were apparent during months of October and November.

Trends in WNT1 and SQW2 sediment concentrations and loads were evaluated individually using their own discharge record as a covariate. WNT1 upstream concentrations and loads generally decreased over the 10 year monitoring period, with the magnitude of decrease approximately 60-70 percent for the March to July periods (Table 15). Most of the monthly decreases were statistically significant. After adjusting for discharge, downstream SQW2 concentrations and loads decreased over the 10 years in the spring and summer months, but increased during the winter months (Table 15). The magnitude of decrease was approximately 35-65 percent for the May to July periods.

For the time-series regression model for WNT2 with downstream discharge and seasonal sine/cosine covariates, no significant changes over time were observed. There was an 11%

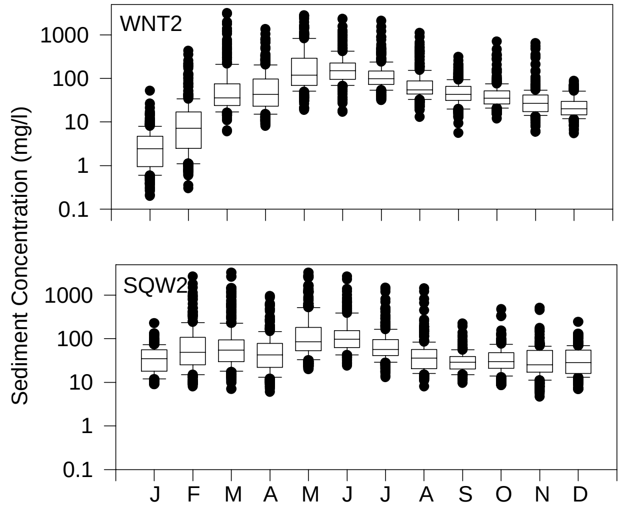


Figure 27. Box plots of suspended sediment concentrations by month at WNT2 and SQW2.

reduction in sediment concentrations at WNT2 and a 3% reduction in sediment loads, but the trends were not significant. When the upstream covariate (WNT1) was added to the model, WNT2 sediment concentrations showed a statistically significant decrease over time but loads showed a non-significant increase over the 10-years.

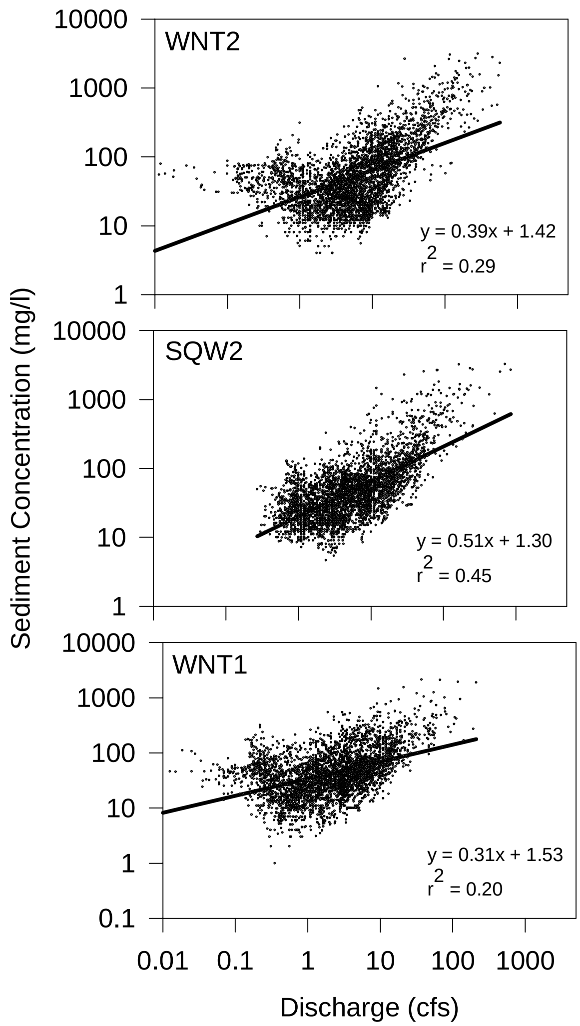


Figure 28. Relation of log daily discharge to log suspended sediment concentrations at USGS gaging sites.

Discussion

Daily monitoring indicated that suspended sediment transport in the Walnut and Squaw Creek watersheds was highly event driven. While single day discharge events typically accounted for six to eight percent of the annual discharge, single day suspended sediment loads accounted for 25 to 37 percent of annual sediment total. A 20-day period in any given water year accounted for as much as 98 percent of the annual sediment total. This pattern of rapid conveyance of discharge and sediment loads is typical of incised channels (Happ et al., 1940; Knox, 1987; Shields et al., 1995; Faulkner and McIntyre, 1996). In incised channels, flood events peak higher and faster as more water is contained within the channel, promoting efficient transport of suspended sediment downstream. The degree of channel incision is similar in both Walnut and Squaw Creek watersheds. At the downstream gage in Walnut Creek, the channel was measured to be approximately 40 feet wide and 10.4 feet deep (width-depth ratio of 3.8), whereas the Squaw Creek channel near SQW2 was 54 feet wide and 12.5 feet depth (width-depth ratio of 4.3). Thus, channel incision in the Walnut and Squaw Creek watersheds contained all but the most exceptional flood flows and contributed to the rapid downstream conveyance of sediment.

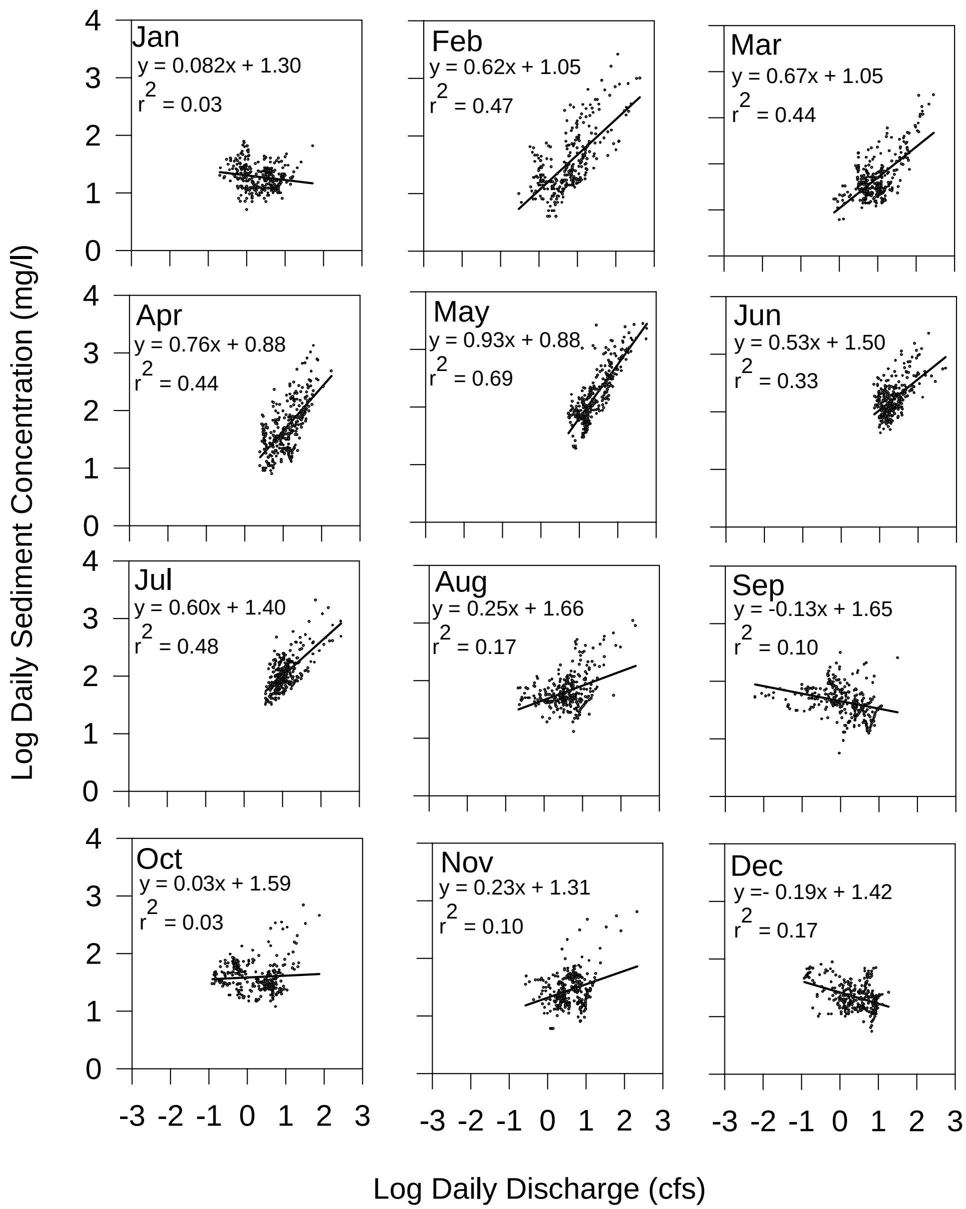


Figure 29. Relation of log daily discharge to log suspended sediment concentrations by month at WNT2.

Sediment yields and concentrations at Walnut and Squaw creeks were very similar over the 10-year monitoring program. Both watersheds move the majority of sediment during May and June each year during peak discharge events, and annually have a strong relation between discharge and sediment yield. The months of May and June accounted for 59.3 and 68.2 percent of the annual sediment loss at WNT2 and SQW2, respectively, whereas the months of February through July accounted for 94.7 to 96.3 percent of the annual sediment loss. Suspended sediment concentrations were less than 50 mg/l for more than half of the measurement days, and less than 100 mg/l approximately 80 percent of the

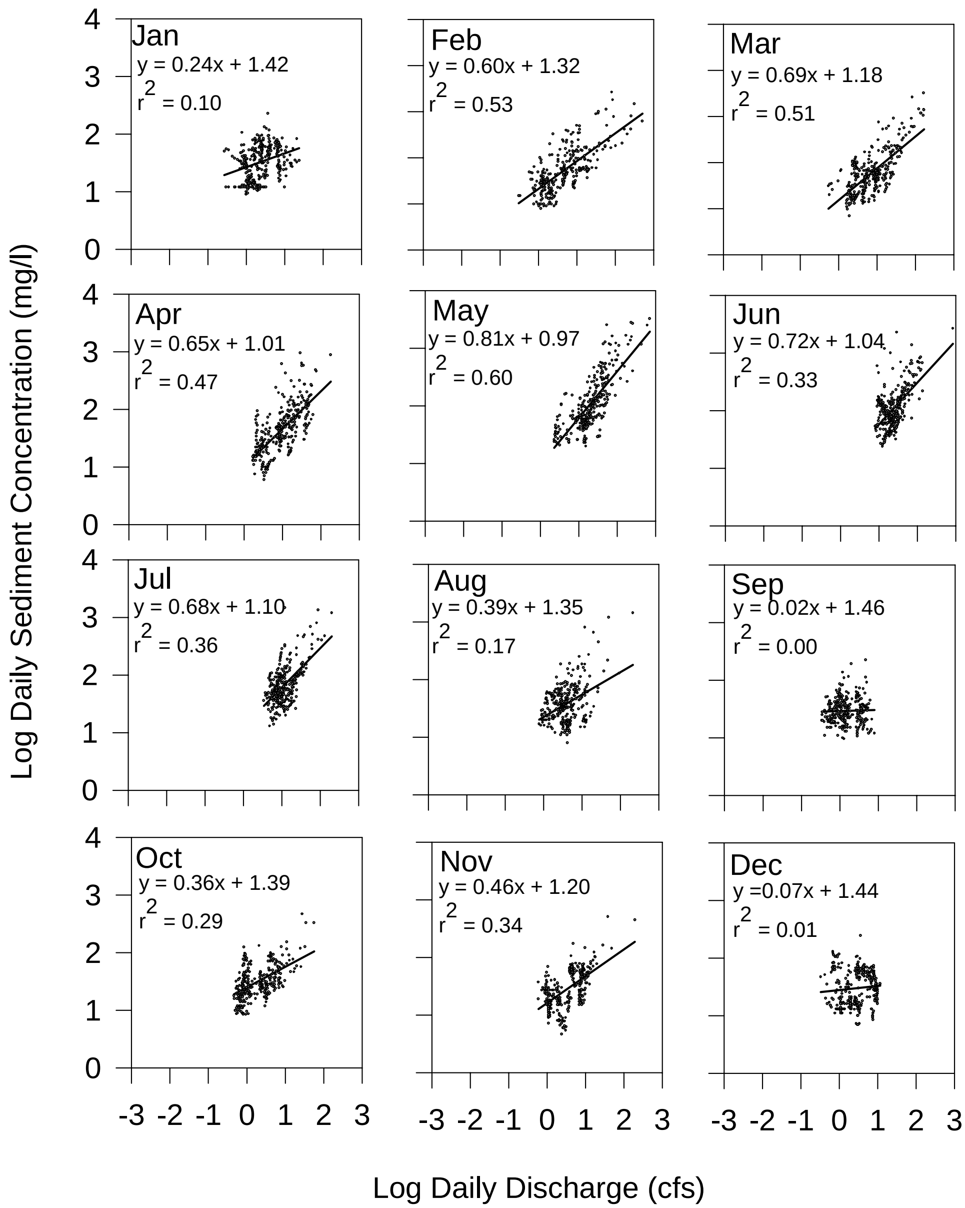


Figure 30. Relation of log daily discharge to log suspended sediment concentrations by month at SQW2.

time. Concentrations exceeded 500 mg/l approximately 3 percent of the time.

Trends in daily sediment concentrations and loads at WNT2 reflect the variable nature of sediment transport and appear to give mixed signals depending on the regression model used. With one model, a decreasing trend in sediment concentrations and loads over time was observed at WNT2 using the model with WNT2 discharge as a single covariate. In this case, sediment at WNT2 was reduced by 20 to 30 percent during the months of May and June, the period when most sediment loss occurred. Minor increases that appeared to occur in winter months would not be particularly important since these months contribute very

Table 13. Trend test for mean daily sediment concentrations and loads at WNT2. Each month was allowed a different slope and intercept. Trends were adjusted for autocorrelation and covariates (as indicated).

WNT2 Covariates: log WNT2 discharge

	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate (compared to Dec.)	Sig.
Suspended	January	15.1	29.8	97	0.6327	ns
Sediment	February	38.6	35.8	-7	0.1226	ns
Conc.	March	57.3	45.4	-21	0.0615	+
(mg/l)	April	66.6	61.4	-8	0.1421	ns
	May	152.7	106.7	-30	0.0331	*
	June	166	133.4	-20	0.0744	+
	July	113.1	106.6	-6	0.1601	ns
	August	65	72.7	12	0.3193	ns
	September	28.3	36.9	30	0.5318	ns
	October	40.4	26.7	-34	0.0153	*
	November	44.6	31.4	-30	0.0078	*
	December	17.6	29.6	69	0.075	+
Suspended	January	0.1	0.2	84	0.7828	ns
Sediment	February	0.6	0.6	6	0.2672	ns
Loads	March	1.4	1.2	-17	0.0967	+
(tons)	April	2.3	2	-12	0.135	ns
	May	11	7.8	-29	0.0448	*
	June	11	8.5	-22	0.0741	+
	July	3.9	3.5	-12	0.1333	ns
	August	0.8	0.7	-4	0.1913	ns
	September	0.1	0.1	-9	0.1572	ns
	October	0.2	0.1	-46	0.0054	*
	November	0.5	0.3	-30	0.0142	*
	December	0.1	0.2	67	0.0889	+

WNT2 Covariates: log WNT2 discharge, log WNT1 sediment

	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate (compared to Dec.)	Sig.
Suspended	January	15.4	28.9	87	0.8084	ns
Sediment	February	30.4	42.3	39	0.4854	ns
Conc.	March	42.8	53.4	25	0.3113	ns
(mg/l)	April	49.8	68.6	38	0.4807	ns
	May	132.2	133.7	1	0.1054	ns
	June	123.9	174.1	41	0.5197	ns
	July	91.3	138.7	52	0.6786	ns
	August	63.2	84.5	34	0.4254	ns
	September	34.4	45.5	32	0.4112	ns
	October	38.7	32.9	-15	0.0259	*
	November	30.8	30.9	0	0.05	*
	December	16.3	28.5	75	0.0194	*
Suspended	January	0.1	0.2	45	0.9053	ns
Sediment	February	0.5	0.8	51	0.8232	ns
Loads	March	1.2	1.4	12	0.5152	ns
(tons)	April	1.7	2.1	20	0.6533	ns
	May	10.3	9.2	-11	0.1828	ns
	June	8.9	11.2	26	0.7637	ns
	July	3.3	4.3	32	0.8667	ns
	August	0.7	0.9	22	0.6808	ns
	September	0.1	0.1	-29	0.0656	+
	October	0.2	0.1	-36	0.0205	*
	November	0.4	0.3	-19	0.0718	+
	December	0.1	0.2	40	0.1654	ns

ns = not significant
+ = significant at P<0.1
* = significant at P<0.05
** = significant at P<0.01

Table 14. Trend test for mean daily sediment concentrations and loads at WNT2. Each month was allowed a different slope and intercept. Trends were adjusted for autocorrelation and covariates (as indicated).

	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate (compared to Dec.)	Sig.
WNT2 Covariates: log WNT2 discharge, log WNT1 sediment, log SQW2 sediment						
Suspended	January	18.1	24.9	38	0.7964	ns
Sediment	February	30.7	42.1	37	0.8307	ns
Conc.	March	46.6	52.1	12	0.6496	ns
(mg/l)	April	49.1	61.2	25	0.9206	ns
	May	141.6	135.4	-4	0.34	ns
	June	122.9	181.5	48	0.659	ns
	July	90.9	133.2	46	0.6767	ns
	August	63	78.1	24	0.9044	ns
	September	36.9	45.8	24	0.9125	ns
	October	39.6	32.6	-18	0.1388	ns
	November	33.9	25.5	-25	0.0439	*
	December	19.1	24.5	29	0.2532	ns
Suspended	January	0.1	0.1	17	0.8815	ns
Sediment	February	0.5	0.8	50	0.3709	ns
Loads	March	1.3	1.3	3	0.8077	ns
(tons)	April	1.8	1.9	9	0.9442	ns
	May	11	8.8	-20	0.3141	ns
	June	9.4	11.1	19	0.8502	ns
	July	3.5	4	15	0.9292	ns
	August	0.7	0.8	6	0.8761	ns
	September	0.1	0.1	-29	0.2039	ns
	October	0.2	0.1	-35	0.099	+
	November	0.4	0.3	-33	0.0831	+
	December	0.1	0.2	12	0.6367	ns

	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate (compared to Dec.)	Sig.
WNT2 Covariates: log WNT2 discharge, log SQW2 sediment						
Suspended	January	18.8	23.5	25	0.6652	ns
Sediment	February	34.3	38.1	11	0.9962	ns
Conc.	March	56.9	45.6	-20	0.3186	ns
(mg/l)	April	57.8	52.9	-9	0.5579	ns
	May	167.2	114.8	-31	0.1446	ns
	June	151.6	155.1	2	0.8081	ns
	July	107.9	108.7	1	0.7714	ns
	August	66.1	68.4	3	0.8329	ns
	September	35.1	42.7	22	0.7759	ns
	October	43.4	27.9	-36	0.0866	+
	November	42.4	23.4	-45	0.0129	*
	December	21	23.2	11	0.6578	ns
Suspended	January	0.1	0.1	19	0.8128	ns
Sediment	February	0.5	0.7	24	0.7545	ns
Loads	March	1.4	1.2	-18	0.3831	ns
(tons)	April	2	1.7	-13	0.4817	ns
	May	12.3	7.8	-37	0.1119	ns
	June	11	9.6	-13	0.4952	ns
	July	4.1	3.4	-17	0.4161	ns
	August	0.8	0.7	-14	0.4704	ns
	September	0.1	0.1	-12	0.5111	ns
	October	0.2	0.1	-43	0.0543	+
	November	0.4	0.3	-42	0.0328	*
	December	0.1	0.2	11	0.6746	ns

ns = not significant

+ = significant at P<0.1

* = significant at P<0.05

** = significant at P<0.01

Table 15. Trend test for mean daily sediment concentrations and loads at WNT1 and SQW2. Each month was allowed a different slope and intercept. Trends were adjusted for autocorrelation and covariates (as indicated).

	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate (compared to Dec.)	Sig.
WNT1 Covariates: log WNT1 discharge						
Suspended	January	23.5	18.2	-23	0.8188	ns
Sediment	February	42.9	20.3	-53	0.2756	ns
Conc.	March	65.4	21.4	-67	0.0485	*
(mg/l)	April	60.1	21.8	-64	0.0925	+
	May	119.7	40.1	-66	0.0606	+
	June	201.3	50.8	-75	0.0105	*
	July	158.6	47.5	-70	0.0314	*
	August	75.1	51.1	-32	0.8942	ns
	September	32.8	42.4	29	0.149	ns
	October	54.3	26.6	-51	0.3178	ns
	November	50.4	20.8	-59	0.091	+
	December	26.5	19.1	-28	0.2504	ns
Suspended	January	0.1	0	-31	0.8111	ns
Sediment	February	0.2	0.1	-48	0.3699	ns
Loads	March	0.5	0.2	-64	0.0709	+
(tons)	April	0.6	0.2	-63	0.091	+
	May	2.7	0.9	-64	0.0737	+
	June	5.6	1.3	-76	0.0064	*
	July	2.2	0.6	-73	0.0152	*
	August	0.4	0.2	-41	0.5803	ns
	September	0.1	0	-31	0.8583	ns
	October	0.1	0	-64	0.0697	+
	November	0.2	0.1	-58	0.0994	+
	December	0.1	0	-25	0.3285	ns

	Month	1995 Discharge Estimate (Adjusted)	2005 Discharge Estimate (Adjusted)	% Change over 10 years	Prob.> t on slope estimate (compared to Dec.)	Sig.
SQW2 Covariates: log SQW2 discharge						
Suspended	January	21	45.3	116	0.997	ns
Sediment	February	52.5	52.1	-1	0.039	*
Conc.	March	53.4	54.9	3	0.055	+
(mg/l)	April	57.7	53.1	-8	0.03	*
	May	117.5	75.6	-36	0.0021	*
	June	201.5	80	-60	<.0001	**
	July	104.1	54.9	-47	0.0004	**
	August	46.5	33.8	-27	0.0056	*
	September	25.3	21.1	-17	0.0153	*
	October	33.5	25	-26	0.0045	*
	November	26.6	51.7	94	0.7424	ns
	December	20.9	45.2	116	0.0058	*
Suspended	January	0.1	0.3	107	0.9106	ns
Sediment	February	0.6	0.8	21	0.1438	ns
Loads	March	1.1	1.1	9	0.0895	+
(tons)	April	1.5	1.3	-10	0.0312	*
	May	6.9	4.4	-37	0.0025	*
	June	14.9	5.3	-65	<.0001	**
	July	3.3	1.7	-47	0.0005	**
	August	0.5	0.4	-24	0.01	*
	September	0.1	0.1	-13	0.0261	*
	October	0.2	0.1	-34	0.0027	*
	November	0.2	0.4	81	0.614	ns
	December	0.2	0.3	116	0.0076	*

ns = not significant
+ = significant at P<0.1
* = significant at P<0.05
** = significant at P<0.01

little to annual sediment losses. However, when the upstream covariate was added to the regression model, downstream concentrations and loads appeared to increase over time. Thus, like discharge trends, it would appear that much of the apparent decrease in sediment concentrations and loads at WNT2 was the result of decreasing trends measured at the WNT1 gage. Trends measured at WNT1 were considerably more significant than those measured at WNT2, and most months showed evidence for a 60-70 percent decrease in sediment concentration and loads over 10 years. Thus, it is difficult to attribute decreasing sediment losses at the Walnut Creek outlet to project land restoration and management changes when the much of the reduction appeared to occur outside the area of treatment.

The reduction in sediment in the watershed area above the WNT1 gage may be traced to a pasture that was located immediately upstream of the gaging station. During the first years of the project (pre-2000), a pasture was used in this area, but later (post-2000), usage of the pasture was apparently discontinued. The 1990 statewide aerial photographs show this area actively used as a grazed pasture, whereas the 2000 aerial photographs show the area dominated by ungrazed grass. The 1999 project report (Schilling and Thompson, 1999) noted that cattle were located in this pasture as of 1997. High fecal coliform bacteria counts were also found at WNT1 during the 1995-1997 study period, but bacteria counts have declined significantly since this time (see fecal coliform section). Hence, it would appear that much of the sediment reduction measured at WNT1 was probably the result of abandonment of a pasture located immediately upstream.

Comparisons of trends in Walnut Creek and Squaw Creek sediment losses indicate little significant difference between the two watersheds. When SQW2 sediment concentrations and loads were added as covariates to the WNT2 regression model, it was evident that WNT2 concentrations and loads did not change relative to SQW2. A significant decreasing trend in October and November was observed in Walnut Creek relative to Squaw Creek, but this change would not affect annual sediment losses to any great degree. Interestingly, statistically significant decreases in sediment concentrations and loads occurred during many months at SQW2 but these changes did not substantially affect the paired comparisons with WNT2 (i.e., resulting in significant increasing trend at WNT2 relative to SQW2). The reason for the decreasing trend at SQW2 over time is unclear, but it is probable that climate and discharge patterns played a significant role. Several large discharge and sediment loss events occurred in Squaw Creek in the first five years of the project, whereas the last few years, and 2002 in particular, were marked by low precipitation and streamflow. Thus, the benefits of a paired watershed comparison become especially evident when attempting to explain trends over time. When Squaw Creek results were paired with Walnut Creek, no significant trends over time were evident.

Overall, statistical analyses of sediment concentrations and trends at WNT2 offer little conclusive evidence that prairie restoration and land management changes at the Neal Smith refuge reduced sediment transport in Walnut Creek watershed.

FIELD PARAMETER MEASUREMENTS

Field parameters have been measured since 1995, including temperature, pH, turbidity, specific conductance, dissolved oxygen, and reduction-oxidation potential (redox). Collection of field data assists in characterizing surface water quality in the two watersheds and lends support to conclusions drawn about other chemical constituents. Field measurements refer to analytical determinations that document water conditions at the time of sample collection. Field measurements are also important for parameters that may be altered during storage and shipment to the laboratory (e.g. pH).

Field parameters were measured at ten sites in the Walnut and Squaw creek watersheds (Figure 3). A Hydrolab H20 multiprobe water analyzer was used to measure temperature, pH, turbidity, specific conductance, redox, and dissolved oxygen. Turbidity was measured using a Hach 2100P turbidimeter. All field equipment was calibrated on a regular basis prior to use. Field measurements of alkalinity, not analyzed during the latter portion of the project, are reported in Schilling and Thompson (1999) and Schilling et al. (2002).

The mean water temperature in the main channel of Squaw Creek ranged between 12.1 and 12.6 ºC, whereas temperature values in Walnut Creek averaged 13.2 and 14.0 ºC (Table 16). Temperature differences were likely related to the timing of routine sample collection in the watersheds since water samples were typically collected in Squaw Creek before Walnut Creek. Temperatures measured in tributary sites were higher than main stem sites because these sites were only sampled between April and September and did not have colder winter temperatures factored into the overall statistics.

The hydrogen-ion activity of the water is measured as pH. Field measurements of pH are more likely to represent the natural conditions of the water than laboratory results of pH (Hem, 1989). Walnut and Squaw creeks exhibited similar pH values over the sampling period (Figure 31), with mean annual pH values nearly the same at WNT2 and SQW2 (Table 16). The mean pH of SQW1 was slightly lower than other main stem sites and t-tests revealed that the mean pH measured at SQW1 was significantly lower than both SQW2 (7.86) and WNT1 (7.87) ($p<0.05$). Lowest pH values were observed at tributary sites, with lowest mean pH measured at SQW5 (7.36). Low pH and redox values have been occasionally observed at SQW5 due to upstream releases from a dairy operation (Schilling et al., 2002).

Specific conductance measures the ability of water to conduct electrical current and is directly related to the amount of dissolved ions in the water. Higher dissolved ion concentrations correspond to higher specific conductance. Both Walnut and Squaw creek watersheds had statistically higher specific conductance in upstream samples compared to downstream samples (Table 16, $p<0.05$). Mean annual specific conductance at SQW1 (574 μmhos/cm) was significantly higher than the mean at WNT1 (548 μmhos/cm), and mean specific conductance at SQW2 (549 μmhos/cm) was significantly higher than the mean at WNT2 (502 μmhos/cm) (Figure 32). Lowest specific conductance were observed in tributaries to Walnut Creek (WNT5 and WNT6) where values averaged 495 and 467 μmhos/cm, respectively. Seasonally, lowest specific conductance tended to occur in February and March following snowmelt and increased through the summer and fall (Figure 33). Increasing specific conductance values during this time is typical of streamflow consisting of predominantly baseflow that contains greater concentrations of dissolved solutes than runoff. Increasing dissolved solids concentrations in streamflow have been found in other southern Iowa watersheds during low baseflow periods (Horick and Steinhilber, 1973). All specific conductance values were within the normal range of specific conductance for surface water in Iowa. No gradual changes in specific conductance over time were detected in either watershed.

Dissolved oxygen is a measure of the oxygen concentration in water and is an important constituent in the aquatic health of a stream. Highest mean dissolved oxygen was found in upstream main stem sites SQW1 and WNT1 where dissolved oxygen concentrations averaged 10.28 and 10.17 mg/l, respectively (Figure 34; Table 16). DO concentrations were slightly lower at their respective downstream sites SQW2 and WNT2 at 10.12 and 9.91 mg/l. However no significant differences were evident in DO among the main stem sites

Table 16. Summary of values for field parameters measured at 10 monitoring sites for water years 1996 to 2005.

	Temperature (deg C)	pH	Specific Conductance (umhos/cm)	Dissoved Oxygen (mg/l)	Reduction Oxidation Potential (mv)	Turbidity (NTU)		Temperature (deg C)	pH	Specific Conductance (umhos/cm)	Dissoved Oxygen (mg/l)	Reduction Oxidation Potential (mv)	Turbidity (NTU)
SQW1							WNT1						
n	181	171	176	177	105	183	n	184	173	181	178	104	186
min	-0.13	6.20	299	4.85	41	2	min	-0.15	6.28	295	2.97	138	2
max	23.4	8.56	670	17.59	553	768	max	28.6	8.87	686	18.74	562	827
mean	12.11	7.77	574	10.28	339	35	mean	13.16	7.87	548	10.17	343	55
median	13.1	7.82	581	10.30	330	17	median	14.62	7.87	559	10.00	335	26
stddev	5.48	0.40	57	2.42	80	82	stddev	6.16	0.39	56	2.66	75	98
SQW2							WNT2						
n	186	177	183	184	105	187	n	184	175	181	181	104	186
min	-0.18	6.49	6	5.53	106	5	min	-0.14	6.33	225	3.70	156	2
max	24.4	9.30	665	18.55	586	1000	max	29.3	9.02	645	18.10	546	1000
mean	12.62	7.86	549	10.12	351	65	mean	14.04	7.87	502	9.91	343	86
median	13.39	7.92	556	9.85	340	24	median	15.05	7.90	511	9.71	334	38
stddev	6.10	0.37	77	2.25	80	163	stddev	6.77	0.38	60	2.51	67	176
SQW3							WNT3						
n	130	124	127	127	76	136	n	128	121	126	126	75	134
min	0.33	6.12	289	5.29	163	1	min	2.16	6.39	244	4.30	214	3
max	21.17	9.02	697	19.13	552	517	max	25.8	8.90	672	15.88	551	183
mean	13.38	7.70	565	9.90	343	32	mean	14.18	7.85	526	9.63	341	24
median	14.34	7.72	563	9.86	329	13	median	14.61	7.88	532	9.72	325	13
stddev	4.12	0.39	61	2.18	75	70	stddev	3.81	0.37	64	2.01	70	29
SQW4							WNT5						
n	129	123	125	126	75	136	n	128	122	125	126	75	134
min	0.65	6.41	260	4.04	33	1	min	4.2	6.86	330	5.15	222	5
max	22.09	8.98	712	17.87	570	648	max	27.26	9.15	606	17.83	541	1000
mean	13.29	7.62	548	9.21	334	28	mean	16.72	7.99	495	9.80	341	54
median	14.4	7.64	553	9.21	321	8	median	17.57	8.01	494	9.65	326	26
stddev	4.14	0.36	68	2.13	89	82	stddev	4.91	0.38	51	2.20	65	122
SQW5							WNT6						
n	128	122	125	125	74	135	n	125	119	123	122	72	130
min	2.83	6.34	269	1.04	-66	2	min	3.7	6.50	261	3.83	183	1
max	23.75	8.59	693	17.80	581	821	max	27.74	9.08	609	16.00	541	1000
mean	13.29	7.36	554	7.79	323	21	mean	17.29	7.91	467	8.98	341	61
median	14.05	7.37	553	7.96	320	6	median	17.7	7.94	475	8.77	328	23
stddev	3.84	0.33	57	3.00	116	79	stddev	4.92	0.35	60	2.01	67	125

($p>0.34$) and no statistically significant changes in DO over time occurred at any of the main stem monitoring sites. Dissolved oxygen concentrations were lower at tributary sites than main stem sites, primarily due to limited seasonal sampling times (April to September).

Dissolved oxygen concentrations are influenced by many factors including water temperature (affects solubility of oxygen), algae respiration, sample collection time, and water turbulence. Combined effects of these factors were evident in the dissolved oxygen data (Figure 35). Lowest DO concentrations were detected at WNT2 during the months of July through September when average monthly DO levels were less than 8 mg/l (Figure 35). In general higher water temperatures and increased algae production likely contributed to reduced DO levels in late summer and fall. In contrast, mean DO concentrations between December and April were greater than 11-12 mg/l when water temperatures were lower and algae growth was negligible. A dissolved oxygen standard of 5 mg/ has been established by the State of Iowa for maintaining stable fish populations.

Reduction-oxidation potential (redox) reflects the intensity of the oxidizing or reducing conditions in the water. Positive potentials indicate the solution is oxidizing, whereas negative potentials indicate the solution is reducing. Mean redox values measured at the four main stem sampling sites were similar ranging from 339 to 351 mv (Table 16). The redox values were within the range found in natural surface waters, however data on specific redox values in Iowa was not available for comparison.

Turbidity measures the optical properties of water that cause light to be reflected and is closely related to concentrations of total suspended solids contained in the water including sand, silt, clay, plankton and other organic matter. Mean turbidity was highest at the two downstream sites in Walnut and Squaw creeks compared to their respective upstream sites (Table 16), with highest mean turbidity observed at WNT2 (86 nephelometric turbidity units or NTU). Turbidity values ranged from 1 NTU at WNT6 and SQW4 to values greater than 1,000 NTU at the downstream sites SQW2 and WNT2 after precipitation events (Figure 36). It should be noted that the upper measurement limit of turbidity was 1000 NTU and that in all likelihood, the maximum limit of detection was exceeded during the storm events. A t-test found that mean turbidity at WNT2 (86 NTU) was significantly higher than SQW2 (65 NTU), and WNT1 (55 NTU) was significantly higher than SQW1 (35 NTU) ($p<0.05$). SQW2 turbidity was also significantly higher than SQW1. Turbidity in Walnut Creek did not change relative to Squaw Creek during the monitoring project and neither downstream sampling site showed a statistically significant change in turbidity over time. Average annual turbidity at WNT2 ranged from a low of 42 NTU in WY 2000 and WY2004 to a high of 138 NTU in WY2003 (Figure 37). For comparison, average annual turbidity at SQW2 ranged from 22 NTU in WY2004 to 106 NTU in WY2000. Strong seasonal trends in turbidity were evident with greater runoff occurring in May through July producing higher average monthly turbidity than late fall and winter months (Figure 38).

Correlation among the field parameters measured at WNT1, WNT2, SQW1 and SQW2 site was assessed using Pearson's correlation coefficient. As expected, the strongest correlation was associated between temperature and dissolved oxygen. All four sites showed a strong negative correlation ranging between –0.77 to –0.71 ($p<0.05$). A strong negative correlation was also observed between specific

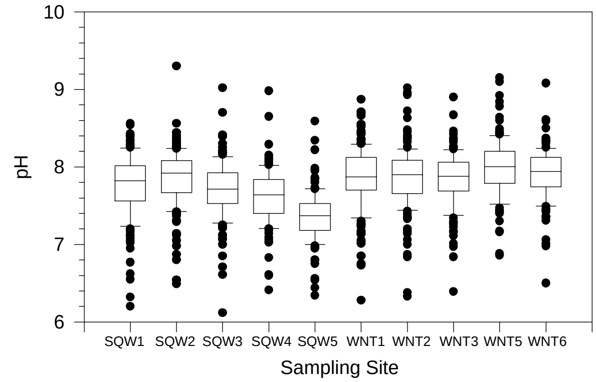


Figure 31. Box plots of pH measured at Walnut and Squaw creek monitoring sites.

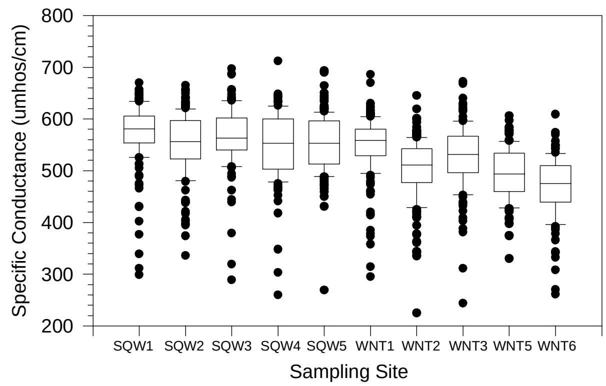


Figure 32. Box plots of specific conductance measured at Walnut and Squaw creek monitoring sites.

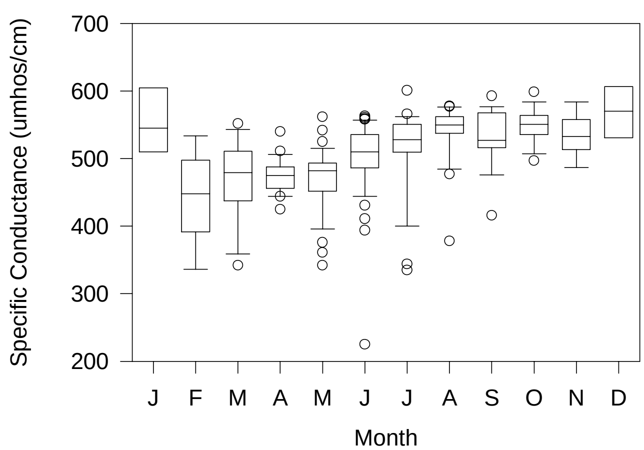


Figure 33. Box plots of specific conductance by month measured at Walnut and Squaw creek monitoring sites.

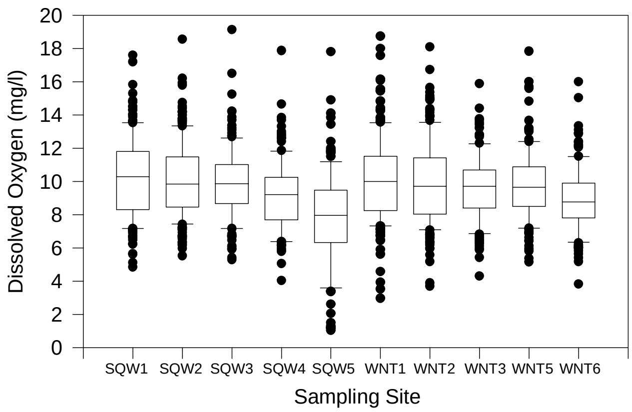


Figure 34. Box plots of dissolved oxygen measured at Walnut and Squaw creek monitoring sites.

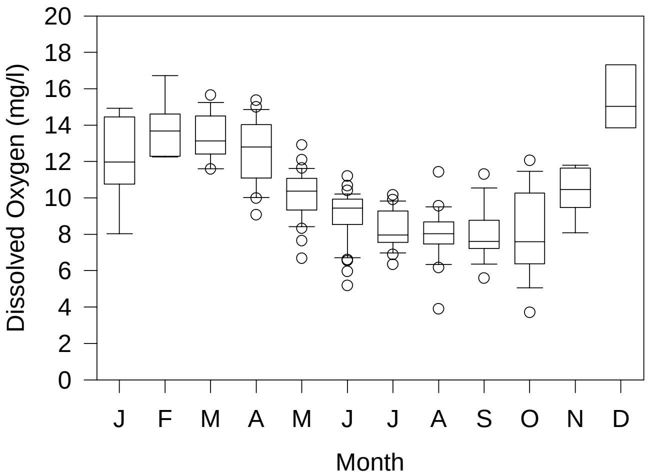


Figure 35. Box plots of dissolved oxygen by month measured at WNT2 monitoring site.

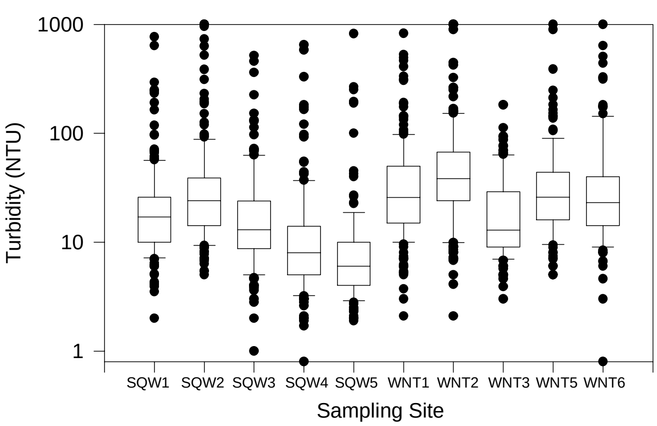


Figure 36. Box plots of turbidity measured at Walnut and Squaw creek monitoring sites.

conductance and turbidity at the four monitoring sites, ranging from –0.57 to –0.44 ($p<0.05$). The negative correlation indicates that when specific conductance increases during late summer and fall when baseflow dominates (Figure 33), turbidity decreases during that time period because runoff is reduced (Figure 38). In spring and early summer, runoff events produce high turbidity levels but contain lower dissolved ion concentrations in runoff (i.e., low specific conductance). Statistically significant but weaker positive associations were apparent between temperature and pH ($r = 0.24$ at WNT2 and SQW2) and pH and specific conductance ($r = 0.23$ to 0.28 at all four sites). A weak negative correlation between dissolved oxygen and turbidity (-0.16 to -0.32) is consistent with lower DO values and higher turbidity occurring during warmer months, particularly intersecting during late spring and early summer.

ANIONS

Nitrate-nitrogen (nitrate), chloride and sulfate are common anions detected in surface and groundwater and these constituents have been monitored in Walnut and Squaw creek surface water since 1995. These constituents are of interest for the Walnut Creek project because of their sources and manner in which they are delivered to streams. Nitrate is a common agricultural pollutant that is primarily delivered to streams through baseflow groundwater discharge and tile drainage (Hallberg, 1987; Schilling and Wolter, 2001; Schilling, 2002). Chloride concentrations in surface water have both natural and agricultural land use sources, with agricultural inputs associated with KCl fertilizer application in the watersheds. Sulfate may provide the best marker for tracking groundwater inputs to surface water independent of land use. Schilling and Thompson (2000) noted that sulfate concentrations in surface water, unlike nitrate and chloride did not relate significantly to the percentage of row crop land use in Walnut and Squaw creek watersheds.

Nitrate Concentrations

Nitrate concentrations have ranged between <0.5 to 14 mg/l at the Walnut Creek outlet (WNT2) and 2.1 to 15 mg/l at the downstream Squaw Creek outlet (SQW2) (Table 17). Mean nitrate concentrations were 1.7 mg/l higher at SQW2 than WNT2, and highest at the upstream monitoring sites in both watersheds, averaging 11.2 mg/l at WNT1 and 12.4 mg/l at SQW1 (Figure 39). The maximum nitrate concentration detected in either watershed was 19 mg/l at SQW1 measured on June 20, 2001 and May 4, 2005. Of the 204 samples collected for nitrate analysis over the 10-year monitoring period in Walnut Creek, 133, or 65.2 percent of total were greater than the EPA maximum contaminant level (MCL) for drinking water of 10 mg/l at WNT1 and 32.8 percent (67 samples) exceeded 10 mg/l at WNT2. In Squaw Creek, 150 out of 202 samples (74.3 percent) exceeded the 10 mg/l standard at SQW1 and 104 out of 202 samples (51.5 percent) exceeded 10 mg/l at SQW2. Both Walnut and Squaw Creek watersheds have shown a similar temporal pattern of detection, with higher concentrations observed in the spring and early summer months coinciding with periods of application, greater precipitation and higher stream flow (Figure 40).

Figures 41-43 show box plots for nitrate concentrations detected in water samples collected from the 10 sampling sites. Annually, mean nitrate concentrations ranged between 10.0 to 12.7 mg/l at WNT1, 6.8 to 9.5 m/l at WNT2, 10.5 to 13.8 mg/l at SQW1 and 8.2 to 11.5 mg/l at SQW2 (Figure 41). Highest mean annual concentrations at the four main stem sites all occurred in 1998, whereas the lowest annual concentration varied among water years 2000, 2002 and 2004 (Table 17). Greater differences among water year nitrate concentrations occurred in the subbasins (Figures 42 and 43). In Squaw Creek subbasins SQW4 and SQW5, a large increase in annual nitrate concentrations occurred from water year 1996 to 2005. In SQW4 subbasins, mean annual nitrate

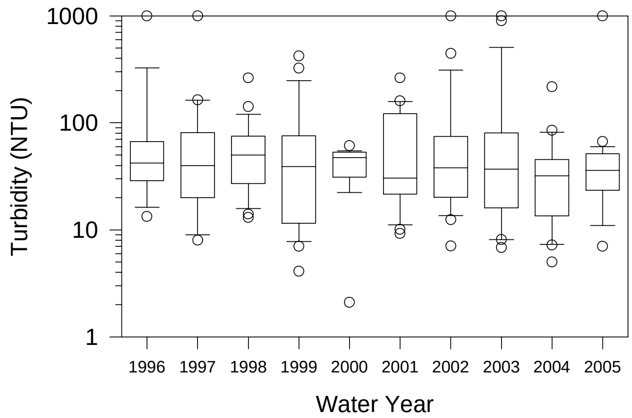


Figure 37. Box plots of turbidity by water year measured at WNT2 monitoring site.

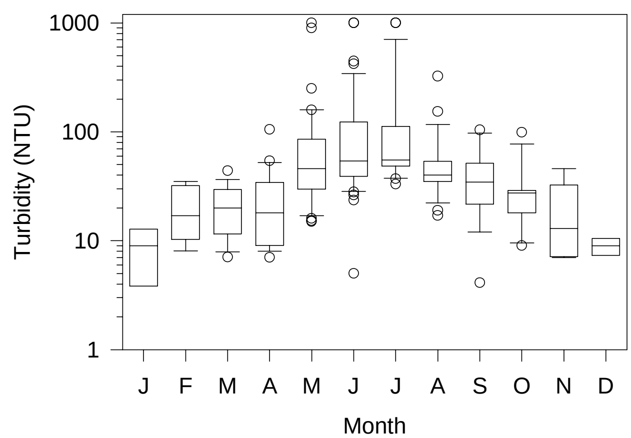


Figure 38. Box plots of turbidity by month measured at WNT2 monitoring site.

concentrations increased from mean values between 2.0 and 2.9 mg/l in water years 1996 to 1998 to values greater than 10.2 mg/l in water years 2003 to 2005 (Figure 42). From WY1999 to WY2003, mean annual nitrate concentrations in SQW4 subbasin increased an average of 1.6 mg/l per year. In SQW5 subbasin, nitrate concentrations decreased slightly from water years 1996-1998 to WY2000-2002, then increased quite substantially in water years 2003 to 2005. Mean annual concentrations at SQW5 increased from 5.1 mg/l in 2000 to 15.1 mg/l in 2005 (Table 17).

In Walnut Creek subbasins, decreasing nitrate concentrations were observed in all

Table 17. Summary of mean annual nitrate, chloride and sulfate concentrations at 10 project monitoring sites.

	Water	Mean Annual Concentration (mg/l)									
Anion	**Year**	**WNT1**	**WNT2**	**WNT3**	**WNT5**	**WNT6**	**SQW1**	**SQW2**	**SQW3**	**SQW4**	**SQW5**
Nitrate	1996	10.6	7.8	12.2	11.3	6.4	12.6	8.5	10.6	2.9	8.1
	1997	11.4	8.3	11.9	10.4	7.1	12.4	8.2	11.0	2.0	8.2
	1998	12.7	9.5	12.0	11.0	8.8	13.8	11.5	12.3	2.9	9.3
	1999	10.9	8.2	8.9	9.6	8.1	11.4	9.2	10.8	4.5	7.9
	2000	10.0	6.9	8.4	8.6	5.7	11.6	8.2	9.0	6.8	5.1
	2001	10.7	7.1	8.5	8.3	5.5	11.9	9.5	10.3	8.4	5.5
	2002	10.8	6.8	8.8	7.7	3.1	10.5	8.6	10.0	9.4	6.8
	2003	10.7	7.1	8.8	8.6	3.9	12.7	9.9	10.9	10.9	11.7
	2004	12.3	8.1	8.0	8.4	5.0	13.5	10.9	11.0	10.2	13.9
	2005	11.2	8.3	10.1	10.2	5.5	13.0	10.0	11.4	10.9	15.1
	average	11.1	7.8	9.7	9.4	5.9	12.3	9.5	10.7	6.9	9.2
Chloride	1996	15.5	13.2	12.2	10.2	12.4	24.7	15.5	19.3	9.9	16.6
	1997	16.2	13.6	12.8	11.8	14.8	16.0	16.9	18.1	9.4	17.5
	1998	17.1	12.7	13.2	9.4	12.1	14.7	16.6	17.2	9.4	16.2
	1999	13.6	10.5	12.5	9.5	9.3	12.1	13.4	14.5	11.5	13.5
	2000	13.9	11.3	10.9	9.1	9.0	12.6	14.1	15.8	13.9	18.6
	2001	16.1	13.3	12.5	9.5	10.0	13.7	15.6	17.2	13.4	18.8
	2002	18.0	13.4	7.3	9.0	10.2	16.1	17.8	20.3	16.2	21.4
	2003	20.0	12.7	11.4	9.2	9.4	15.5	17.7	17.5	15.6	21.5
	2004	16.7	12.3	13.5	9.2	9.6	15.3	17.8	19.9	14.1	16.5
	2005	20.1	12.8	13.2	9.6	10.6	15.8	15.8	18.6	12.5	14.8
	average	16.7	12.6	12.0	9.6	10.7	15.7	16.1	17.8	12.6	17.5
Sulfate	1996	23.7	28.7	22.8	22.8	18.0		30.8	30.5	24.9	31.9
	1997	22.8	26.0	21.3	20.1	17.0	23.7	29.3	28.4	22.0	28.8
	1998	20.7	23.1	20.2	18.0	12.6	19.9	25.3	23.4	17.6	22.6
	1999	17.1	19.6	18.7	16.5	11.8	18.5	23.4	22.7	31.9	21.0
	2000	18.3	21.1	17.3	16.4	11.3	19.4	24.9	24.7	33.5	25.7
	2001	18.1	22.4	17.4	16.9	12.2	18.4	24.3	26.9	30.5	25.2
	2002	17.6	20.5	14.1	16.6	10.1	18.8	25.1	25.8	33.5	23.8
	2003	16.1	20.3	14.5	15.3	8.6	16.6	21.7	20.8	28.3	22.6
	2004	17.8	19.5	17.9	14.2	9.9	17.2	21.9	24.6	25.2	23.4
	2005	15.0	20.2	14.9	13.5	9.4	15.7	19.6	19.7	19.7	19.9
	average	18.7	22.1	17.9	17.0	12.1	18.7	24.6	24.7	26.7	24.5

subbasins from water year 1998 to 2004, and then a higher mean annual value was noted in Water Year 2005 (Figure 43). Mean annual concentrations decreased to a low value of 8.0, 7.7 and 3.1 mg/l in subbasins WNT3, WNT5 and WNT6, respectively. Concentrations in these subbasins increased to 10.1, 10.2 and 5.5 mg/l in Water Year 2005 (Table 17). However, Water Year 2005 was also characterized by dry conditions in the late summer and fall that dried up tributary streams and prevented water samples from being collected in August and September. Thus, average nitrate concentration values in 2005 may have been weighted higher than normal without collection of nitrate data from low nitrate months of August and September.

Monthly nitrate concentrations exhibited clear seasonality, with higher concentrations occurring during May, June and July (Figure 44). Nitrate concentrations were typically lowest in August, September and October when stream flow was also at a minimum and biological uptake of nitrogen was particularly evident. Although sampling frequency was limited during November and December, concentrations appeared to increase in the late fall. This pattern is consistent with long-term monthly nitrate concentrations in the Raccoon River (Schilling and Lutz, 2004).

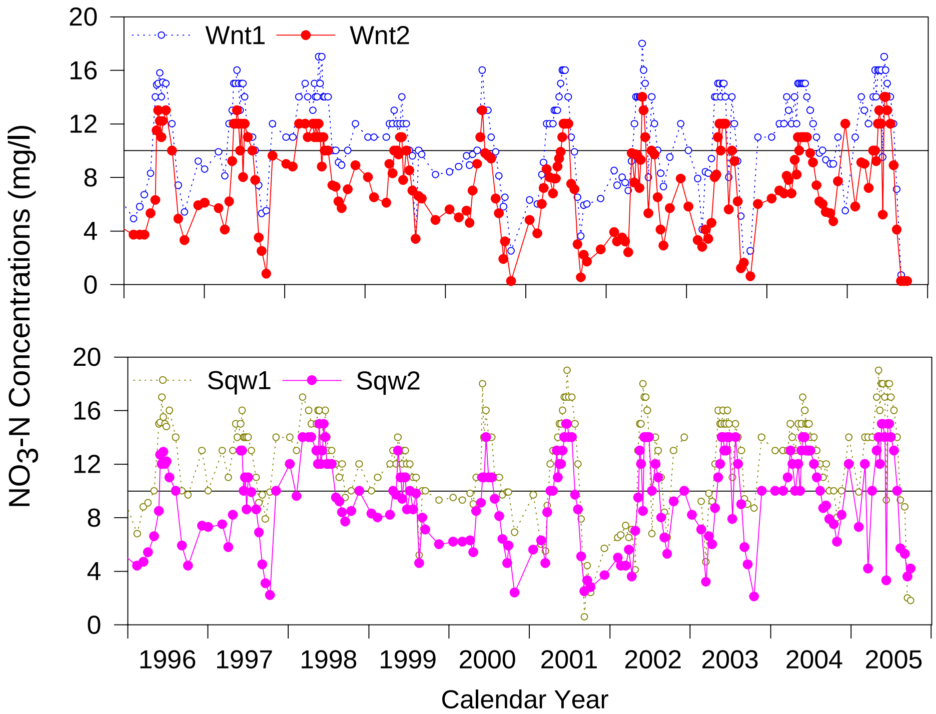


Figure 39. Time series of nitrate concentrations measured at upstream/downstream sites in Walnut and Squaw creek watersheds.

Chloride and Sulfate Concentrations

Mean chloride concentrations were higher at SQW2 compared to WNT2 averaging 16.2 and 12.6 mg/l, respectively, but were relatively similar at upstream sites WNT1 and SQW1 (16.9 and 15.3, respectively) (Table 17). Chloride concentrations ranged from 4.6 to 98 mg/l in Walnut Creek watershed and from 4.2 to 42 mg/l in Squaw Creek watershed, although in general, data are clustered between 10 and 20 mg/l (Figure 45). Occasional high values at WNT1 may be associated with road conditions on Highway 163 that crosses Walnut Creek near the WNT1 sampling site. Several elevated chloride concentrations detected at WNT1 were observed in January when road salt is typically applied to highways. Similarly, elevated values detected throughout the year may be influenced by road salt that may have accumulated along the bridge right-of-way.

Mean chloride showed a downstream decrease in concentration from WNT1 to WNT2, but mean chloride concentrations increased or were similar from upstream to downstream sites in Squaw Creek (Table 17). Lower mean chloride concentrations were measured in Walnut Creek subbasins compared to the main stem, with lowest average Cl concentration detected at WNT5 (9.6 mg/l). In Squaw Creek watershed, Cl concentrations were similar among the subbasin sites and main stem site, with lowest values observed in SQW4 subbasin. Like nitrate, mean annual concentration of chloride noticeably increased in SQW4 over the 10-year monitoring project, ranging from 9.4 to 9.8 mg/l in water years 1996 to 1998 to 16.2 mg/l in Water Year 2002 (Table 17).

Mean sulfate concentrations were lower in the Walnut Creek watershed (means ranging from 16.6 to 21.8 mg/l) compared to the Squaw Creek watershed (means ranging from 18.6 to 26.8 mg/l) (Table 17). Sulfate concentrations typically ranged between 10 and 30 mg/l (Figure 46). However, a decreasing trend in sulfate concentrations was observable at all sampling sites over the 10-year monitoring period (Table 17; Figure 46). At all watershed sampling sites, the lowest mean annual sulfate

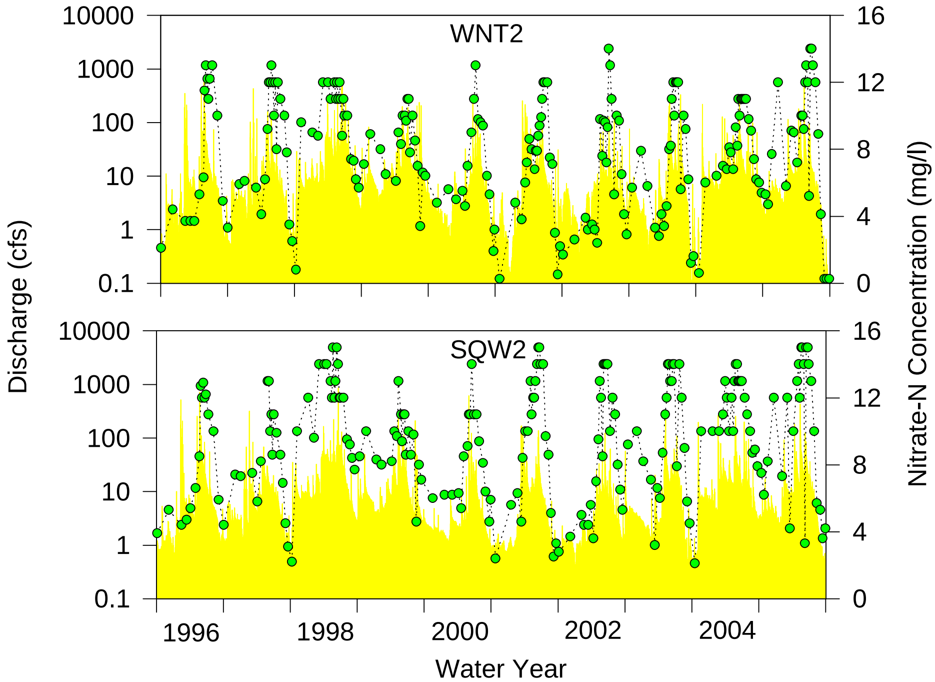


Figure 40. Relation of nitrate concentrations to discharge at WNT2 and SQW2.

concentration was measured during Water Year 2005. In most cases, the mean value in 2005 was 8-10 mg/l lower than the mean value in 1996. Less decrease in mean annual sulfate concentration was observed in subbasin SQW4, where sulfate concentration increased during the mid-sampling period before decreasing in 2005. The reason for the decrease in sulfate concentration over time is unknown. If not for the differing SQW4 sulfate pattern, a systematic cause may be suspected such as sampling or analytical error. However, SQW4 was sampled and analyzed in the same manner as all other sites but did not show the same decreasing annual pattern. A natural cause for the decreasing sulfate concentrations over time is unknown at this time.

N-Cl Ratios

Decrease in nitrogen concentrations between upstream and downstream stations observed in both watersheds can be caused by biological uptake, denitrification, or dilution by water lower in nitrogen. Ratios of upstream to downstream samples for chloride and nitrate can be used to clarify which of these processes contribute to concentration differences. Ratios of one indicate no in-stream change in concentration between upstream and downstream stations, whereas ratios greater or less than one indicates additional inputs or reductions. In both Walnut and Squaw Creeks, nitrate-N ratios are less than one, suggesting in-stream reductions caused by denitrification and biological processes (Figure 47). However, in Walnut Creek, chloride ratios are also less than one, suggesting that inputs of both nitrate-N and chloride are reduced in this watershed. Reduced chloride inputs may be associated with decreased use of potassium chloride (KCl) fertilizer in the watershed or possibly dilution from other water sources (surface water or groundwater) with low chloride concentrations.

Chemical Loads

Total export of nitrate, chloride and sulfate from Walnut Creek (WNT2) was lower than Squaw Creek (SQW2) (Table 18). Losses averaged 22.0 and 26.1 kg/ha of nitrate, 28.8 and 36.6 kg/ha of chloride and 48.4 and 54.7

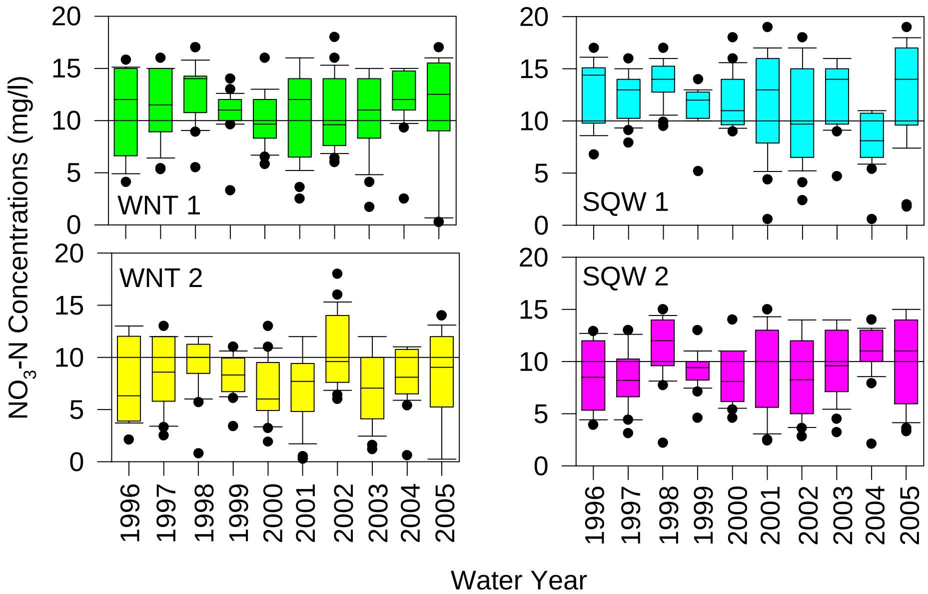


Figure 41. Box plot of nitrate concentrations by water year at upstream/ downstream sites in Walnut and Squaw creek watersheds.

kg/ha of sulfate, respectively. Annual losses of nitrate were greatest in 1998 when they ranged from 51.1 (WNT2) to 68.5 kg/ha (SQW2), whereas losses were smallest in 2002, ranging from 10.4 to 10.6 kg/ha. Differences in nitrate and chloride losses between Walnut and Squaw creek watersheds are due to substantially lower losses emanating from the lower portion of Walnut Creek watershed (WNT2-WNT1) (Table 18). Nitrate and chloride losses from upper Walnut Creek (WNT1) were similar to slightly higher than Squaw Creek, but losses from lower Walnut Creek were substantially less. Average annual loss of nitrate from upper Walnut Creek was nearly double the loss from lower Walnut Creek (Table 18). Chloride losses were 40 percent lower in lower Walnut Creek compared to upper Walnut Creek. Few consistent differences are noted in sulfate loads among watershed areas (Table 18). Mean sulfate losses were highest in Squaw Creek and higher in lower Walnut Creek compared to upper Walnut Creek, ranging between 43.7 to 54.7 kg/ha among all sites.

Regardless of watershed area, losses of nitrate, chloride and sulfate were all greatest in years with higher precipitation and streamflow (Table 18). However, the relation between chemical loss rates and discharge varied among watersheds areas and constituent (Figure 48). Annual chemical loss was plotted against streamflow (in) from each watershed area to assess the relation between loss rates and streamflow. Although these relations are clearly a function of the ESTIMATOR model to predict chemical losses from the streamflow record, the results are nonetheless informative by providing a means to estimate average chemical losses from the streamflow record alone. Hence, for every inch of streamflow in Walnut Creek (WNT2), an average loss of 2.1 kg/ha would be predicted by the ESTIMATOR model. In contrast, nitrate losses of 2.7 to 3.2 kg/ha would be expected for every one inch of streamflow in upper Walnut Creek and Squaw Creek. Only 1.4 kg/ha nitrate loss would be predicted for every one inch of streamflow in lower Walnut Creek (Figure 48). Similar relations developed for chloride and sulfate reveal similar chloride loss rates to nitrate, but all watershed areas exhibit variable unit loss values for sulfate. For every one inch of streamflow measured at WNT2, SQW2, WNT1 and WNT2-1, chloride losses would be 4.7, 5.9, 6.6 and 3.8 kg/ha, respectively. Sulfate losses would be 3.3, 6.0, 1.1 and 4.5 kg/ha, respectively.

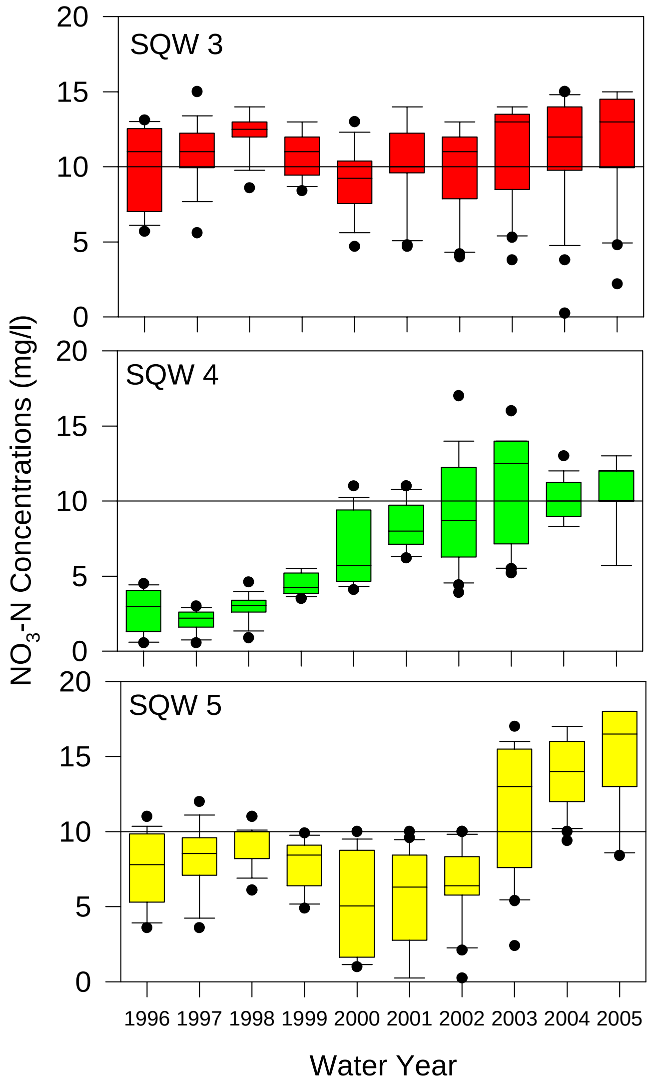


Figure 42. Box plots of nitrate concentrations by water year at Squaw Creek subbasin sites.

Seasonal nitrate losses followed discharge and concentration patterns with greatest loss occurring in May when export of nitrate exceeded 30 Mg/month (Figure 49). Nitrate losses exceeded 20 Mg in June and 5 Mg in February, March, April and August each year. Due to low discharge, comparatively little nitrate export occurred in fall and winter (Figure 49).

Flow-weighted concentrations followed a similar pattern exhibited by chemical mass losses (Table 19). Flow-weighted concentrations of NO_3-N and Cl were higher in Squaw Creek and upper Walnut Creek than lower Walnut Creek. Average flow-weighted concentration of NO_3-N were 8.6 mg/l in Squaw Creek and 10.4 mg/l in upper Walnut Creek but was 4.9 mg/l in lower Walnut Creek (Table 19). Similarly, concentrations of Cl were 16.5 mg/l in Squaw Creek and 16.9 mg/l in upper Walnut Creek but 10.5 mg/l in lower Walnut Creek. Sulfate concentrations were slightly higher at SQW2 compared to WNT2, and concentrations were higher in lower Walnut Creek compared to upper Walnut Creek (Table 19). Discharge of groundwater from Pennsylvanian bedrock in lower reaches of Walnut Creek is believed to contribute to SO_4 differences within the Walnut Creek watershed (Schilling and Wolter, 2001). In general, average flow-weighted concentrations compared favorably to the mean of all analyses measured during the monitoring period (Table 12).

Trends

Regression Model Development

Anion concentrations data were examined for values below the detection limit. The detection limit for nitrate was <0.25 mg/l and no values were recorded below this value. Although nitrate concentrations did not fit a normal, nor lognormal distribution particularly well, a lognormal distribution underestimated the frequency of high values. Thus, no transformation was performed on the nitrate data. Chloride and sulfate concentrations were slightly more log normal than normal. However, since they were used primarily as an explanatory variable, it was decided not to perform a log transformation.

The anion concentration data were analyzed as a time series. Autocorrelation time series analysis was performed on the anion concentration and discharge data for the main stem and subbasin sites. Autocorrelation was stronger for nitrate than chloride and sulfate, ranging from 0.60 to 0.86 for nitrate, 0.34 to 0.73 for chloride and 0.43 to 0.68 for sulfate.

Similar autocorrelation was seen in log discharge at WNT2 (0.72) and SQW2 (0.84). It should be noted that discharge data evaluated in this section of the report only refers to discharge measured on the day of sampling, not the entire data set. Discharge measured on the day of sampling was needed to serve as an explanatory variable for evaluating anion concentrations measured on that day.

It was evident from the monitoring program that nitrate concentrations and stream discharge data exhibits strong seasonality, with peaks occurring in May/June and late fall. To account for the seasonality in the statistical trend analysis, a seasonal adjustment was made to account for differences between each month. The least square means (LSMEANS) was calculated for each season (e.g., the average value adjusted for each month evaluated on a comparable basis). This was not the average value for each month because it accounted for differing sample frequencies and adjusted for any trends that may have occurred over time. A separate test was subsequently performed to determine if there was a significant interaction between date and month (was there evidence that the relationship over time was different for different months). There was no statistical evidence that trends were different for various seasons.

Overall, the effect of adding the seasonal component was significant for most variables and stations. The seasonal signal was strong for nitrate, discharge and sulfate and weakest for chloride. In fact, chloride at WNT6 and SQW5 did not have statistically different seasonal signals after adjusting for any possible linear trend.

Downstream stations for both Walnut and Squaw had lower concentrations of nitrate compared to their upstream stations whereas total discharge, baseflow discharge, chloride and sulfate concentrations were higher at downstream stations. The linear regression between downstream concentrations to upstream concentrations was significant for all variables. The slope was positive – as the concentration

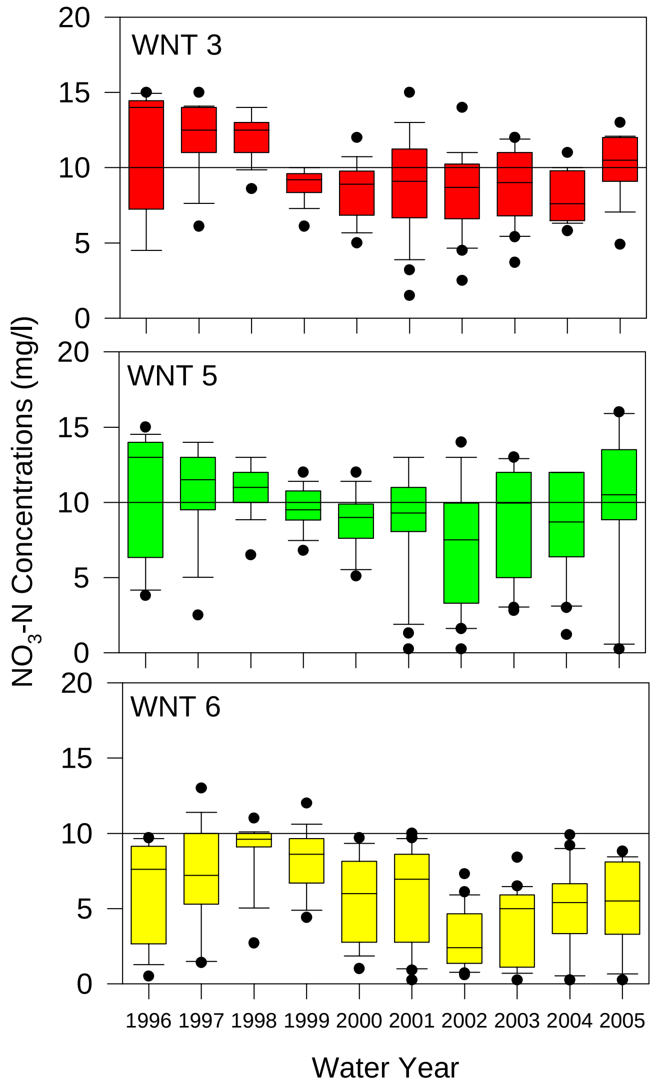


Figure 43. Box plots of nitrate concentrations by water year at Walnut Creek subbasin sites.

increases for the upstream station, so does the downstream station. The correlation between downstream concentrations was strongest for nitrate in both Walnut and Squaw creeks and discharge in Walnut Creek watershed ($r^2 > 0.78$ and slopes approaching 1) (Figure 50). Although there was a statistically significant relationship between downstream and upstream chloride in Walnut Creek, it was weak and explained very little of the variability in the downstream concentrations ($r^2 = 0.05$). In contrast, the relationship between downstream

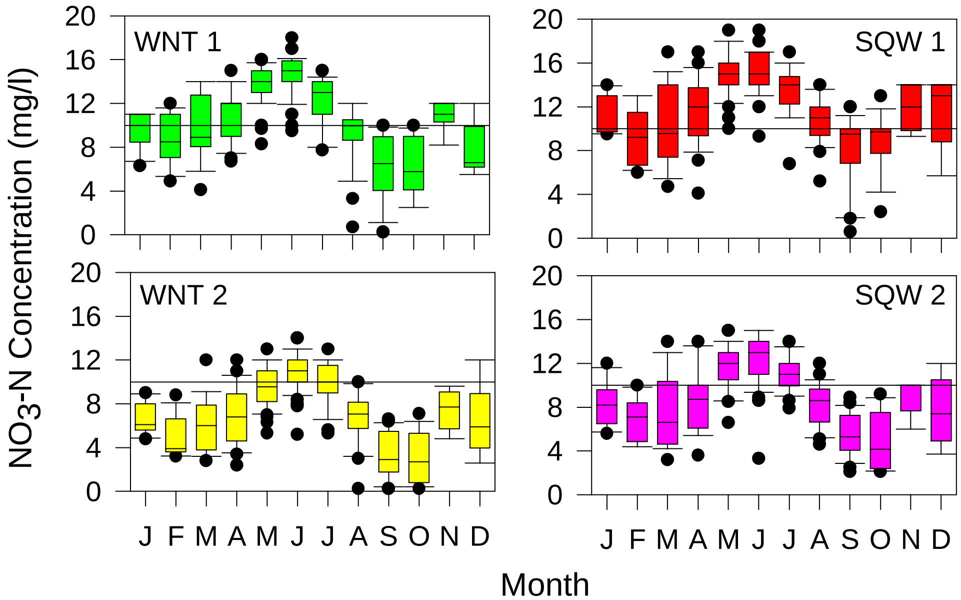


Figure 44. Box plots of nitrate concentrations by month at upstream/ downstream sites in Walnut and Squaw creek watersheds.

and upstream chloride was much stronger in Squaw Creek compared to Walnut Creek (r^2 = 0.67). The relation between downstream to upstream sulfate concentrations was better in Walnut Creek (r^2 = 0.31) than Squaw Creek (r^2 = 0.17).

Regression relationships were examined between the WNT2 (treatment) and SQW2 (control) stations. The regression for nitrate was highly significant (Figure 50). The nitrate concentrations were slightly lower in the Walnut Creek watershed compared to the Squaw Creek watershed, averaging 0.71 mg/l lower over the entire sampling period. Interestingly, the nitrate concentration in the upstream Walnut Creek watershed (WNT1) was very similar to the concentrations in the upstream Squaw Creek (SQW1) station (slope almost 1, 0 intercept, and r^2 = 0.78). Similarly, total discharge and base flow discharge were significantly related at WNT2 and SQW2. The slope between the discharge values was near 1.0 and the intercept was not statistically different from 0.0 (r^2 = 0.87 for both discharge and baseflow). This implies that the discharge values from the downstream Walnut Creek and Squaw Creek watershed at the time of sampling were very similar. The regressions were also statistically significant for chloride and sulfate with positive slopes (if concentrations increased in the control watershed, they increased in Walnut). The r^2 was moderate for sulfate (0.50) but low for chloride (0.28).

The best relationship with discharge was with nitrate for both Walnut Creek and Squaw Creek watersheds with moderately high r values (r = 0.65 and 0.68, respectively) (Figure 50). Overall, nitrate had a stronger relationship with baseflow as compared to mean daily discharge, with baseflow correlation coefficients at WNT2 (0.73) slightly lower than at SQW2 (0.77). Relations were positive, that is, as discharge increased, so did nitrate concentrations. Interestingly, the best relationship was with the upstream stations baseflow discharge as compared to the downstream – however, the goodness of fit was not significantly different. For consistency, it was decided to utilize the downstream baseflow as the explanatory variable or covariate for detection of trends for nitrate. The relationship between chloride concentrations and discharge was statistically significant, but moderate in strength (r = -0.40 at WNT2). The correlation

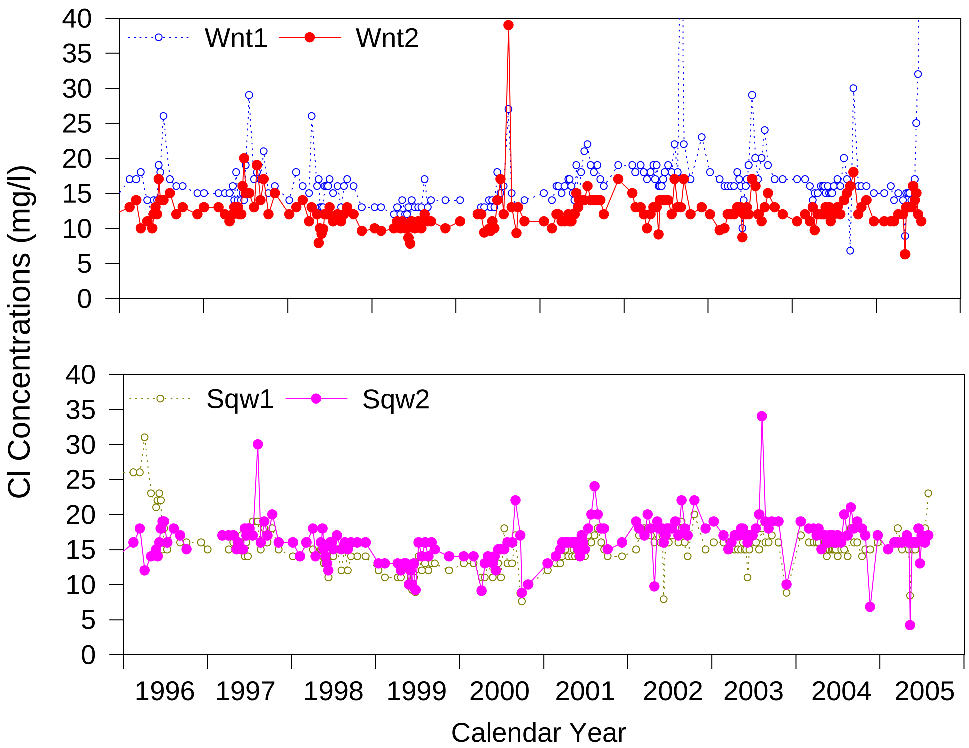


Figure 45. Time series of chloride concentrations measured at up-stream/downstream sites in Walnut and Squaw creek water-sheds.

coefficients were negative (as discharge increased, chloride concentrations decreased). Overall, slightly stronger relationships were seen with baseflow as compared to mean daily discharge (r = -0.43 at WNT2). Therefore, where appropriate, baseflow was used as the covariate. Similar relations were seen with sulfate (negative correlation). The correlation of sulfate with discharge was moderate for stations WNT2, SQW1, SQW2, SQW3, and SQW5 (r >0.45), but weak at other sites. Mean daily discharge was only slightly better correlated than baseflow for sulfate.

Final Multiple Linear Regression Model to Test for Changes Over Time

All the multiple regression models have date as a trend variable. The addition of covariates that were shown to have significant relationships with WNT2 conditions were examined. From the previous sections, it was determined that for most stations and variables the important covariates include season, discharge, upstream concentrations, and control watershed. These were variables for which data were collected as part of the experiment design of the project and have a mechanistic basis for inclusion in the models. Since the four covariates are themselves highly correlated, it is possible that the addition of all covariates is not the best model. The appropriate covariates were added to the model in the order of their individual significance and were only retained if they added significant information beyond the other covariates.

For the downstream station in Walnut Creek (WNT2) the best set of covariates or explanatory variables was:

- MONTH (“Season”)
- WNT1 (upstream nitrate concentration)
- SQW2 (downstream control watershed nitrate concentrations)

The results of the trend analyses and a summary of the covariates used are shown in Table 20. Although an adjustment for season

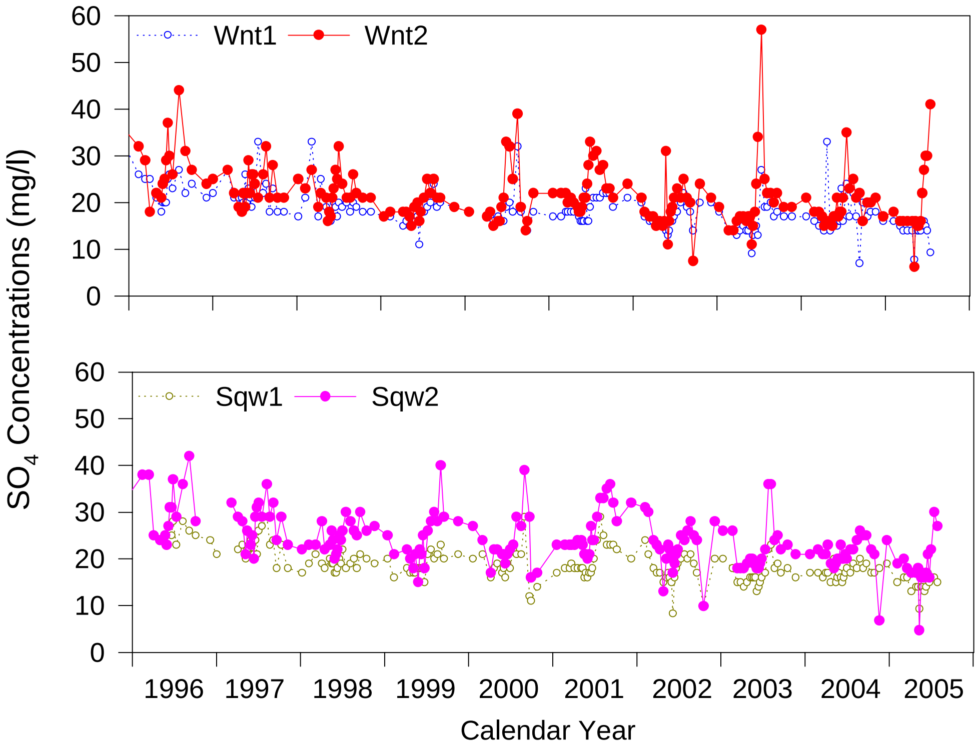


Figure 46. Time series of sulfate concentrations measured at upstream/downstream sites in Walnut and Squaw creek watersheds.

and baseflow discharge at WNT2 alone did not indicate a statistical trend, the addition or either the paired site or the upstream concentration did result in a statistically significant decrease over the 10-year sample time period. Baseflow discharge explained a significant amount of variability in nitrate at WNT2 (a regression with Date, month, and WNT2Qb), but baseflow discharge became non-significant when upstream WNT1 concentrations were added to the trend model (r^2=0.85) because the upstream concentration was highly correlated with discharge. The addition of the paired downstream Squaw site (SQW2) was a significant covariate. Since the nitrate concentrations increased in the Squaw downstream site, two trend models are provided in Table 20 (one with the control watershed concentration covariate and one without).

For the Walnut Creek outlet (WNT2), the trend analysis indicates that nitrate concentrations decreased 0.12 mg/l/year or 1.2 mg/l over the 10-year project period when the Squaw Control watershed was utilized as a covariate. Without adjusting for the control, the decrease was 0.7 mg/l over the 10-year period. Interestingly, without the upstream covariate, there was no significant trend in nitrate at WNT2. There was an increase in WNT1 nitrate.

Because the nitrate concentrations were not log transformed, a percent change would vary depending upon the magnitude of the initial concentration. Therefore an estimated change over the entire 10-year period was given. To assess what this change means for each month, the estimated mean values at the project beginning and project end are given for each

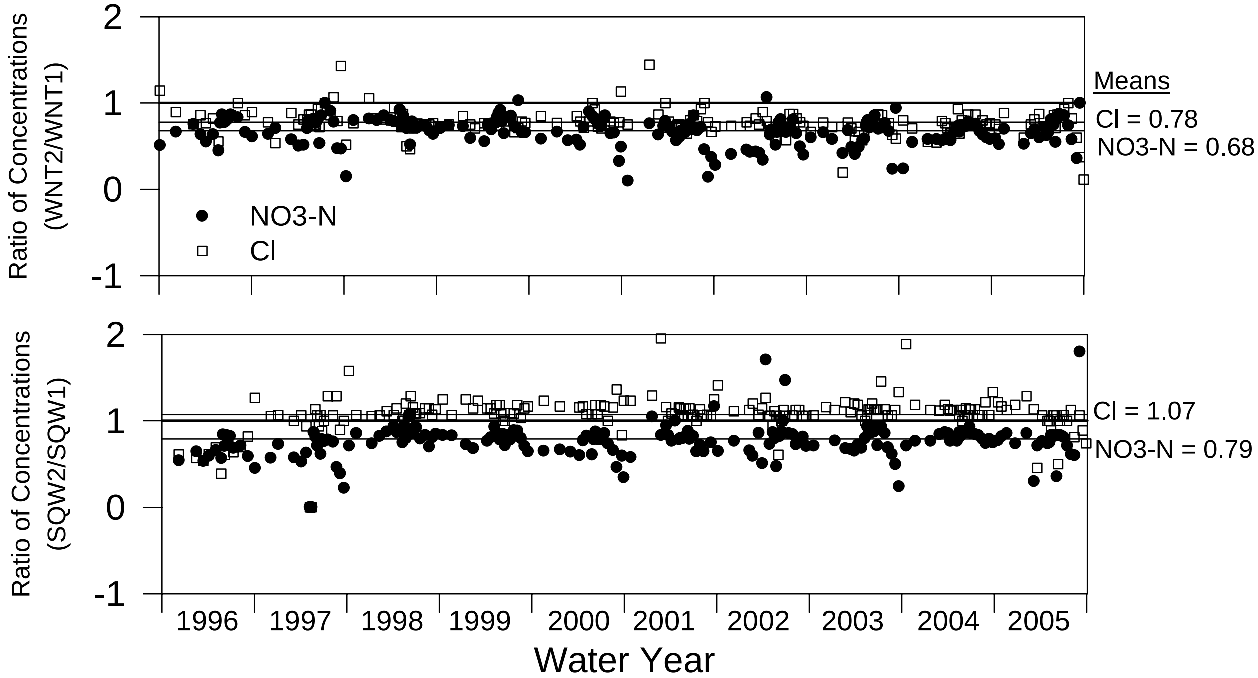


Figure 47. Ratios of nitrate and chloride concentrations at upstream to downstream sites.

month for WNT2. These values were evaluated using the LSMEANS values of the covariate to allow for equable comparisons. The concentrations do not represent the actual values observed, but instead represent a regression estimate of what the actual average values would have been IF the other covariates were at their monthly estimated values. This technique allows for the comparison of “apples to apples” and adjusts for natural variation (Table 21). With this approach, the greatest percentage decrease in monthly nitrate concentrations occurred during September and October.

The magnitude of change was also estimated for each of the subbasins. (Table 20). The decrease in nitrate concentrations for each subbasin was significant and of greater magnitude compared to the downstream WNT2 station. Nitrate concentrations decreased 3.4, 1.2 and 2.7 mg/l at WNT3, WNT5 and WNT6, respectively. Nitrate concentrations increased 1.9 mg/l over 10 years in the downstream Squaw stations SQW2 and 1.1 mg/l over 10 years in the upstream Squaw stations SQW1. All subbasins in the Squaw Creek increased in nitrate concentrations, with subbasins SQW4 and SQW5 having quite dramatic increases. Over the 10-year monitoring program, nitrate in surface water in SQW4 and SQW5 subbasins increased 11.6 and 8.0 mg/l, respectively. The magnitude of increase in the Squaw Creek subbasins was considerably greater than the decrease measured in Walnut Creek.

Discussion

Results from surface water monitoring in Walnut and Squaw creek watersheds for nitrate, chloride and sulfate indicate that changes in anion concentrations have occurred during the 10-year monitoring program. Nitrate concentrations significantly decreased in Walnut Creek watershed, both at the watershed outlet and in monitored subbasins. On the other hand, nitrate concentrations significantly increased in

Table 18. Discharge and loss of anions, herbicides and phosphorus from various watershed areas.

Site	Water Year	Discharge (in)	Baseflow (in)	Chemical Loads Nitrate-N (kg/ha)	Chloride (kg/ha)	Sulfate (kg/ha)	Atrazine (g/ha)	DEA (g/ha)	Phosphorus (kg/ha)
WNT2	1996	10.60	4.71	25.66	34.21	68.89	4.31	0.37	
	1997	7.43	3.42	18.32	25.04	48.20	1.17	0.19	
	1998	16.61	9.69	41.57	51.11	92.01	5.22	1.01	
	1999	10.37	6.48	26.38	33.37	57.77	1.48	0.50	
	2000	4.63	3.20	11.59	15.04	25.55	0.90	0.29	
	2001	8.91	5.35	20.70	27.94	44.50	0.80	0.28	1.16
	2002	4.31	2.55	10.64	14.45	22.75	0.70	0.23	0.35
	2003	8.10	4.74	19.61	24.68	36.69	3.00	0.65	1.34
	2004	9.77	5.86	24.70	32.78	46.85	2.26	0.43	1.10
	2005	8.56	4.68	21.19	28.95	40.38	3.81	0.39	1.28
	Average	8.93	5.07	22.04	28.76	48.36	2.36	0.43	1.05
SQW2	1996	10.84	4.76	27.97	42.74	79.73	2.92	0.21	
	1997	5.83	3.74	15.51	25.43	45.89	0.55	0.08	
	1998	16.91	11.79	51.49	68.54	113.80	5.53	0.51	
	1999	8.71	5.95	26.54	37.47	59.66	1.40	0.23	
	2000	4.59	2.49	13.03	19.06	29.47	2.14	0.19	
	2001	7.07	5.08	22.20	29.76	43.86	0.99	0.20	1.00
	2002	3.36	2.23	10.39	15.20	20.98	0.63	0.09	0.23
	2003	6.94	4.96	23.08	30.53	39.03	1.27	0.21	0.65
	2004	11.79	6.94	38.30	52.03	62.92	1.76	0.32	1.61
	2005	10.15	5.53	32.93	45.25	51.81	1.77	0.24	1.41
	Average	8.62	5.35	26.15	36.60	54.72	1.90	0.23	0.98
WNT^1	1996	10.35	5.20	31.08	45.35	61.17	2.60	0.46	
	1997	6.43	3.91	21.96	28.45	38.82	0.62	0.24	
	1998	16.70	11.15	58.95	71.19	87.44	3.40	1.07	
	1999	10.73	7.30	39.09	46.11	55.36	1.21	0.66	
	2000	5.28	3.74	18.16	23.31	25.72	0.86	0.38	
	2001	7.71	5.47	26.71	35.35	36.20	0.54	0.42	0.79
	2002	4.84	3.13	17.36	21.68	22.00	0.66	0.34	0.32
	2003	7.32	5.01	26.42	32.92	31.45	1.09	0.51	0.59
	2004	9.26	6.27	33.05	42.66	40.17	1.09	0.52	0.79
	2005	8.92	5.28	30.48	42.38	38.31	1.34	0.45	0.81
	Average	8.75	5.65	30.33	38.94	43.66	1.34	0.50	0.66
WNT2-1	1996	10.73	4.47	22.91	28.57	72.70	5.16	0.32	
	1997	7.93	3.17	16.47	23.30	52.88	1.45	0.16	
	1998	16.56	8.96	32.80	40.98	94.22	6.13	0.97	
	1999	10.18	6.08	19.96	26.94	58.94	1.62	0.41	
	2000	4.30	2.92	8.28	10.87	25.44	0.92	0.24	
	2001	9.51	5.29	17.66	24.19	48.63	0.94	0.21	1.35
	2002	4.04	2.26	7.25	10.81	23.10	0.72	0.18	0.37
	2003	8.50	4.61	16.18	20.51	39.29	3.95	0.72	1.72
	2004	10.02	5.66	20.48	27.79	50.18	2.85	0.38	1.25
	2005	8.37	4.38	16.49	22.17	41.39	5.04	0.36	1.52
	Average	9.01	4.78	17.85	23.61	50.68	2.88	0.40	1.24

DEA = desethylatrazine

Squaw Creek watershed, again both at the watershed outlet and more significantly in subbasins. Evidence from both watersheds points to the effects of land use on watershed nitrate concentrations.

In Walnut Creek, prairie restoration and land management changes implemented at the Neal Smith NWR have reduced stream nitrate concentration and nitrate loading rates. In subbasins where land use changes have comprised a greater proportion of the watershed area, nitrate concentrations have decreased 1.2 to 3.4 mg/l in 10 years. In the entire watershed (WNT2), nitrate concentrations at WNT2 significantly decreased, though the rate of change was less than the subbasins. The rigorous statistical model developed to estimate the nitrate concentration reductions included explanatory factors to account for seasons, variable discharge, and changes occurring in other areas. Changing concentrations in Walnut Creek cannot be attributed simply to changing weather patterns since this factor was assessed by evaluating effects of seasons and discharge. In addition, the paired design of the study allowed for changes in Walnut Creek to be compared against conditions in other watershed areas to evaluate whether changes occurring in the treatment watershed were any different than changes occurring in the control watershed. Therefore, explanations for the decreasing nitrate concentrations in Walnut Creek watershed focused exclusively on land use change implemented by the USFWS at the Neal Smith NWR.

The decrease at WNT2 occurred despite an increasing trend in nitrate concentration at upstream site WNT1. Given the significant relation of nitrate between the upstream and downstream sites, dilution of stream water with lower nitrate concentration inputs must be occurring between the two sites to produce an otherwise decreasing trend at WNT2. Evidence from chemical loads support less nitrate contribution to streamflow from the lower portion of the Walnut Creek watershed containing the refuge. Losses of nitrate and

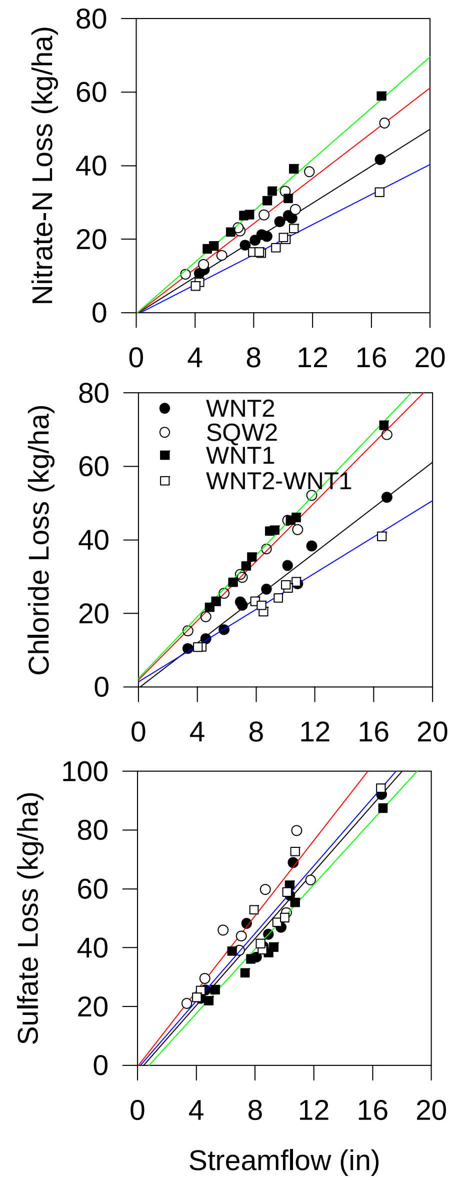


Figure 48. Relation of annual nitrate, chloride and sulfate losses to annual discharge at USGS gaging stations.

chloride were substantially less in lower Walnut Creek than upper Walnut Creek and Squaw Creek. Losses of nitrate and chloride from lower Walnut Creek were approximately one-half the mass lost from upper Walnut Creek and chloride losses were 40 percent lower. This was reflected in average flow-weighted nitrate and chloride concentrations that were more than 5

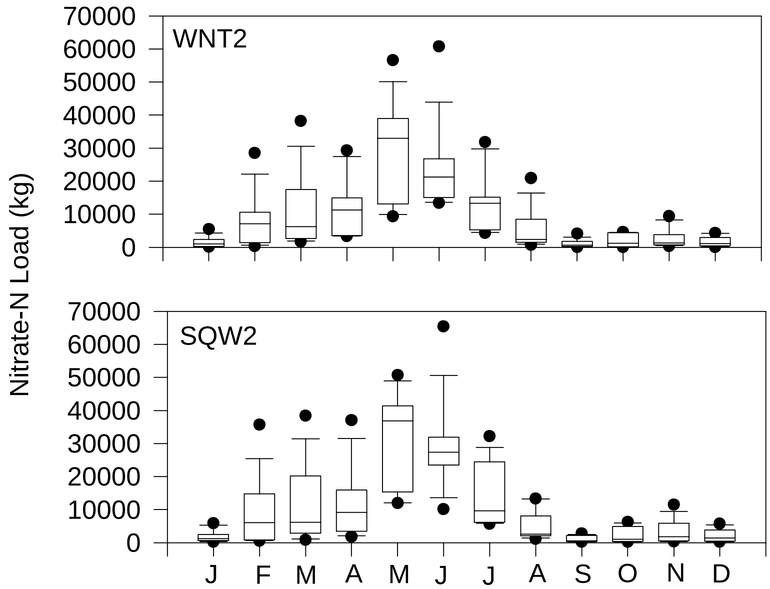


Figure 49. Box plots of monthly nitrate loads at WNT2 and SQW2.

to 6 mg/l less in lower Walnut Creek compared to upper Walnut Creek.

Estimates of chemical loading rates are consistent with synoptic sampling conducted in Walnut and Squaw Creek watersheds on two occasions in 1999 (Schilling and Wolter, 2001; Schilling, 2001) and 2001 (Schilling, 2002). In 1999, a baseflow survey of 81 tributary creeks and tiles showed major differences in pollutant loading rates within the Walnut Creek watershed. Concentrations of nitrate and chloride were lowest in creeks and tiles draining restored prairie areas (<1 and <3 mg/l, respectively) compared to concentrations of water draining row crop areas (>10 and >12 mg/l, respectively). Results indicated that nine headwater areas consisting of 90 percent row crop contributed more than half the total nitrate export from the watershed while comprising only one-third of the land area (Schilling, 2001). A second synoptic survey in 2001 that included both Walnut and Squaw creek watersheds revealed that water draining interior watershed areas of Walnut Creek containing the restored prairie had substantially lower nitrate and chloride concentrations than Squaw Creek and upper Walnut Creek watershed areas draining row crop areas (Schilling, 2002). Furthermore, the study completed during an October 2001 low flow period indicated that nitrate concentrations at that time decreased from upstream to downstream locations in Walnut Creek but increased in Squaw Creek. In data from both synoptic surveys, the portion of the Walnut Creek watershed containing the refuge clearly contributed less nitrate and chloride loads to the watershed. Hence, dilution of stream water with less nitrate coming from restored prairie and land management area is serving to reduce nitrate concentrations in Walnut Creek.

However, it is also evident that contributions of nitrate from upstream areas dominate the nitrate concentrations at the watershed outlet. Both Walnut and Squaw creeks had a substantial portion of their downstream nitrate concentrations explained by upstream values (r2 of 0.94 and 0.78, respectively). Evidence from the chemical load data and two synoptic surveys also indicates that headwater regions in Walnut Creek contribute a greater proportion of nitrate to the stream. Once nitrate is delivered to the stream network from row-crop dominated headwater regions, it is diluted somewhat by the downstream watershed area, but concentrations remain elevated.

Table 19. Estimated flow-weighted concentrations of anions, herbicides and phosphorus from various watershed areas.

		Flow-weighted Concentrations					
Site	Water Year	NO3-N (mg/l)	Cl (mg/l)	SO4 (mg/l)	Atrazine (ug/l)	DEA (ug/l)	P (mg/l)
WNT2	1996	6.93	13.77	30.42	0.49	0.06	
	1997	7.31	13.19	26.84	0.26	0.06	
	1998	8.22	12.13	22.37	0.46	0.11	
	1999	7.92	12.28	21.80	0.26	0.10	
	2000	6.24	12.96	23.67	0.27	0.10	
	2001	6.36	12.51	22.27	0.21	0.10	0.31
	2002	6.57	12.96	21.69	0.26	0.10	0.22
	2003	6.33	12.89	21.01	0.45	0.12	0.30
	2004	7.88	12.52	18.80	0.52	0.12	0.34
	2005	7.01	13.39	20.28	0.68	0.10	0.31
	Average	7.10	12.85	22.89	0.39	0.10	0.29
SQW2	1996	7.15	16.40	32.27	0.36	0.04	
	1997	8.03	16.35	30.05	0.22	0.04	
	1998	9.81	15.35	25.76	0.47	0.07	
	1999	9.33	16.03	26.12	0.27	0.07	
	2000	7.65	16.62	27.51	0.54	0.08	
	2001	7.72	16.88	27.04	0.26	0.07	0.31
	2002	7.61	17.33	26.41	0.26	0.06	0.23
	2003	9.04	16.97	23.21	0.26	0.07	0.25
	2004	10.19	16.34	20.32	0.30	0.08	0.42
	2005	9.36	17.23	20.49	0.26	0.06	0.31
	Average	8.61	16.54	25.90	0.32	0.06	0.30
WNT1	1996	9.03	16.78	24.59	0.34	0.09	
	1997	10.01	16.50	22.99	0.19	0.10	
	1998	12.28	15.39	20.01	0.35	0.16	
	1999	11.99	15.73	19.37	0.23	0.16	
	2000	9.55	17.19	19.51	0.23	0.15	
	2001	9.70	17.57	18.12	0.16	0.14	0.26
	2002	9.23	17.93	18.13	0.19	0.14	0.18
	2003	10.15	17.54	17.13	0.22	0.15	0.21
	2004	11.33	16.98	16.01	0.28	0.16	0.29
	2005	10.43	17.96	16.20	0.27	0.13	0.25
	Average	10.40	16.95	19.20	0.25	0.14	0.24
WNT2-1	1996	5.24	12.12	34.90	0.57	0.04	
	1997	5.63	11.56	29.40	0.30	0.04	
	1998	5.55	10.23	24.18	0.50	0.07	
	1999	5.61	10.41	23.24	0.28	0.06	
	2000	4.07	10.20	26.64	0.29	0.07	
	2001	4.74	10.02	24.68	0.24	0.08	0.33
	2002	5.11	10.71	23.16	0.33	0.09	0.25
	2003	2.94	9.19	24.97	0.55	0.09	0.33
	2004	6.15	10.06	20.81	0.66	0.10	0.36
	2005	4.16	10.57	23.18	0.94	0.07	0.33
	Average	4.93	10.49	25.49	0.47	0.07	0.32

DEA = desethylatrazine

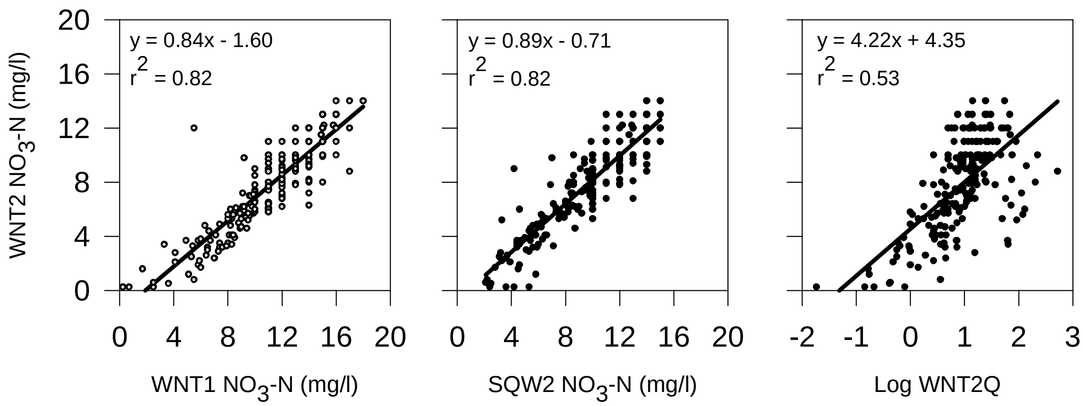


Figure 50. Relation of nitrate concentrations at WNT2 to regression covariates.

In Squaw Creek watershed, a different relation of nitrate concentrations to land use change emerged. Significant increases in nitrate concentrations were measured in two subbasins (>8 mg/l), and nitrate at Squaw Creek watershed outlet increased by nearly 2 mg/l during the 10-year project. In the two subbasins with increasing nitrate (SQW4 and SQW5), the amount of land in row crop increased by 26 and 29 percent, respectively, with a corresponding decrease in CRP grass land cover. Nitrate concentrations changed quite dramatically in SQW4 increasing by over 11 mg/l in the span of 10 years, with most of the change concentrated within a span of four years. Even in Walnut Creek, an increasing trend in stream nitrate concentrations is evident in upstream WNT1 where row crop in the watershed area increased by nine percent. It is unknown whether the increase in nitrate concentrations in these areas is attributed to increased fertilizer inputs or mineralization of organic N, but the influence of row crop land cover on stream nitrate concentrations is plainly evident.

Water quality changes in stream nitrate concentration from land use change were more easily measured in smaller watersheds. The rate of decrease in downstream Walnut Creek nitrate was less than the decreasing slope measured in smaller subbasins and the rate of increase in Squaw Creek was considerably greater in the subbasins than watershed outlet. The downstream watershed outlets (WNT2 and SQW2) integrated water contributions from a large landscape area and because of this, do not isolate areas of change particularly well. With headwater contributions of stream nitrate playing such an important factor in downstream nitrate concentrations, changes in stream nitrate concentrations at the watershed scale were easily obscured by upstream areas. When areas of land use change were isolated at the subbasin scale, substantially greater water quality changes were observed.

HERBICIDES

Concentrations

Atrazine and the atrazine breakdown product desethylatrazine (DEA) were the most frequently detected (>0.1 ug/l) herbicides during the monitoring project. In 158 water samples, atrazine was detected 71.5 percent of the time at both Walnut and Squaw creek outlets (WNT2 and SQW2), and DEA was detected at a frequency of 67.7 and 77.2 percent, respectively (Table 22). The median detected atrazine concentration at both watershed outlets was the same (0.28 ug/l), whereas the mean detected concentration was higher at WNT2 than SQW2 (1.28 vs. 0.94 ug/l, respectively). The median DEA concentration detected at

Table 20. Trend tests for changes in nitrate concentrations over time at project monitoring sites, adjusted for appropriate covariates as indicated.

Station	Covariates	Slope (mg/l/year) (Negative = decrease)	Prob.>t on slope estimate	r^2	Decrease over 10 years (mg/l)
WNT2	Season WNT1-nitrate SQW2-nitrate	- 0.119	<.0001	0.89	- 1.1
WNT2	Season WNT1-nitrate	- 0.066	<.0001	0.86	- 0.7
WNT1	Season Log(WNT1-Qb)	+ 0.116	.0166	0.73	+ 1.2
WNT3	Season Log(WNT2-Qb)	-0.340	<.0001	0.50	- 3.4
WNT5	Season Log(WNT2-Qb)	-0.116	.0820	0.66	- 1.2
WNT6	Season Log(WNT2-Qb)	-0.274	<.0001	0.57	- 2.7
SQW2	Season Log(SQW2-Qb)	+ 0.191	.0001	0.71	+ 1.9
SQW1	Season Log(SQW2-Qb)	+ 0.108	.0629	0.60	+ 1.1
SQW3	Season Log(SQW2-Qb)	+ 0.091	.0935	0.000	+ 0.9
SQW4	Season Log(SQW2-Qb)	+ 1.158	<.0001	0.635	+ 11.6
SQW5	Season Log(SQW2-Qb)	+ 0.797	<.0001	0.206	+ 8.0

Qb = baseflow

WNT2 and SQW2 was lower than the atrazine median and nearly the same for both watersheds (0.17-0.16 ug/l; Table 22). Despite the frequency of occurrence, atrazine was not often found above the USEPA MCL of 3 ug/l at WNT2 and SQW2, with seven samples exceeding 3 ug/l at both outlets (4.4 percent).

Atrazine concentrations at WNT2 and SQW2 were highly skewed, with concentrations ranging between <0.1 to 46 ug/l and <0.1 to 22 ug/l, respectively (Figure 51). DEA concentrations were less skewed, ranging between <0.1 to 1.5 ug/l at WNT2 and <0.1 to 1.9 ug/l at SQW2. Atrazine concentrations were highest during periods of high stream flow associated with rainfall runoff (Figure 52). Following peak events, atrazine concentrations decreased in the late summer and fall. The timing of peak concentrations in the late spring/ early summer with high streamflow events is consistent with the "spring flush" described by Thurman et al. (1991). Jaynes et al. (1999) reported similar patterns in Story County, Iowa at stream sites, county drain sites and field tile sites.

Both atrazine and DEA were detected at a higher frequency in upstream main stem sites WNT1 and SQW1, with atrazine detected at a

Table 21. Estimated nitrate concentrations (mg/l) at project start (WY1996) and after 10 years (WY2005) for each month for downstream Walnut Creek station WNT2. Concentrations have been adjusted for the mean values of the covariates for each month and reflect estimated values from the predictive regression multivariate models.

Month	1995 Nitrate Estimated Adjusted Concentration	2005 Nitrate Estimated Adjusted Concentrations	Estimated decrease over 10 years (mg/l)	% Decrease
January	7.14	5.95	1.19	-16.7
February	5.76	4.57	1.19	-20.6
March	6.83	5.64	1.19	-17.4
April	7.53	6.34	1.19	-15.8
May	10.18	8.99	1.19	-11.7
June	11.82	10.63	1.19	-10.1
July	10.32	9.13	1.19	-11.5
August	7.20	6.01	1.19	-16.5
September	4.20	2.91	1.19	-29.0
October	3.79	2.60	1.19	-31.4
November	8.04	6.85	1.19	-14.8
December	7.18	5.99	1.19	-16.6

frequency of 76.6 and 80.4 percent, respectively (Table 22). DEA was commonly detected in water samples collected at SQW1 and WNT1 (90.5 and 81.0 percent, respectively). Median and mean atrazine and DEA concentrations in upstream sites were similar to downstream sites. Atrazine concentrations ranged between <0.1 to 15 ug/l at WNT1 and between <0.1 to 7.7 ug/l at SQW1. In the subbasins, atrazine was also frequently detected, ranging from 59.8 to 75.8 percent in Walnut Creek and 50.8 to 81.1 percent in Squaw Creek, with median detected concentrations falling within a narrow range among all sites (0.20 to 0.26 ug/l; Table 22). Atrazine concentrations ranged between <0.1 to 29 ug/l in Walnut Creek subbasins and between <0.1 to 53 ug/l in Squaw Creek subbasins (Figures 53 and 54). With exception of SQW4 and SQW5, DEA was detected at a frequency of 66.7 to 78.0 percent in other subbasin sites with median concentration ranging between 0.13 to 0.19 ug/l. DEA was only detected in 15 out of 132 samples collected at SQW4 (11.4 percent) and a third of the samples collected at SQW5. Maximum DEA concentration in either watershed subbasin was 3.6 ug/l at WNT6 (Table 22).

In contrast to atrazine and DEA, other herbicides were not detected as frequently (Table 22). Among all sites, acetochlor was found at a frequency ranging between 7.6 percent at WNT3 to 27.7 percent at SQW2, with median detected concentrations ranging between 0.16 ug/l at WNT5 to 0.37 ug/l at SQW4. A second atrazine breakdown product, desisopropylatrazine (DIA) was found much less often than DEA, with detection frequencies ranging between 3.0 to 30.3 percent. Metolachlor was typically detected in less than 10 percent of all samples, although when detected, its median concentration was higher than most, ranging between 0.17 to 1.13 ug/l among all sites. Alachlor, butylate and metribuzin were rarely detected with 10 detections of these compounds in 2216 total samples over the 10-year monitoring period.

Cyanazine detections changed markedly during the project (Figure 55). While detected often during the 1996 to 1998 period, cyanazine

Table 22. Summary of herbicide detections and concentrations at project monitoring sites for water years 1996 to 2005. All concentrations in ug/l.

Site	n		Acetochlor	Alachlor	Atrazine	Butylate	Cyanazine	Deethyl atrazine	Desisopropyl atrazine	Metolachlor	Metribuzin
WNT2	158	mean	1.16		1.28		0.42	0.20	0.14	0.54	
		st. dev.	3.08		5.32		0.54	0.16	0.08	0.82	
		median	0.21		0.28		0.27	0.17	0.12	0.11	
		# detects	35	0	113	0	20	107	12	5	0
		det.freq.	22.2%		71.5%		12.7%	67.7%	7.6%	3.2%	
		max	18		46		2.5	1.5	0.39	2	
WNT1	158	mean	0.44		0.54		0.47	0.20	0.14	0.51	
		st. dev.	0.62		1.47		1.01	0.13	0.04	0.27	
		median	0.19		0.25		0.17	0.18	0.12	0.66	
		# detects	26	0	121	0	25	128	18	3	0
		det.freq.	16.5%		76.6%		15.8%	81.0%	11.4%	1.9%	
		max	2.5		15		5	1.3	0.27	0.66	
WNT3	132	mean	0.40		0.29		0.26	0.14	0.10		
		st. dev.	0.44		0.26		0.35	0.05	0.01		
		median	0.26		0.20		0.13	0.13	0.10		
		# detects	10	0	79	0	11	88	40	0	0
		det.freq.	7.6%		59.8%		8.3%	66.7%	30.3%		
		max	1.6		1.6		1.3	0.33	0.17		
WNT5	132	mean	0.46		0.72	0.15	0.29	0.19	0.18	0.75	
		st. dev.	0.84		2.91		0.34	0.20	0.16	1.64	
		median	0.16		0.26	0.15	0.16	0.16	0.12	0.14	
		# detects	31	0	100	1	11	105	5	8	0
		det.freq.	23.5%		75.8%	0.8%	8.3%	79.5%	3.8%	6.1%	
		max	4.6		29	0.15	1.3	2.1	0.46	4.8	
WNT6	132	mean	1.38		0.87	0.14	0.84	0.27	0.21	0.47	
		st. dev.	3.78		2.96	0.01	2.19	0.39	0.15	0.89	
		median	0.26		0.25	0.14	0.18	0.19	0.14	0.21	
		# detects	30	0	100	3	12	104	16	10	0
		det.freq.	22.7%		75.8%	2.3%	9.1%	78.8%	12.1%	7.6%	
		max	20		28	0.15	7.8	3.6	0.66	3	
SQW2	158	mean	0.92		0.94		0.60	0.21	0.18	0.90	0.11
		st. dev.	1.68		2.72		1.34	0.23	0.09	0.98	
		median	0.23		0.28		0.18	0.16	0.13	0.56	0.11
		# detects	43	0	113	0	18	122	13	6	1
		det.freq.	27.2%		71.5%		11.4%	77.2%	8.2%	3.8%	0.6%
		max	7		22		5.9	1.9	0.37	2.8	0.11
SQW1	158	mean	0.58		0.62		0.44	0.23	0.14	0.37	
		st. dev.	0.79		1.23		0.75	0.14	0.05	0.35	
		median	0.25		0.29		0.21	0.20	0.12	0.19	
		# detects	36	0	127	0	12	143	24	8	0
		det.freq.	22.8%		80.4%		7.6%	90.5%	15.2%	5.1%	
		max	3.6		7.7		2.8	1.4	0.34	1.1	
SQW3	132	mean	0.37	0.10	0.53		0.43	0.18	0.12	0.17	
		st. dev.	0.37	0.00	1.29		0.95	0.10	0.02	0.12	
		median	0.20	0.10	0.26		0.14	0.16	0.11	0.12	
		# detects	24	2	107	0	14	103	11	9	0
		det.freq.	18.2%	1.5%	81.1%		10.6%	78.0%	8.3%	6.8%	
		max	1.3	0.1	12	0	3.7	0.83	0.18	0.46	
SQW4	132	mean	0.56		0.79		0.20	0.31	0.23	0.83	0.24
		st. dev.	0.54		2.63		0.12	0.40	0.22	1.16	
		median	0.37		0.23		0.16	0.16	0.13	0.18	0.24
		# detects	26	0	67	0	3	15	4	8	1
		det.freq.	19.7%		50.8%		2.3%	11.4%	3.0%	6.1%	0.8%
		max	2.2	0	21	0	0.33	1.7	0.56	2.8	0.24
SQW5	132	mean	0.47	9.15	1.09		0.47	0.19	0.22	1.13	
		st. dev.	0.51	11.10	5.89		0.60	0.32	0.20	2.16	
		median	0.29	9.15	0.20		0.26	0.12	0.13	0.32	
		# detects	26	2	82	0	10	44	4	16	0
		det.freq.	19.7%	1.5%	62.1%		7.6%	33.3%	3.0%	12.1%	
		max	2.2	17	53		2.1	2.2	0.52	8.7	

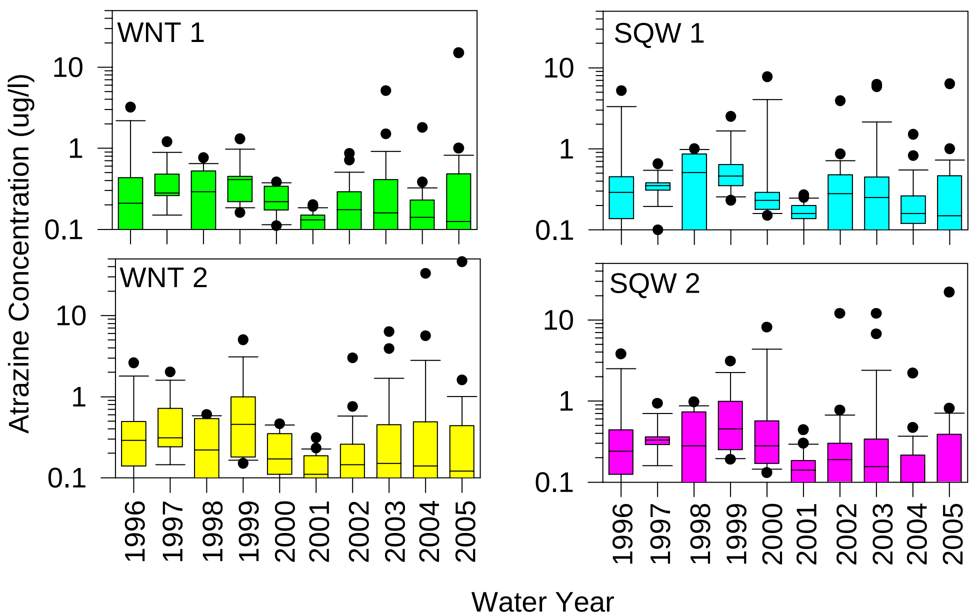


Figure 51. Box plots of atrazine concentrations by water year at upstream downstream sites in Walnut and Squaw creek watersheds.

was not detected in any water samples collected after Water Year 2001.

Many of the maximum concentrations of herbicides as well as one-time detections of other pesticides occurred on sample days corresponding closely to runoff events. On June 8, 2005, water sampling occurred immediately following approximately one inch of rain that fell in both watersheds. Field turbidity levels were greater than 1000 NTU at both WNT2 and SQW2 during sampling. Atrazine was detected at 46 and 22 ug/l at WNT2 and SQW2, respectively, representing the maximum concentration detected in either watershed during the project. Also on that day, acetochlor and metolachlor were detected at concentrations of 2.4 and 0.39 ug/l at WNT2 and 6.6 and 2.8 ug/l at SQW2. The concentrations of metolachlor were the maximum detected during the project at the watershed outlets. Maximum acetochlor concentrations at WNT2 (18 ug/l) occurred on May 13, 2004, which had the second highest atrazine concentration measured during the project (33 ug/l). In Squaw Creek (SQW2), maximum acetochlor concentration (7 ug/l) occurred on May 15, 2003, a day that also had the second highest atrazine concentration measured in the watershed (12 ug/l). Overall, locations where maximum herbicide concentrations occurred were variable across the watershed and subbasins, with highest acetochlor concentration (20 ug/l) detected at WNT6, highest alachlor concentration at SQW5 (17 ug/l), highest atrazine concentration at SQW5 (53 ug/l), highest DEA concentration at WNT6 (3.6 ug/l) and highest metolachlor concentration at SQW5 (8.7 ug/l) (Table 22). Other herbicides detected on a rare basis at main stem sites included dimethenamid (0.1 to 0.53 ug/l), propazine (0.17 to 0.52 ug/l), simazine (0.11-0.14 ug/l), fludioxonil (0.55 ug/l) and carbofuran (5.2 ug/l).

Atrazine and acetochlor concentrations by month were highest in May and June at WNT2 and SQW2 sites (Figure 56). After peaking in May and June, concentrations decreased slowly for atrazine and very rapidly for acetochlor

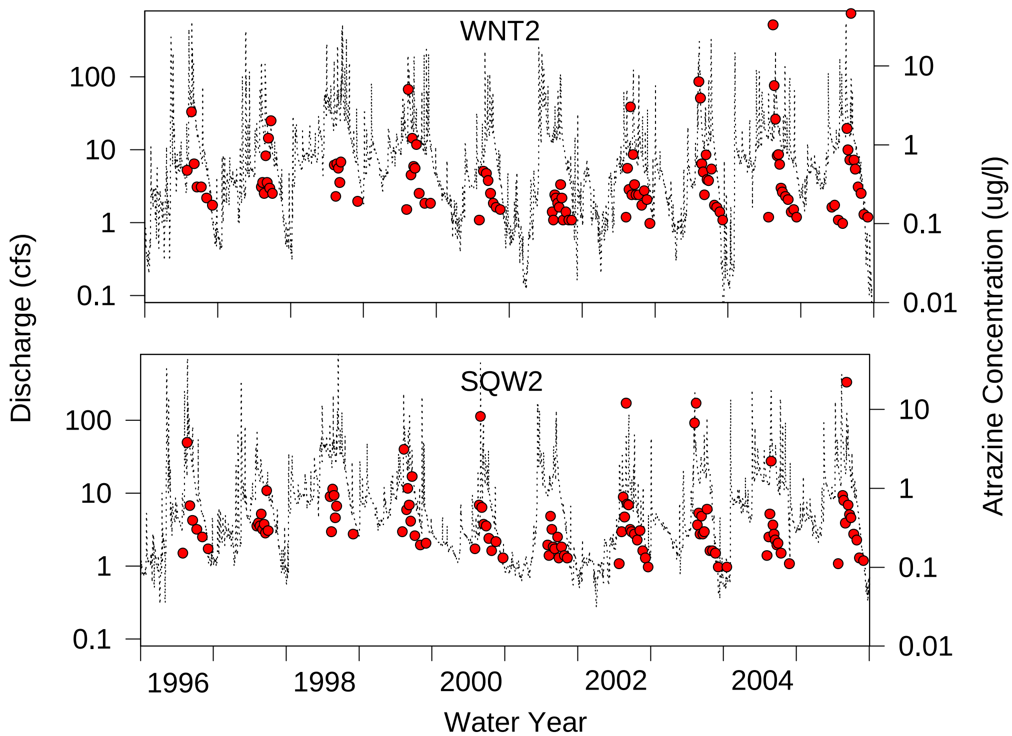


Figure 52. Relation of atrazine concentrations to discharge at WNT2 and SQW2.

through the summer and fall. The seasonal pattern of DEA was different than its parent compound atrazine (Figure 56). DEA concentrations increased from May to June and remained rather constant from June to August. Concentrations decreased in September and DEA was only sporadically detected during the October-March period. This may reflect groundwater contribution of DEA to streams and the breakdown of atrazine over time.

Loads

Annual export of atrazine from the Walnut and Squaw creek watersheds was estimated using the ESTIMATOR model and ranged from 3.7 to 27.2 kg at WNT2 and 3.0 to 26.0 kg at SQW2 (Table 23). Export was greatest in wet years 1996 and 1998 and least during dry years 2000 to 2002 at all measurement sites. Overall export was higher from Walnut Creek (WNT2) than Squaw Creek (SQW2) and lowest from upstream Walnut Creek (WNT1). However, high atrazine concentrations detected during flow events in Walnut Creek (i.e., June 8, 2005 sampling event) may have biased the ESTIMATOR model estimates to the high side. Atrazine loads often exceeded 1 kg in May and June in both watershed areas each year (Figure 57). Over a ten-year period, the months of May and June accounted for approximately 78-83% of the export load of atrazine, and the period of April through July accounted for 96 to 97% of the annual atrazine load each year (Table 23). DEA loads were substantially less than atrazine with peak loads in May and June exceeding 0.1 kg (Figure 58). Though May and June accounted for the majority of DEA loss (58-63 percent), July and August were a much greater contributors to total annual DEA loads than for atrazine, accounting for 16-27 percent of the annual export (Table 24). Loads of DEA

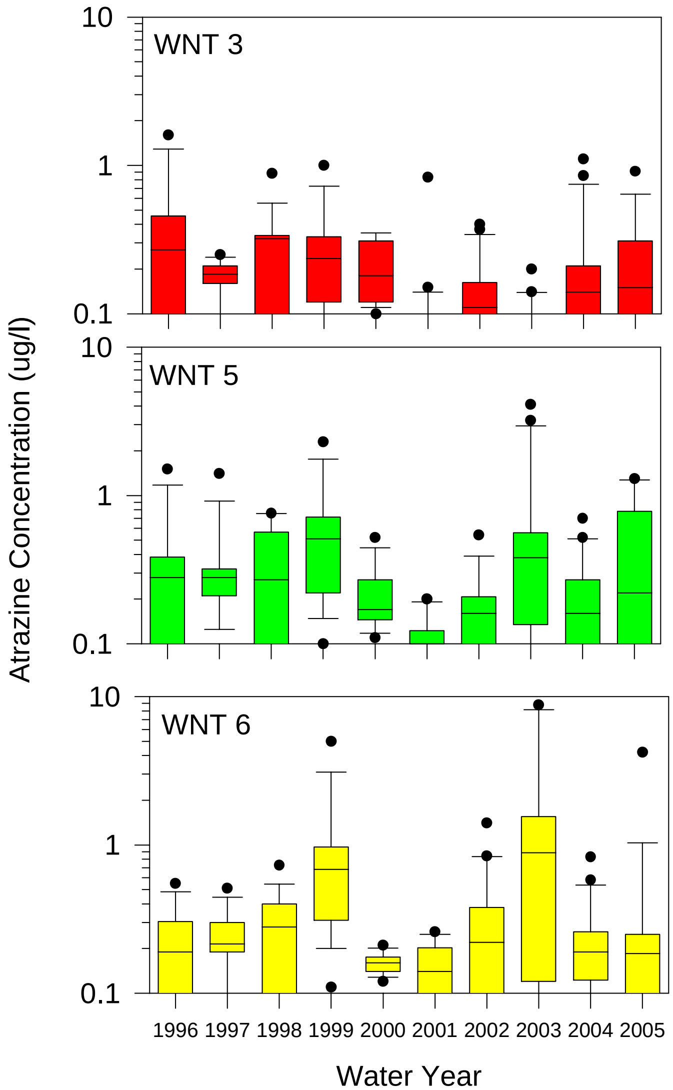


Figure 53. Box plots of atrazine concentrations by water year at Walnut Creek subbasin sites.

from months other than May and June comprised 37-42 percent of the annual loss, compared to 3-4 percent of atrazine losses.

On a unit area basis, atrazine loads ranged from 0.70 to 5.22 g/ha in Squaw Creek watershed and 0.63 to 5.53 g/ha in Walnut Creek watershed (Figure 59). Average annual atrazine loads in Walnut Creek (2.36 g/ha) were higher than Squaw Creek (1.90 g/ha). Within Walnut Creek watershed, atrazine loading rates tended to be higher in lower Walnut Creek (WNT2-1) compared to upper Walnut Creek (WNT1) (Figure 59).

Trends

Trends in herbicide concentrations over time were evaluated for atrazine and DEA since these two constituents were most frequently detected in surface water in the watersheds. However, concentrations of both analytes were periodically below laboratory detection limits (<0.1 ug/l), so consideration was given to selecting statistical tests capable of addressing arbitrary censored data. The maximum likelihood estimation (MLE) regression model for arbitrary censored data and the Seasonal Kendall tau test were chosen to evaluate trends.

The MLE test is commonly used in survival analyses in which the subject's length of time is not known precisely. While subject's length of time might be reported as >5 years, atrazine and DEA censored concentrations are reported as <0.1 ug/L. The MLE regression for arbitrary censored data allows for the censored data to be specified as between 0.0 and 0.1 while fully specifying the concentration of non-censored data. The remainder of the analysis is similar to the more common regression analyses that do not have censored data. This analysis method was recommended in a recent text by Helsel (2005).

The Seasonal Kendall tau test requires that the data be organized by season. Given the varied sampling throughout the course of the year (monthly to weekly), it was not clear whether to define the number of seasons in a year as 12 (corresponding to monthly sampling), 24 (corresponding to biweekly sampling), or 52 (corresponding to weekly sampling). Considering the number of seasons that would have missing values, the Seasonal Kendall tau test was chosen to evaluate the data with 12 and 24 seasons. The Seasonal Kendall tau test divides the year into evenly spaced time intervals and then selects the observation closest to the midpoint of the time interval to construct the time series for analysis.

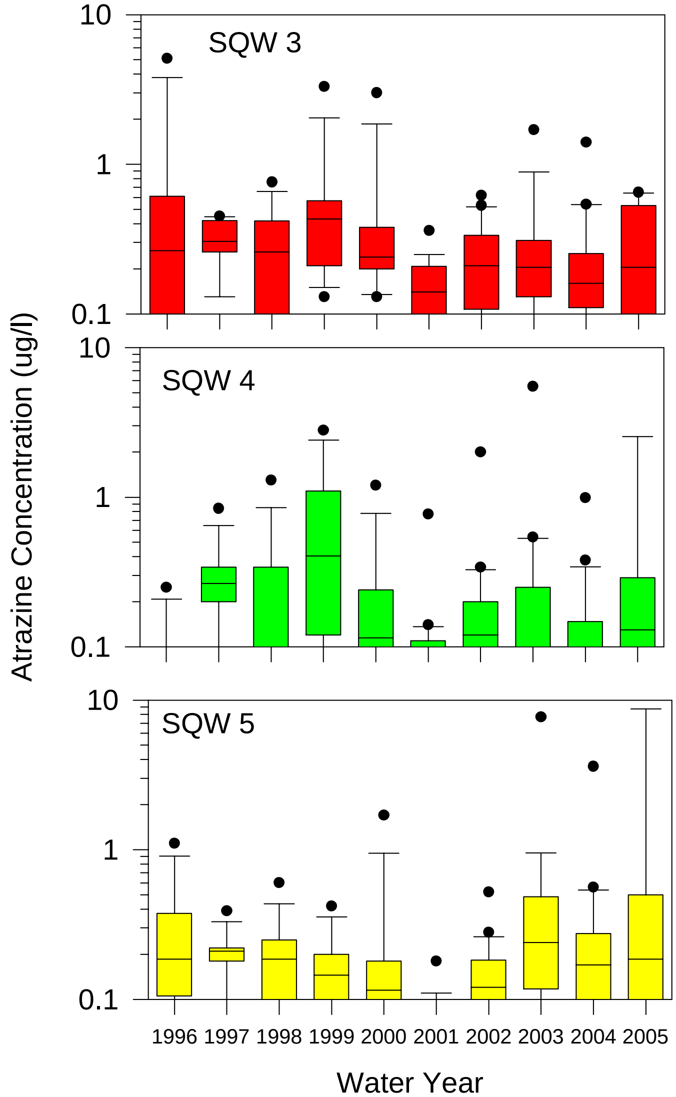


Figure 54. Box plots of atrazine concentrations by water year at Squaw Creek subbasin sites.

Because both atrazine and DEA were highly correlated with stream flow at the stream gaging sites, discharge was used as an explanatory variable in the statistical analysis. However, daily flow data were only available at WNT1, WNT2, and SQW2. Since daily flow data were significantly cross-correlated to one another, it was presumed that daily flows at the non-gaged locations were strongly correlated to the gaged stations. Therefore the downstream flows (i.e., WNT2 and SQW2) were used as explanatory variables for ungaged sites in their respective watershed.

For the MLE regression, log transformation of concentration and flow was applied. Based on examining probability plots of residuals, log transformation of concentration and flow was judged to be superior to other alternatives. Herbicide concentrations were significantly correlated with flow in every instance (Table 25). Both decreasing and increasing trends over time were observed. Sites WNT3 and SQW2 had decreasing trends in atrazine concentration with respect to time whereas sites WNT5, WNT6, and SQW5 had increasing trends in DEA concentration with respect to time.

The Seasonal Kendall tau test identified decreasing trends in atrazine concentration with respect to time at sites WNT3 (12 seasons), SQW2 (12 seasons), and SQW3 (12 and 24 seasons) but increasing trends in DEA concentration over time at sites WNT5 (24 seasons), WNT6 (24 seasons), and SQW5 (24 seasons) (Table 25).

MLE regression was also conducted with additional explanatory variables to further test for trends in the concentration of atrazine and DEA at WNT2. Concentrations at the upstream site, WNT1, and the downstream site in Squaw Creek, SQW2, were used. For atrazine, the model excluding the upstream Walnut Creek concentration data was more appropriate whereas for DEA, the model including the upstream Walnut Creek concentration data was better (Table 26). However, in both cases, no significant trend with time was found.

Discussion

Results from surface water monitoring at Walnut and Squaw creek watersheds indicated that atrazine and DEA were the most commonly detected herbicides in both watersheds with detection frequencies greater than 70 percent. Acetochlor was occasionally detected (up to 27 percent) whereas alachlor and metolachlor were rarely detectable (less than 5%). Cyanazine detections were also rare

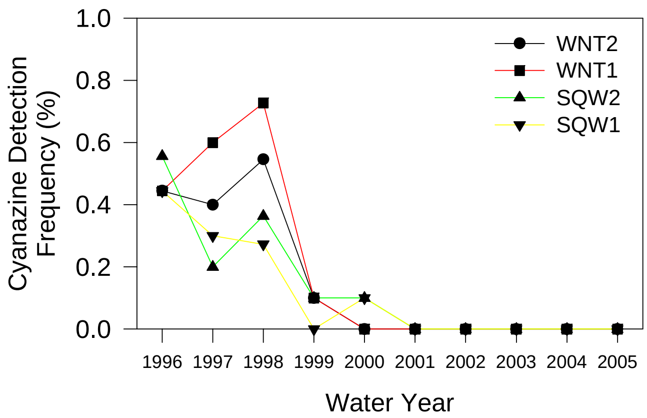


Figure 55. Detection frequency of cyanazine by water year at upstream and downstream sites in Walnut and Squaw creek watersheds.

during the last five years of the project. Concentrations of atrazine often exceeded 1 ug/L during high streamflows in late spring/early summer; however, overall median concentrations of atrazine and DEA were less than 0.3 ug/l. May and June accounted for approximately 80 percent of the export load of atrazine, and the period of April through July accounted for 96 percent of the annual atrazine load.

Statistical trends in atrazine and DEA concentrations over time indicated that when trends were significant, atrazine concentrations decreased but DEA concentration increased. In Walnut Creek watershed, a decreasing trend in atrazine concentration was measured in subbasin WNT3, but increasing trends in DEA concentrations were evident at WNT5 and WNT6. Similar decreasing and increasing atrazine and DEA trends were also observed in the control watershed Squaw Creek, with decreasing atrazine concentrations at SQW2 and SQW3 and increasing DEA concentration at SQW5.

The concentrations and stream loading patterns of herbicides and the relation of trends in atrazine to trends in DEA may reflect the manner in which herbicides are delivered to streams, their chemical breakdown, and overall sampling patterns. Herbicides are primarily delivered to streams via runoff, and because of this, concentrations show a high degree of variability. Depending on the timing of sample collection in relation to storm events and application periods, herbicide concentrations can vary orders of magnitude. This was particularly illustrated during the last year of monitoring when a sampling event in June 2005 produced the highest atrazine concentrations measured in both watersheds during the entire 10-year project. Thus, the timing of sampling relative to runoff events and application periods becomes the driving factor in determining herbicide concentrations in surface water. With a set weekly to monthly sampling schedule established for the Walnut Creek project, monitoring surface water for runoff-driven herbicides was a rather hit or miss proposition. Periodic capture of elevated concentrations occurred randomly depending on whether sampling coincided with a runoff event.

The decreasing and increasing trends in atrazine and DEA concentrations over time may be influenced by the relation of parent to daughter product during the year. Peak atrazine concentrations clearly occurred in May and June whereas peak DEA concentrations were often found in June and July. Concentrations of DEA often occurred in streams throughout the fall whereas atrazine concentrations often dropped below detection limits during this time. DEA detections in late summer and fall may be related to the slow breakdown of atrazine during the course of the year and possibly greater contribution of baseflow discharge to streams. In Water Year 2001, water sampling for herbicides was expanded from an April to July period to include late summer, fall and winter samples (same schedule as anions). Thus the second half of the project contained a greater proportion of samples collected during the late summer and fall when atrazine concentrations decreased to nondetected levels and DEA concentrations remained detectable. Hence it is possible that some of the apparent changes in herbicide concentrations over time

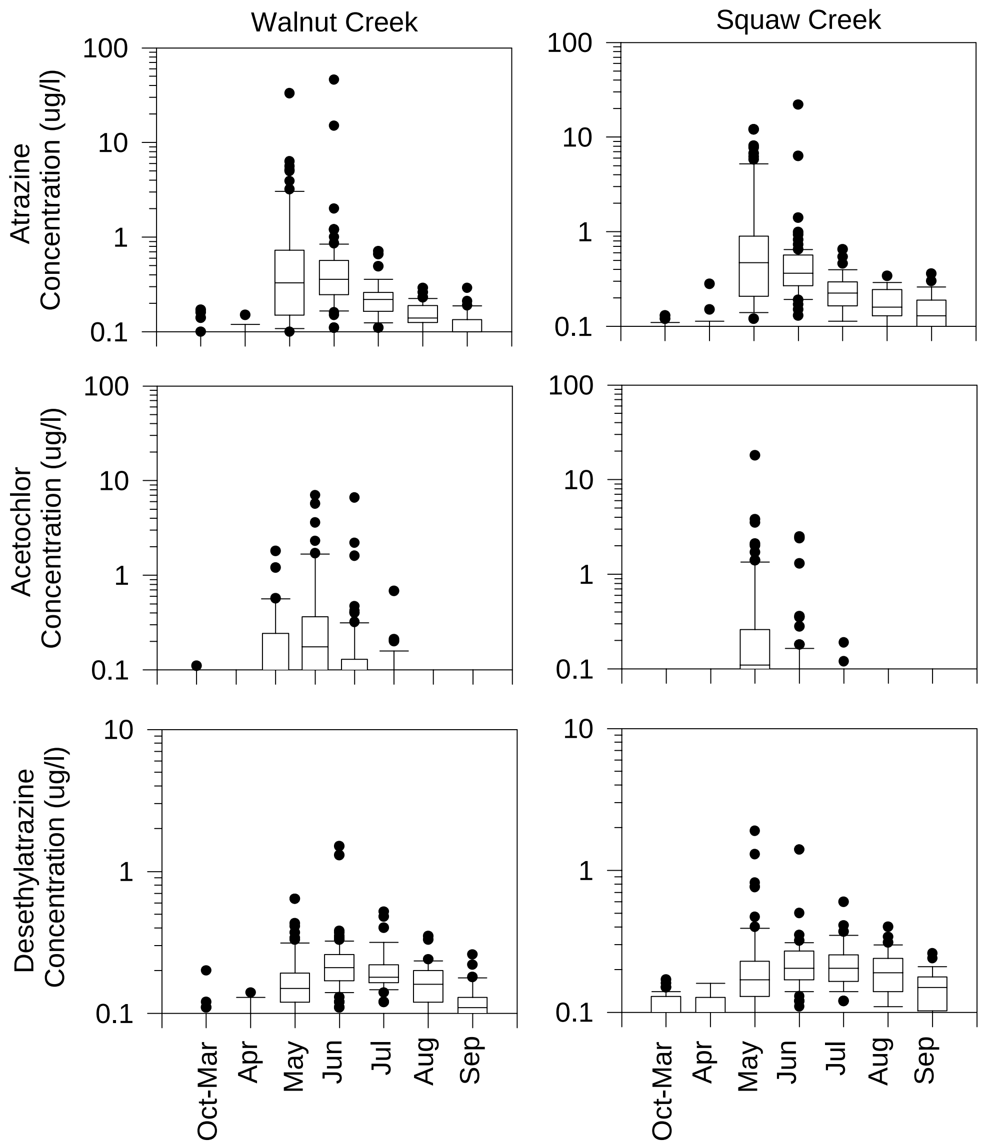


Figure 56. Box plots of monthly concentrations of atrazine, acetochlor and desethylatrazine at downstream monitoring sites (WNT2 and SQW2) in Walnut and Squaw creek watersheds.

may be related to the sampling change relative to the breakdown and persistence of atrazine and DEA.

Despite the effects of sampling on measuring herbicide concentrations, it was evident that some basins showed statistically significant changes whereas others did not. Part of the detection of water quality change may be related to land use patterns in Walnut and Squaw creeks. As reported in Schilling et al. (2002), atrazine losses were measured in 45 subwatersheds during a two-day period in May 1999 and major differences were evident within the Walnut Creek watershed. Highest atrazine loads per ha were located in many headwater areas. Of the 45 subwatersheds sampled in this study, nine subwatersheds (20 percent) showed annualized atrazine loads greater than 0.72 g/ha, six of which were located in the watershed area above the WNT1 gage. In contrast, the

Table 23. Summary of total annual atrazine export from various watershed areas.

	Atrazine Export Load (kg)				DEA Export Load (kg)			
Water Year	WNT2	SQW2	WNT1	WNT2-1	WNT2	SQW2	WNT1	WNT2-1
1996	22.5	13.7	4.5	17.9	1.9	1.0	0.8	1.1
1997	6.1	2.6	1.1	5.0	1.0	0.4	0.4	0.6
1998	27.2	26.0	5.9	21.3	5.3	2.4	1.9	3.4
1999	7.7	6.6	2.1	5.6	2.6	1.1	1.1	1.4
2000	4.7	10.1	1.5	3.2	1.5	0.9	0.7	0.8
2001	4.2	4.7	0.9	3.3	1.5	0.9	0.7	0.7
2002	3.7	3.0	1.2	2.5	1.2	0.4	0.6	0.6
2003	15.6	6.0	1.9	13.7	3.4	1.0	0.9	2.5
2004	11.8	8.3	1.9	9.9	2.2	1.5	0.9	1.3
2005	19.9	8.3	2.3	17.5	2.0	1.1	0.8	1.3
Avg.	12.3	8.9	2.3	10.0	2.3	1.1	0.9	1.4

core of the watershed occupied by the Neal Smith NWR showed annualized atrazine loads less than 0.007 g/ha. Results of this one-time sampling event suggest that differences in atrazine loading rates within Walnut Creek watershed are more pronounced than implied by the concentration and loading patterns measured at watershed outlets. Atrazine loads varied by more than two orders of magnitude in the Walnut Creek watershed. Thus, herbicide losses are not equal across watershed areas and concentrations and trends measured at watershed outlets may primarily reflect herbicide losses from susceptible areas where atrazine is applied. Greater atrazine losses were measured in smaller catchments within the WNT5 and WNT6 subbasins. In all likelihood, herbicide contributions from these areas dominated detections of atrazine and DEA at their subbasin outlets to Walnut Creek and may be responsible for the increasing DEA trends. Decreasing trends in atrazine at WNT3 would be consistent with reduced row crop occurring in the subbasin and reduced application of atrazine. In Squaw Creek subbasin SQW5, increasing row crop production in this watershed from 1990 to 2005 would be consistent with increasing DEA concentrations in the stream draining this area. Interestingly, atrazine concentrations at SQW5 did not show a significant increase during the project.

FECAL COLIFORM

The sanitary quality of water is often assessed using bacteriological methods that detect the presence of certain bacteria that indicate the presence of fecal material from warm-blooded animals (USEPA, 1986). Concentrations of the fecal coliform bacteria, such as *Escherichia coli* and *Aerobacter aerogenes*, do not necessarily present risks for waterborne disease, such as gastroenteritis, bacillary dysentery, or others, but are found in association with pathogenic microorganisms (*Salmonella*, *Shigella*, etc.) that do present a risk of infection. The origin of fecal coliform contamination can be from point sources, such as outfall from sewage treatment plants, or nonpoint sources. Nonpoint sources include a variety of diffuse sources, including agricultural animal waste storage and manure application, agricultural runoff from pastures or manure-applied fields, failed septic systems, urban and construction runoff, landfill leakage and wildlife waste. The EPA primary contact water quality standard for fecal coliform bacteria is 200 colony-forming units per 100 ml of water (200 CFUs or counts/100 ml).

Counts

Fecal coliform counts varied widely among sampling sites and water years, ranging from less than 10 counts/100 ml to 13 million counts/ 100 ml at SQW2 (Table 27). Maximum values also exceeded one million counts at WNT1 and SQW5 during the project and annual high concentrations often one to ten thousand counts each year at all sites. Figures 60-62 show box plots for fecal coliform counts detected in water samples collected from various sample sites. Highest median values occurred at WNT1 where the median fecal coliform count was 905 counts/100 ml (Table 27). Median values at downstream watershed outlets WNT2 and SQW2 were slightly higher at WNT2 (720 and 650 counts/100 ml) whereas all other subbasin sites were less than 440 counts/100 ml. All median fecal coliform values exceeded the water quality criterion of 200 counts/100 ml. At the main stem sampling sites, measured fecal coliform values exceeded 200 counts/100 ml about 25 percent of the time, ranging from 22.3 percent at SQW2 to 29.2 percent at SQW1. In the subbasins, lowest overall median values were measured in WNT3 and WNT6 (265 to 285 counts/100 ml) and highest median values were measured in SQW3 and SQW5 (410 to 440 counts/100 ml) (Table 27).

Annually, median fecal coliform counts tended to be higher at WNT2 than SQW2, with median annual values at WNT2 higher in seven out of ten years of monitoring (Table 28). Similarly at upstream sites, WNT1 median values were routinely higher than SQW1. Median annual fecal coliform values at WNT1 exceeded 1000 counts/100 ml in the first four years of monitoring but were less than 1000 in water years 2000 to 2005. At all sites, median annual fecal coliform concentrations were lowest in Water Year 2002, ranging between 165 to 440 counts/100 ml at all sites.

Seasonally, monthly median values were typically highest in summer through early fall (June to October) (Figure 63). Monthly median values exceeded 1000 counts/100 ml at the four

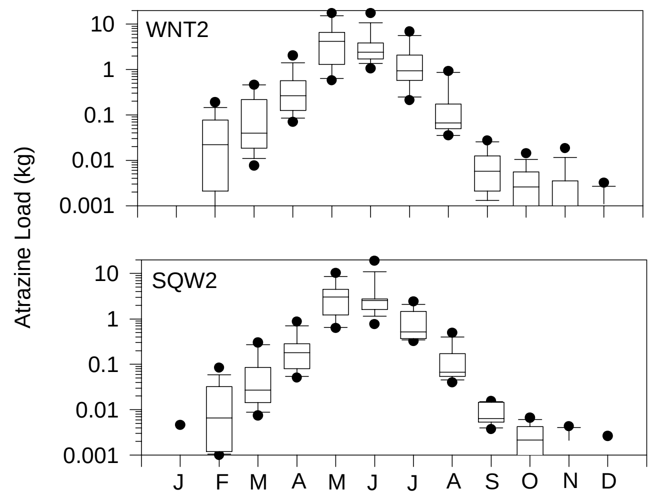


Figure 57. Box plots of atrazine loads by month at WNT2 and SQW2 monitoring sites.

main stem sites during the months of July through October (with an exception of a median July value for SQW1 at 945 counts/100 ml). In contrast, with one exception, median fecal coliform counts at the four main stem sites in May and June were less than 720 counts/ 100 ml (the exception was a median value of 1500 counts/100 ml at WNT2 in June). However, maximum annual values (as detected from routine sampling) occurred during any month of the year. An assessment of the upper 25 percent of fecal coliform values (50 highest detections) measured during sampling at the four main stem sites revealed that peak concentrations typically occurred during the months of May through August. Of the top 50 measured values, nine to 17 of the top 50 occurred in June, seven to nine occurred in July and eight to 10 occurred in August. Elevated fecal coliform was also measured occasionally in the months of May, September and October. Thus, while median monthly values tended to be highest in late summer and fall, peak concentrations typically occurred at any time between May and October each year (Figure 64).

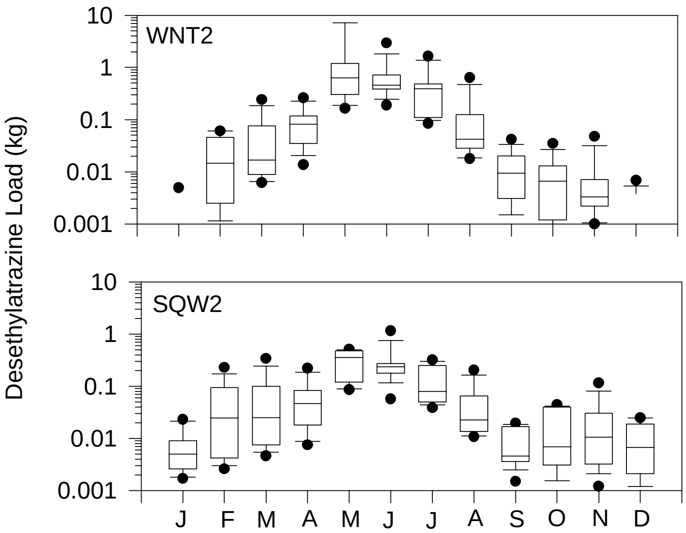


Figure 58. Box plots of desethylatrazine loads by month at WNT2 and SQW2 monitoring sites.

Trends

Statistical analyses of trends in fecal coliform counts over time were conducted in the same manner as the anion trends. Data were evaluated for distribution and autocorrelation, and exploratory analysis was conducted to assess covariates of season, downstream vs. upstream, treatment vs. control, and concentrations vs. discharge. Fecal coliform data distribution was approximately log transform, so the log10(x) transformation was utilized in the analysis. A strong to moderate bimodal seasonal pattern was evident in the fecal coliform data but autocorrelation was slight for log(FC). The autocorrelation coefficients for lag 1 time series ranged from 0.20 to 0.38, and averaged 0.31 at all 10 monitoring sites. Downstream fecal coliform counts were significantly related to upstream values at both Walnut and Squaw creeks with r^2 values of 0.65 and 0.50, respectively. Similarly, fecal coliform concentrations at WNT2 (treatment) were significantly related to those measured at SQW2 (control) with an r^2 of 0.50. Finally, fecal coliform was marginally correlated with discharge at all sites except WNT5, WNT6, SQW3 and SQW4. When correlations were statistically significant, the coefficients ranged from 0.14 to 0.28. The low correlation of fecal coliform to discharge may be related to periodic occurrence of high values under low discharge levels.

The best set of explanatory variables for WNT2, downstream Walnut Creek Watershed were:

- Season
- Log(WNT2Q), Mean daily discharge, downstream Walnut Creek
- Log(WNT1FC), Log(FC) upstream station, Walnut Creek
- Log (SQW2FC), Log(FC) downstream control watershed, Squaw Creek

For the subbasins, covariates of season and log discharge were used.

For fecal coliform, there was strong seasonal signal that was adjusted for in the multivariate models. However, there was no evidence that the change over time was statistically different for any month so the parallel slope model was run for the upstream and downstream station. There was a statistically significant decrease in fecal coliform at upstream WNT1 (r^2=0.49, slope -9.909e-10

Table 24. Percentage by month of average annual loss of atrazine and desethylatrazine (DEA) from downstream WNT2 and SQW2 sites.

	Atrazine Load		DEA Load	
Month	WNT2	SQW2	WNT2	SQW2
Jan	0.01%	0.01%	0.08%	0.74%
Feb	0.38%	0.20%	1.03%	5.18%
Mar	1.11%	0.86%	2.37%	6.49%
Apr	3.81%	2.93%	4.26%	6.39%
May	46.12%	41.60%	30.71%	30.08%
Jun	31.88%	42.78%	32.38%	28.98%
Jul	14.67%	9.91%	21.78%	12.07%
Aug	1.88%	1.56%	5.94%	4.74%
Sep	0.08%	0.09%	0.59%	0.77%
Oct	0.03%	0.03%	0.41%	1.49%
Nov	0.02%	0.01%	0.40%	2.29%
Dec	0.01%	0.00%	0.09%	0.94%

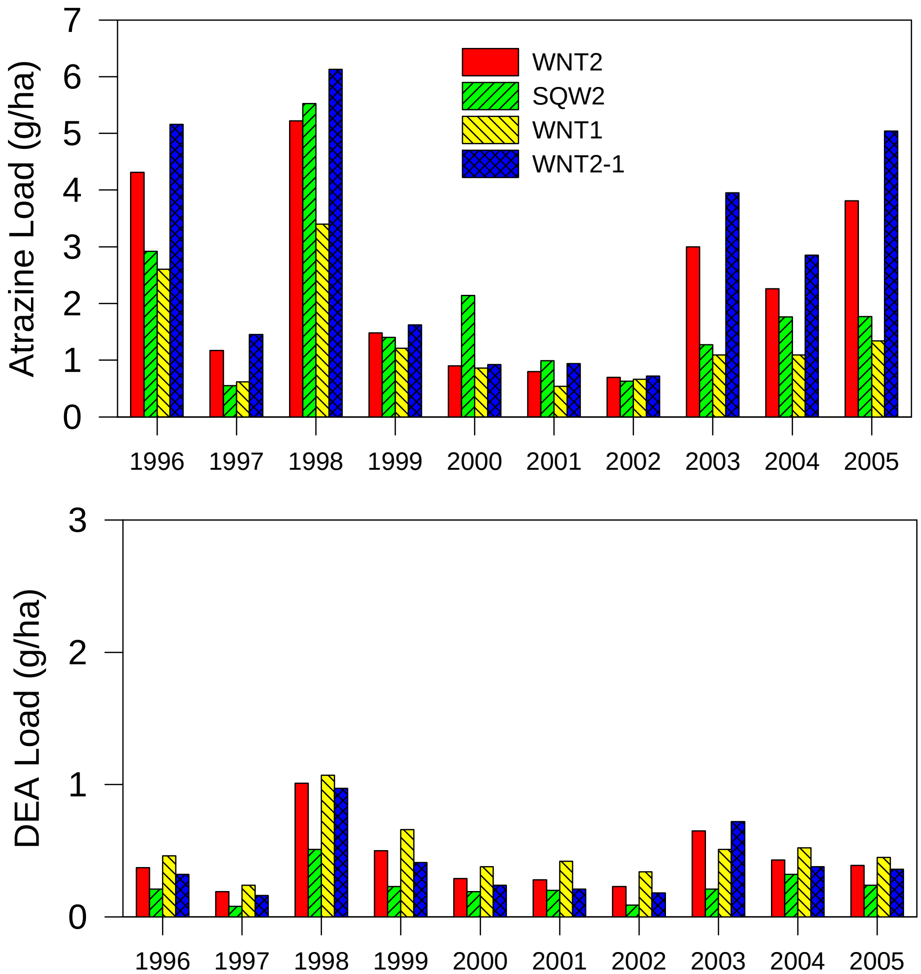


Figure 59. Annual atrazine and desethylatrazine loads estimated for various watershed areas in Walnut and Squaw creek watersheds.

or 48 percent change), but there was no statistical evidence of a change in fecal coliform for the downstream Walnut Creek station (Table 29). There was a 300% increase in fecal coliform in subbasins WNT5 and WNT6, but no significant change in subbasin WNT3.

For the Squaw Creek watershed, no significant change was detected in either the upstream or downstream main stem stations. However, increases were observed in SQW3 and SQW5, but no change in SQW4 (Table 29).

Discussion

Results from surface water monitoring at Walnut and Squaw Creek watersheds indicate that fecal coliform bacteria were detected frequently above the EPA water quality standard of 200 count/100 ml in both watersheds. Elevated detections were occasionally observed at all monitored watersheds with highest fecal coliform counts occurring at any time between May and October during high stream flow periods associated with rainfall runoff. No changes in fecal coliform concentrations were observed during the 10-year monitoring project at downstream Walnut Creek (WNT2). Increases in fecal coliform concentrations were noted in two Walnut subbasins but this did not appear to affect downstream patterns. Similarly, subbasin changes in Squaw Creek watershed did not result in changes in downstream Squaw Creek levels measured at SQW2.

Table 25. Atrazine and desethylatrazine trends analysis results using MLE regression for arbitrary censored data and seasonal Kendall tau.

	Atrazine					Desethylatrazine			
	MLE Regression		Seasonal Kendall tau p-value for Time			MLE Regression		Seasonal Kendall tau p-value for Time	
Site	p-value Time	p-value Flow	12 season	24 season	Site	p-value Time	p-value Flow	12 season	24 season
WNT1	0.491	<0.001	0.4318	0.4746	WNT1	0.308	<0.001	1.0000	0.3347
WNT2	0.668	<0.001	0.4262	0.6988	WNT2	0.461	<0.001	0.9631	0.3161
WNT3	▼0.052	<0.001	▼0.0343	0.1092	WNT3	0.875	0.004	0.8402	0.6121
WNT5	0.414	<0.001	0.4756	0.3193	WNT5	▲<0.001	<0.001	0.0755	▲0.0047
WNT6	0.341	<0.001	0.6648	0.5688	WNT6	▲0.001	<0.001	0.3398	▲0.0472
SQW1	0.148	<0.001	0.2261	0.1853	SQW1	0.139	<0.001	0.6079	0.3086
SQW2	▼0.053	<0.001	▼0.0375	0.1550	SQW2	0.494	<0.001	0.7952	0.2684
SQW3	0.191	<0.001	▼0.0281	▼0.0603	SQW3	0.846	<0.001	0.2602	0.3876
SQW4	0.693	<0.001	0.4995	0.1646	SQW4	0.142	0.001	0.2010	0.0777
SQW5	0.067	<0.001	0.4348	0.5186	SQW5	▲0.007	<0.001	0.1703	▲0.0117

▼ and ▲ indicate significant downward and upward trends in concentration with time, respectively.

In the Walnut and Squaw Creek watersheds, primary sources of fecal coliform include pastures and manure application to cropped fields. Outfall from the sewage treatment plants of Prairie City and Colfax does not enter into the Walnut or Squaw Creek watersheds. Outfall from the Prairie Learning Center at the Neal Smith National Wildlife Refuge does enter the Walnut Creek watershed, but the wastewater is directed through constructed wetlands where bacteria and nutrients are removed prior to discharge.

Most permanent pastures in the watersheds have access to a waterway for direct contact of livestock with surface water supplies. Hence, pasture sites located near several monitoring sites likely contributed to elevated fecal coliform counts. Grazing impacts and manure from pastures probably affected phosphorus concentrations and loads as well. Direct deposition of waste into streams, destruction of riparian vegetation, and trampling of streambanks and streambeds are all problems associated with unrestricted livestock grazing. Pasture sites are located upstream of WNT1 and WNT2 and the WNT5 subbasin includes much of the bison enclosure for the refuge. However, one pasture located immediately upstream of the WNT1 gage was discontinued at some point early in the project (pre-2000), which may explain the decrease in fecal coliform at the WNT1 site. Although median and maximum fecal coliform levels were not particularly elevated in the WNT5 subbasin, fecal coliform levels increased in the WNT5 subbasin over the 10-year project. The low stocking density of the bison in the prairie area may have reduced fecal loading in the prairie pasture compared to typical pastures in the area, but still increased bacterial counts above early project levels. In Squaw Creek watershed, a decreasing trend in fecal coliform in SQW5 subbasin may be related to the decrease in grassland in the subbasin. Reduced grassland in the subbasin may signal less land available for grazing. The reasons for the significant increases in fecal coliform at WNT6 and SQW3 are unknown.

PHOSPHORUS

Phosphorus (P) monitoring began in Water Year 2001 and thus five years of monitoring data are available for analysis. P concentrations

Table 26. Atrazine and desethylatrazine trends analysis results at WNT2 using MLE regression for arbitrary censored data and covariates of date, log discharge, SQW2 concentrations and with or without upstream WNT1 concentration as a covariate.

	*Atrazine excluding WNT1 (p values)	Atrazine including WNT1 (p values)	Desethylatrazine excluding WNT1 (p values)	*Desethylatrazine including WNT1 (p values)
Date	0.829	0.639	0.842	0.465
Log_Q (flow)	<0.001	<0.001	<0.001	0.001
SQW2 Concentration	<0.001	0.003	<0.001	<0.001
WNT1 Concentration	Not included	0.247	Not included	<0.001

* Best model

were highly skewed (skewness ranging between 1.8 and 6.1 with and averaging 5.8), so concentration patterns were typified by many low concentration values punctuated by occasional elevated values. Concentrations ranged from less than the detection limit to a maximum of 4.2 mg/l measured at SQW2 (Table 30). It should be noted that laboratory detection limits for P changed during the project. Before August 22, 2001, the detection limit was reported to be <0.1 mg/l. On August 22 and September 5, 2001, several water samples were reported with laboratory detection limits less than 0.02 mg/l. After February 5, 2002, the detection limit was reported to be <0.05 mg/l. Thus, characterization of P concentrations less than 0.1 mg/l may not be entirely valid for the five year P monitoring program.

Figures 65 and 66 show box plots for phosphorus concentrations detected in water samples collected from various sample sites. Highest overall median values occurred at WNT2 where the median P concentration was 0.17 mg/l (Table 30). However, median values were generally similar at all sites ranging between 0.08 to 0.17 mg/l for the monitoring period. Maximum P concentrations generally followed discharge patterns and exceeded 1-2 mg/l with occasional peak values exceeding 3-4 mg/l (Figure 67).

Annually, median P concentrations were also fairly consistent, ranging between 0.14 to 0.2 mg/l at SQW2 and 0.17 to 0.2 mg/l at WNT2 for water years 2001 to 2005 (Table 31). The range in annual median P concentrations varied between 0.06 to 0.2 mg/l at all sites. Lowest annual median values typically occurred in SQW4 and WNT3 subbasins. Maximum annual concentrations in these two subbasins were also lower than maximum values observed in other watersheds, ranging from 0.15 to 1.2 mg/l in SQW4 and 0.21 to 0.68 in WNT3. At all sites, maximum P concentrations were lowest in Water Year 2004, ranging between 0.15 to 0.73 mg/l. A distinct decrease in P concentration was evident in the SQW5 subbasin (Figure 66). Annual median values decreased from 0.2 mg/l in Water Year 2001 to 0.09 mg/l in Water Year 2005.

Seasonally, monthly median P concentrations were variable (Figure 68). Limited sample numbers from October through February limit assessment of this four month period. During the remainder of the year (March to September), monthly trends are difficult to distinguish. At WNT2, P concentrations increased from April to July and then decreased in the late summer and fall. At SQW2, a large range of values occurred in May and P concentrations appeared to increase from April to October.

Table 27. Summary of fecal coliform concentrations at project monitoring sites for water years 1996 to 2005. All concentrations in counts/100 ml.

Statistic	WNT1	WNT2	WNT3	WNT5	WNT6	SQW1	SQW2	SQW3	SQW4	SQW5
n	210	210	144	145	142	203	210	144	143	143
Mean	52813	8291	1158	2865	2170	6402	69721	2912	1922	58374
Stand. Dev.	531634	38486	3581	12873	12947	30597	897171	12735	7103	443773
Median	905	720	285	360	265	500	650	410	340	440
Maximum	7600000	450000	30000	120000	150000	250000	13000000	140000	49000	4100000
Minimum	<10	<10	<10	<10	<10	<10	10	<10	<10	<10

Interestingly, suspended sediment concentrations did not explain much of the variance in P concentrations in either Walnut or Squaw creek watersheds. Using daily suspended sediment concentrations at WNT2 and SQW2 to predict P had an r^2 of 0.22 and 0.15, respectively, although both relations were statistically significant ($p < 0.01$). Thus, while P concentrations are significantly related to suspended sediment, other factors appear to be involved.

Phosphorus export from Walnut and Squaw creek watersheds was similar, ranging from 0.98 at SQW2 to 1.05 kg/ha at WNT2 (Table 18). P losses from the WNT1 watershed area were lower than the overall Walnut Creek average (0.66 kg/ha) whereas losses from downstream Walnut Creek were higher (1.24 kg/ha). Overall, P losses from watershed areas were similar and were all within a relatively narrow range of values (0.32 kg/ha to 1.72 kg/ha) (Table 18). Flow-weighted P concentrations averaged approximately 0.3 mg/l at WNT2 and SQW2 (Table 19), similar to mean values measured during the five-year monitoring program.

Trends

Statistical analyses of trends in phosphorus concentrations over time were conducted in the same manner as the anion and fecal coliform trends. Data were evaluated for distribution and autocorrelation, and exploratory analysis was conducted to assess covariates of season, downstream vs. upstream, treatment vs. control, and concentrations vs. discharge.

For phosphorus, changes in detection limits occurred during the project that must be accounted for in the statistical analysis. The minimum recorded value was 0.01 mg/l on August 8, 2001 at WNT3, but after April 10, 2002, the minimum recorded value was 0.03. In addition, there were some detection limits marked as <0.05 mg/l after April 1, 2001 and on August 8, 2001 and September 5, 2001, there were several limits identified as <0.02. Prior to September 5, 2001, the detection limits were variable. There were numerous <0.1 values before this time. Overall the values below the laboratory detection limit were less than 20% of all measurements. In an attempt to compensate for the change of detection limits during the project, the values represented by <0.1 or 0.01, or <0.05 or <0.02 were replaced with 0.025 mg/l through the period of sampling.

Phosphorus concentrations were skewed (toward lower values) and the Shapira-Wilk W statistic indicated a non-normal distribution (P<W .0000). The W statistic ranges between 0 and 1; low values leading to a rejection of the hypothesis of normality. The fit better approximated a lognormal distribution. The subsequent analysis for phosphorus was performed on the Log(10) transformation of the concentration values. Log(P) did not show significant autocorrelation indicating that P was ‘flashy’ and not as related to previous observations. Seasonal effects were relatively weak for phosphorus compared to other

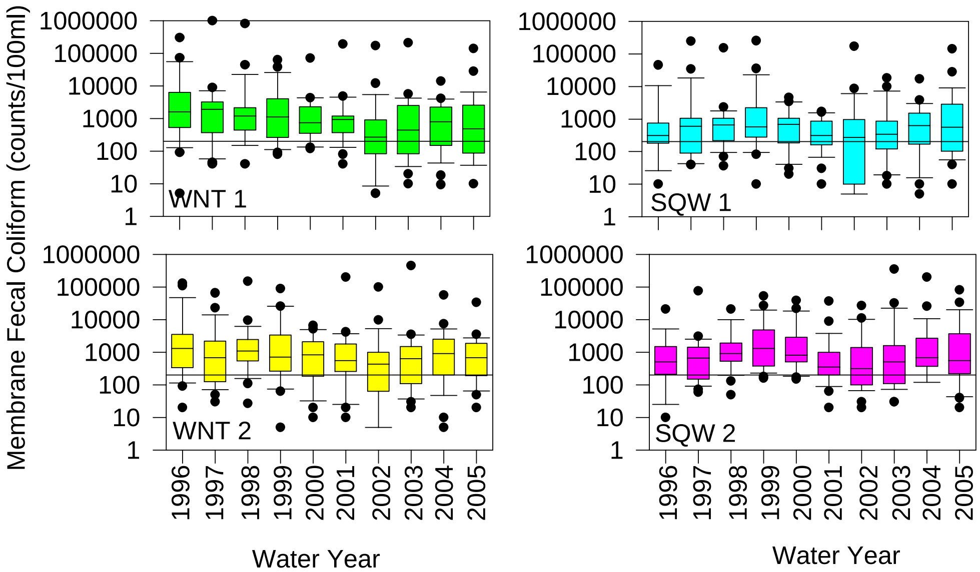


Figure 60. Box plots of fecal coliform concentrations by water year at upstream and downstream sites in Walnut and Squaw creek watersheds. A reference line for 200 counts/100 ml is indicated.

variables. In fact, log(P) at WNT1, WNT2, WNT3, WNT5, SQW2, SQW3 and SQW4 did not have statistically different seasonal signals after adjusting for any possible linear trend. Downstream phosphorus concentrations were significantly related to upstream values at both Walnut and Squaw creeks with r^2 values of 0.48 and 0.36, respectively. Phosphorus concentrations at WNT2 (treatment) were significantly related to those measured at SQW2 (control) with an r^2 of 0.24. Phosphorus concentrations were marginally related to discharge at all stations except WNT5, WNT6, SQW3 and SQW4. When correlations were statistically significant, the coefficients ranged from 0.19 to 0.35. No relation with baseflow was apparent at any sites, reinforcing the concept that phosphorus delivery is primarily related to sediment movement and storm events.

There was no statistical evidence that TP changed over time for either the upstream or downstream Walnut Creek watershed, however, there was a slight decreasing trend for WNT2 and increasing trend for the upstream WNT1 station. The best set of explanatory variables for WNT2, downstream Walnut Creek Watershed were:

- Season
- Log(WNT2Q), Mean daily discharge, downstream Walnut Creek
- Log(WNT1TP), upstream station, Walnut Creek
- Log (SQW2TP), downstream control watershed, Squaw Creek

The multiple linear regression model for phosphorus had an r^2 of 0.60 at WNT2 and 0.46 at SQW2 (Table 32). Phosphorus did not change in any of the main stem streams in either Walnut Creek or Squaw Creek. The only statistically significant trend in phosphorus was an increase in the SQW3 subbasin and a decreasing trend in SQW5.

Discussion

Lack of phosphorus concentration trends in five years of monitoring in the watersheds was not unexpected given the episodic transport and variability in P concentrations detected in water. Like herbicides, phosphorus is primarily delivered to streams via runoff, and because of this, concentrations show a high degree of variability. An event-based sampling protocol rather than a set sampling schedule would have

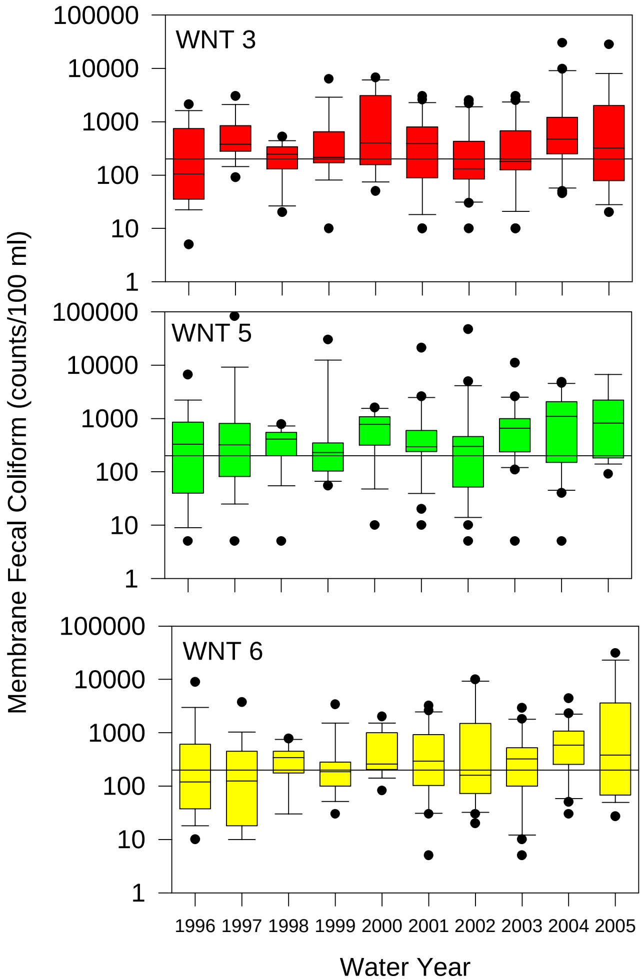


Figure 61. Box plots of fecal coliform concentrations by water year at Walnut Creek subbasin sites. A reference line for 200 counts/100 ml is indicated.

been more appropriate to detect subtle changes in concentration over time.

However, phosphorus concentrations in surface water are not easy to characterize. No differentiation was made in this study between dissolved P (orthophosphate) and total P that would include both dissolved and particulate P. Dissolved P would be primarily delivered to streams with baseflow and tile drainage, whereas particulate P would be primarily associated with runoff and sediment erosion. In a study of P concentrations in Walnut Creek alluvial groundwater, average dissolved P concentrations ranged considerably (Schilling and Jacobson, 2006). Dissolved P ranged from less than 0.1 mg/l to 1.42 mg/l in alluvial groundwater, with average P concentrations typically ranging between 0.16 to 0.26 mg/l. These peak and average concentration levels were within the range of values measured in Walnut Creek and represent only the dissolved phase of P. Given that P concentrations were only weakly related to suspended sediment, the dissolved component of P may be important during non-runoff periods.

The P data are nonetheless illustrative of typical concentrations detected in Iowa's third and fourth-order waterways in the Southern Iowa Drift Plain landscape region. Median annual P concentrations between 0.08 to 0.15 mg/l can be expected for other rural agricultural regions where uplands are mantled by loess and till and the valleys consist of loess and till-derived alluvium. Average annual P concentrations would be higher than median values, with Walnut and Squaw creek monitoring data indicating that average annual P concentrations are approximately 0.3 mg/l. Maximum P concentrations exceeded 4 mg/l in Walnut and Squaw creeks, but because sampling was not event based should probably be considered a lower bound estimate of maximum values. Average annual export of phosphorus of approximately 1 kg/ha may be an appropriate estimate for similar watersheds in the region.

BIOMONITORING

The biological (benthic macroinvertebrate and fish) data collection for the project was initiated in 1995 and continued annually through 2005. The purpose of the biomonitoring was to document changes in the aquatic vegetation, fish and macroinvertebrate populations of Walnut Creek as a result of the land use and management changes implemented in the watershed. Like the water quality analyses, a paired watershed approach was utilized, with

Squaw Creek serving as the control. Biomonitoring sampling sites are shown on Figure 3.

Artificial substrates for benthic macroinvertebrate colonization were placed in early summer and collected in late August/September to minimize community composition variability due to seasonality. Four substrates were placed at SQW2 and WNT2. In addition, 0.5 hours of qualitative (e.g. manually sampling from multiple habitat types that are present) sampling were done at each site. Four data metrics were calculated (Table 33) based on macroinvertebrate quantitative samples. Qualitative samples were used to reflect total taxa richness at each site that may not be reflected in populations collected from artificial substrates.

Electrofishing was conducted during mid to late summer with a single backpack electroshocker. A stretch of each stream at least 35 times the average width (Lyons 1992) was sampled at all sites. This distance was used to ensure sampling was performed on all major habitats present. The IDNR and UHL have been collecting fish community data from wadeable reference streams since 1994. A reference site is one with the least disturbed stream habitat**.** The IDNR used this reference site fish community data to construct a fish index of biotic integrity (FIBI) for wadeable Iowa streams. This FIBI consists of 11 data metrics (Table 34). When calculating the FIBI score for a stream, each metric result receives a score from 0 to 10 based on a comparison to the population of reference stream data gathered by the IDNR. The FIBI score is the sum of the 11 data metric scores, adjusted to range from 0 to 100. The greater the FIBI score, the "healthier" the stream.

Benthic Macroinvertebrates

Quantitative collections from Squaw Creek and Walnut Creek had poor macroinvertebrate colonization during the project. This makes metric evaluations difficult since the matrix used to establish a comparative index is done from a pool of sites around the state of Iowa. The matrix of comparable sites, referred to as reference sites (best currently available sites based on professional opinion), requires a specific level of colonization to result in data that is usable for constructing indices. The minimum level of colonization (~570 organisms/m^2) was occasionally not met by individual substrates collected from Walnut and Squaw creeks. As a result it was necessary to exclude certain years or lump multiple substrates

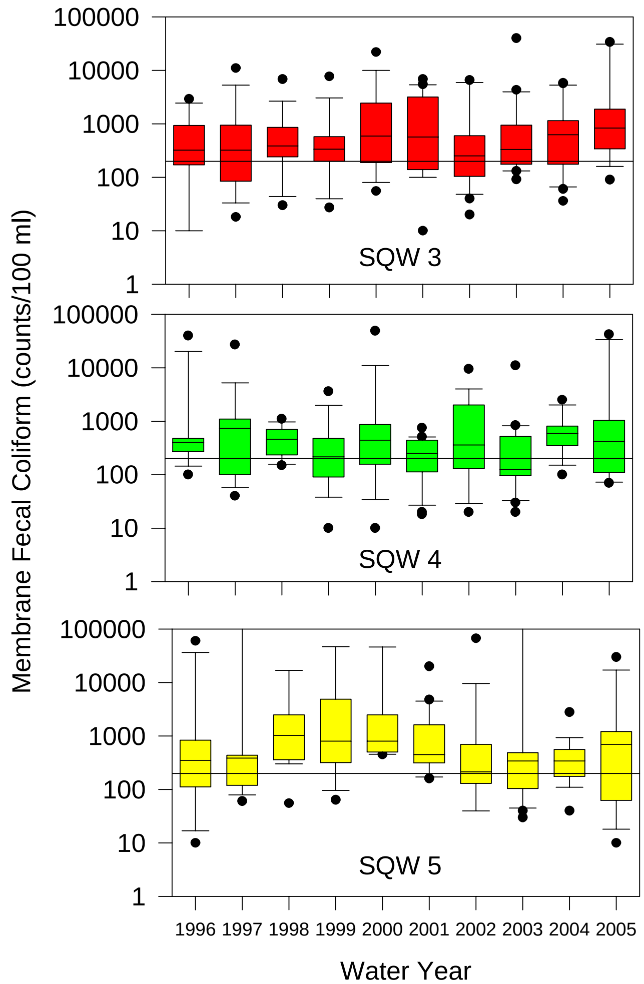


Figure 62. Box plots of fecal coliform concentrations by water year at Squaw Creek subbasin sites. A reference line for 200 counts/100 ml is indicated.

Table 28. Summary of median annual fecal coliform counts at 10 project monitoring sites. All concentrations in counts/100 ml.

Water Year	WNT1	WNT2	WNT3	WNT5	WNT6	SQW1	SQW2	SQW3	SQW4	SQW5
1996	1300	1100	105	136	70	310	510	325	320	300
1997	1900	685	380	320	125	750	770	250	740	385
1998	1200	1100	245	410	340	660	910	385	325	1035
1999	1100	710	215	230	185	570	1300	335	240	805
2000	750	770	400	780	260	680	820	590	520	800
2001	935	550	390	280	295	315	355	540	250	450
2002	270	440	120	305	280	270	315	250	250	165
2003	440	640	180	655	320	340	515	330	125	340
2004	890	915	470	1100	580	625	680	630	580	310
2005	560	680	320	820	380	560	555	840	410	700

together (within year) to have adequate population size for metric generation. It is suspected that low streamflows during late summer sampling periods created low velocity environments around the artificial substrates and resulted in poor representation of lotic communities.

Metric data from 1995-2005 (except Squaw Creek 2000 and 2002 data) were plotted for total taxa (quantitative and qualitative methods), and the four quantitative metrics: total taxa richness, EPT taxa richness, percent of dominant taxon, and percent of three dominant taxa (Figure 69). Taxa richness metrics (Total taxa and EPT) for Walnut Creek initially showed consistent improvement until 2001 after which metrics have steadily declined to lower levels than project inception. The metric measures of community balance (percentage of dominant taxa and percentage of three dominant taxa) showed similar positive trends with values decreasing until 2002, after which values have increased to levels at or higher than project inception levels.

Many of the positive changes in the macroinvertebrate community appeared to be driven by the habitat modification (addition of

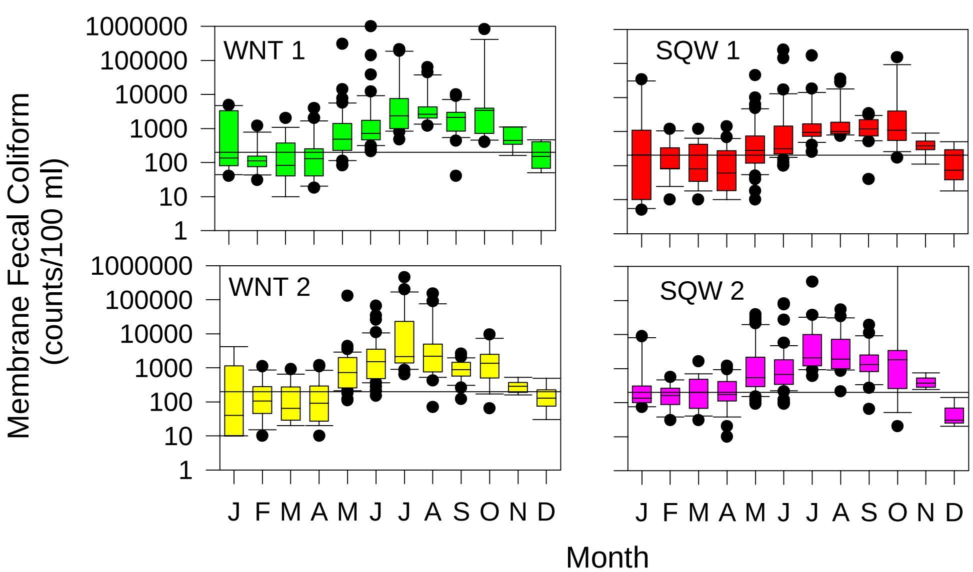


Figure 63. Box plots of fecal coliform concentrations by month at upstream and downstream monitoring sites in Walnut and Squaw creek watersheds. A reference line for 200 counts/100 ml is indicated.

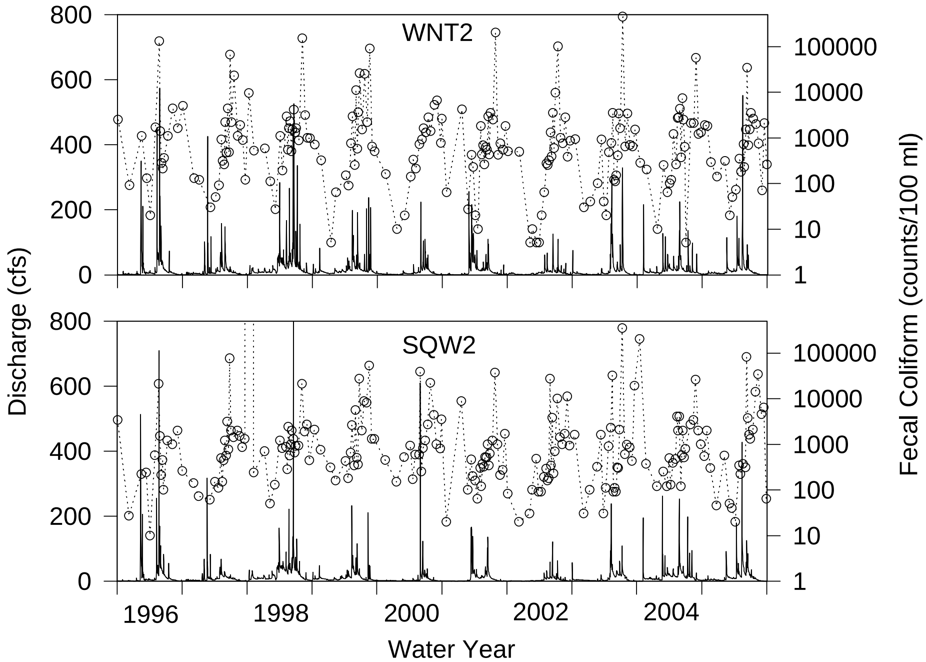


Figure 64. Relation of fecal coliform concentrations to discharge at WNT2 and SQW2.

coarse substrate) that occurred at the Walnut Creek sampling site. The habitat modification was most notably reflected in the proportion of the caddisfly family Hydropsychidae that began to appear in the quantitative samples. Prior to 2000, Hydropsychidae caddisflies never comprised more than ~5% of the community. In 2000, 2001, and 2004 they composed 33%, 19%, and 45%, respectively. In 2002, 2003, and 2005 they constituted 2%, 0%, and 0% of the population. Hydropsychidae caddisflies rely on net retreats to collect suspended food particles in the water column. In environments that do not have adequate flow to allow for this form of food collection, populations will not be prevalent. Hence their presence may function as a surrogate for adequate flow velocity at the sampling location. As a result it is likely many other aquatic taxa that require a "velocity threshold" (e.g. characterized as lotic) may also be selected against. Interestingly in the years after habitat modification, when hydropsychid caddisflies weren't present (e.g. 2003 and 2005), the EPT metric reflected the lowest levels observed for Walnut Creek during the study (Figure 69). The majority of EPT taxa collected over the course of the project rely on a consistent flow regime to provide food resources and adequate oxygenation (Merritt and Cummins, 1996).

Metric means were calculated for both streams and data did not show consistent trends in either watershed (Walnut Creek data shown in Table 35). Except for 2001 when large differences were evident, patterns of the four quantitative metrics have been similar between Walnut Creek and Squaw Creek.

Fish

Thirty-one species of fish from eight families were collected from Walnut Creek since 1995 (UHL, 2005). The fish community at Walnut Creek was dominated by minnows (Cyprinidae) and most of the minnow species collected are considered abundant to common in Iowa streams. Some of the most common minnows collected were the creek chub *Semotilus atromaculatus,* bluntnose minnow *Pimephales notatus,* central stoneroller *Campostoma anomalum*, and bigmouth shiner *Notropis dorsalis*. Sunfishes (Centrarchidae) were often found in Walnut Creek, but in small numbers. The three species found were bluegill

Table 29. Trend tests for changes in fecal coliform concentrations over time at project monitoring sites, adjusted for appropriate covariates as indicated.

Station	Covariates	Slope[1]	Prob.>t on slope estimate	r^2	Decrease over 10 years(%)
WNT2	Season Log(WNT2Q) Log(WNT1-FC) Log (SQW2-FC)	ns	.8564	0.75	ns
WNT2	Season Log(WNT2Q) Log(WNT1-FC)	ns	.4727	0.74	ns
WNT1	Season Log(WNT1Q)	- 0.029	.0896	0.49	48% Dec
WNT3	Season Log(WNT2Q)	ns	.6380	0.40	ns
WNT5	Season Log(WNT2Q)	+0.061	.0027	0.38	304% Inc
WNT6	Season Log(WNT2Q)	+0.061	.0011	0.44	317% Inc
SQW2	Season Log(SQW2Q) Log(SQW1-FC)	ns	.6019	0.57	ns
SQW1	Season Log(SQW2Q)	ns	.9192	0.40	ns
SQW3	Season Log(SQW2Q)	+ 0.043	.0247	0.39	170% Inc
SQW4	Season Log(SQW2Q)	ns	.9674	0.40	ns
SQW5	Season Log(SQW2Q)	- 0.048	.0766	0.23	67% Dec

Q = discharge, FC = fecal coliform concentration, ns = not statistically significant
[1]Slope expressed in change per year, if significant, log scale (Negative = decrease)

Lepomis macrochirus, green sunfish *Lepomis cyanellus*, and largemouth bass *Micropterus salmoides*. In 1998, bluegill comprised 67% of the sample; but they usually comprised less than 5% of the sample. All three sunfish species are widely distributed in Iowa and are most common in lakes and ponds, but can also be found in streams and rivers (Harlan et al. 1987). Sunfishes were most common in the pooled segment of Walnut Creek (WNT2), upstream of the USGS gage.

In 1998 and 1999, gizzard shad *Dorosoma cepedianum* comprised a large proportion of the Walnut Creek fish population (24% and 64% respectively; Clupeidae). Gizzard shad are considered tolerant of degraded environmental conditions (Tom Wilton, IDNR, personal communication). Gizzard shad were found in relatively low numbers in other years. Seven species of suckers (Catostomidae) have been collected from Walnut Creek. Suckers generally indicate favorable stream conditions because they are long lived and many sucker species are habitat specialists. The proportion of white suckers collected in 2002 was unusually large (21%) compared with previous years, but this greater percentage can probably be attributed to the overall low number of fish collected. Suckers were usually collected in low numbers.

Table 30. Summary of phosphorus concentrations at project monitoring sites. All concentrations in mg/l.

Statistic	SQW1	SQW2	SQW3	SQW4	SQW5	WNT1	WNT2	WNT3	WNT5	WNT6
n	106	109	73	65	79	105	110	68	78	71
mean	0.25	0.29	0.19	0.14	0.29	0.23	0.28	0.13	0.22	0.22
stdev	0.45	0.55	0.23	0.19	0.43	0.38	0.39	0.12	0.29	0.18
median	0.12	0.14	0.11	0.08	0.15	0.12	0.17	0.095	0.14	0.15
maximum	4	4.2	1.4	1.2	2.4	3	2.5	0.68	2	0.93

Twenty-two species of fish from six families have been collected from Squaw Creek (UHL, 2005). Similar to Walnut Creek, minnows usually comprised the majority of the population. Twelve species of minnows were collected; the most common minnows collected were the bluntnose minnow and the creek chub. Representatives from other families were collected from Squaw Creek, but they were usually few in number and did not contribute to a major proportion of the population. Two families that were collected at Walnut Creek, Percichthyidae (temperate bass) and Sciaenidae (drums) were not collected from Squaw Creek.

An FIBI score was calculated for all sampling events. Walnut Creek FIBIs ranged from 15 in 1995 to 40 in 1996 and 2002 (Table 36) whereas FIBI scores for Squaw Creek ranged from 21 in 2000 to 38 in 1997 (Table 37). An FIBI score of 40 for Walnut Creek in 2002 was unexpected, because only a small number of fish were collected at only one site. Walnut Creek would have had its highest FIBI score (43) in 2000 if it did not receive a ten point penalty for lesions found on approximately five percent of the fish. During all years, species tolerant of degraded environmental conditions made up a large proportion of the Walnut and Squaw creek fish communities. This was indicated by the low scores for the fish assemblage tolerance index metric (Tables 36 and 37). Sensitive species, such as the northern hog sucker *Hypentelium nigricans*, the blackside darter *Percina maculata*, or the slenderhead darter *Percina phoxocephal*), were rarely found in Walnut Creek. The number or sensitive species metric scores were zero during eight years, and less than two the other three years. All Squaw Creek scores for the number of sensitive species metric were zero, and all scores for the fish assemblage tolerance index were less than four (Table 26). One sensitive species, the northern hog sucker, has been found in Squaw Creek (1998). Three species of suckers (Catostomidae), usually in low numbers, have been collected from Squaw Creek. A Wilcoxon signed rank test was used to compare the FIBI scores for both watersheds; no significant difference was found between Walnut Creek and Squaw Creek FIBI scores (p = 0.66). FIBI scores for Walnut or Squaw Creek did not show any visual improvement or decline since 1995 (Figure 70).

Relation of FIBI to Reference Sites

The IDNR has calculated FIBI scores for approximately 100 reference stream sites in Iowa (Tom Wilton, IDNR, personal communication). Walnut Creek and Squaw Creek were compared to sites within this database from the same Ecoregion 47f (lacking numerous stable riffles and coarse substrate, and are in the Mississippi drainage). Thirty FIBI scores from 10 sites were found to meet these criteria (this group of 10 sites will hereafter be referred to as reference sites). Means and 95% confidence intervals for each metric and FIBI score were calculated for this group of 10 reference sites (Table 38). The Walnut or Squaw Creek FIBI or metric scores within the reference site 95% confidence interval were statistically similar to the reference site. Three of the 11 FIBIs calculated for Walnut Creek

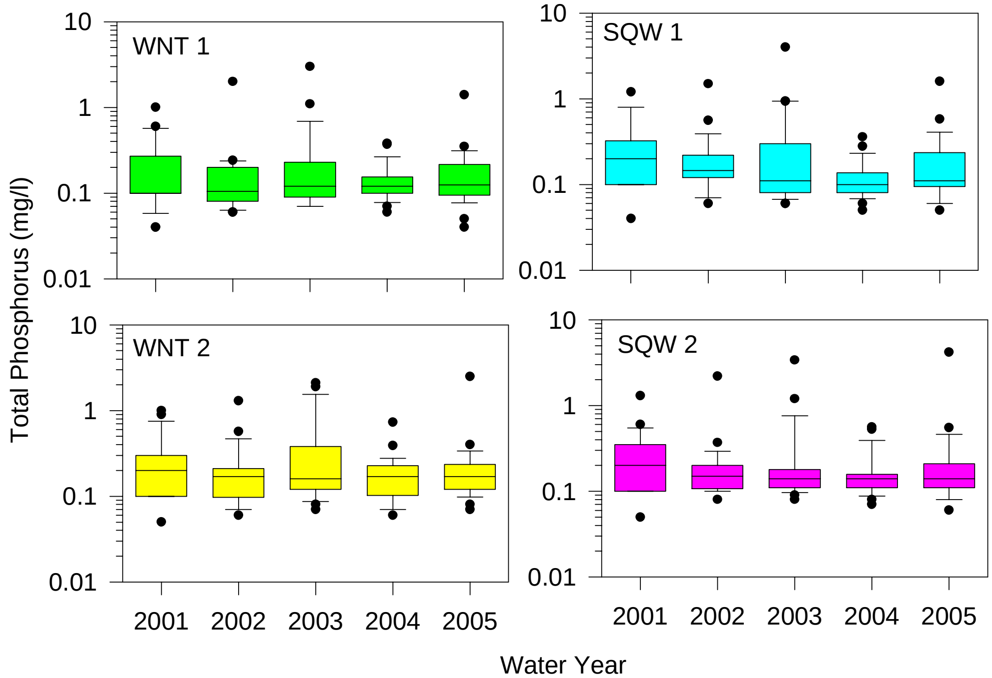


Figure 65. Box plot of phosphorus concentrations by water year at upstream and downstream sites in Walnut and Squaw creek watersheds.

were within or greater than the 95% confidence interval for reference sites, including scores for the number of sucker species metric, the proportion of omnivores metric, and the proportion of simple lithophilus spawners metric. Five of the scores for the number of sucker species metric were greater than the 95% confidence interval for reference sites. Two metrics calculated for Squaw Creek compared favorably to the IDNR reference sites: the proportion three dominant species and the proportion of omnivores.

Since 2002, the IDNR and UHL have also collected fish community data from stream and river sites randomly chosen from each ecoregion as part of the Regional Environmental Monitoring and Assessment Program (REMAP). Fish community data have been collected from 16 sites (19 samples) within Ecoregion 47f. Twelve of these sites are in the Mississippi drainage and also lack abundant stable riffles and coarse substrate (this group will hereafter be referred to as REMAP sites). Means and 95% confidence intervals for each metric and FIBI score were calculated.

It should be noted that the watershed areas of both the IDNR reference sites and REMAP sites were larger than the watershed areas for Walnut and Squaw Creeks (19.9 mi^2 and 18.3 mi^2, respectively). The mean and median watershed areas for the reference sites are 39.0 mi^2 and 86.0 mi^2, respectively. The mean and median watershed areas for the REMAP sites are 159.2 mi^2 and 74.8 mi^2, respectively. Although the watershed area for several streams in each group were considerably larger than Walnut or Squaw Creek, the results for the larger streams were left in each group to provide a larger population from which to calculate means and confidence intervals.

In Walnut Creek, nine of the 11 FIBI scores were within or greater than the 95% confidence interval for REMAP sites, whereas ten of the 11 scores for Squaw Creek were greater than or equal to the 95% confidence interval for REMAP sites.

The IDNR divides FIBI scores into quartiles and designates a score of 0 to 25 as poor, 26 to 50 as fair, 51 to 75 as good, and 76 to 100 as excellent. In streams that are classified as

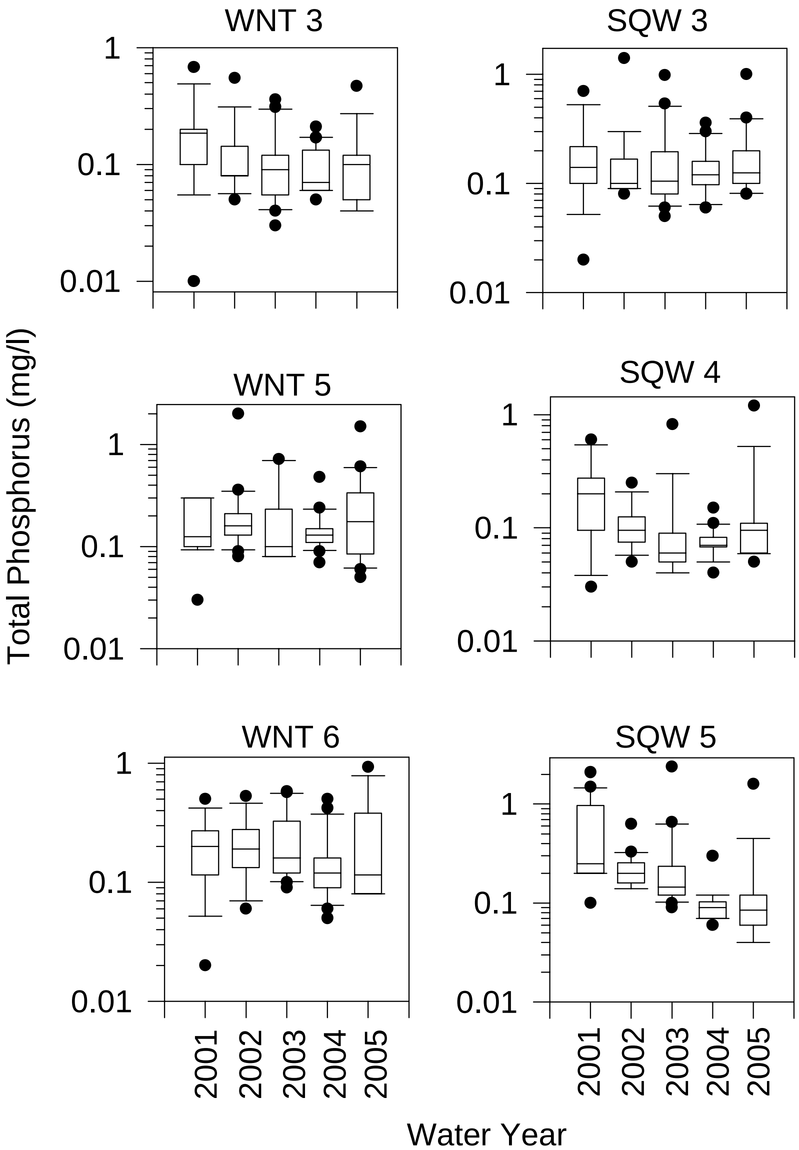


Figure 66. Box plots of phosphorus concentrations by water year at Walnut and Squaw creek subbasin sites.

poor, lower than average numbers of fish are present and the species found are usually short-lived or pioneering species that are tolerant of degraded stream conditions. A higher proportion of fish with deformities, eroded fins, lesions, and tumors is often found. Few species tolerant of degraded environmental conditions are present. In streams that are classified as fair, the fish community is usually dominated by species tolerant of degraded environmental conditions. Sucker species, sensitive species, top carnivores, and habitat specialists are often present, but in low numbers, and omnivores are usually more dominant. Most FIBIs calculated for Walnut Creek and Squaw Creek were considered fair. Walnut Creek received a poor FIBI score in 1995 and 2004 and Squaw Creek received a poor FIBI score in 2000. However, only one Walnut Creek site was sampled in 1995, possibly contributing to the low FIBI score for that year. The mean FIBI scores for reference and REMAP sites in Ecoregion 47F were similar to Walnut and Squaw Creeks and would also be classified as fair by the IDNR.

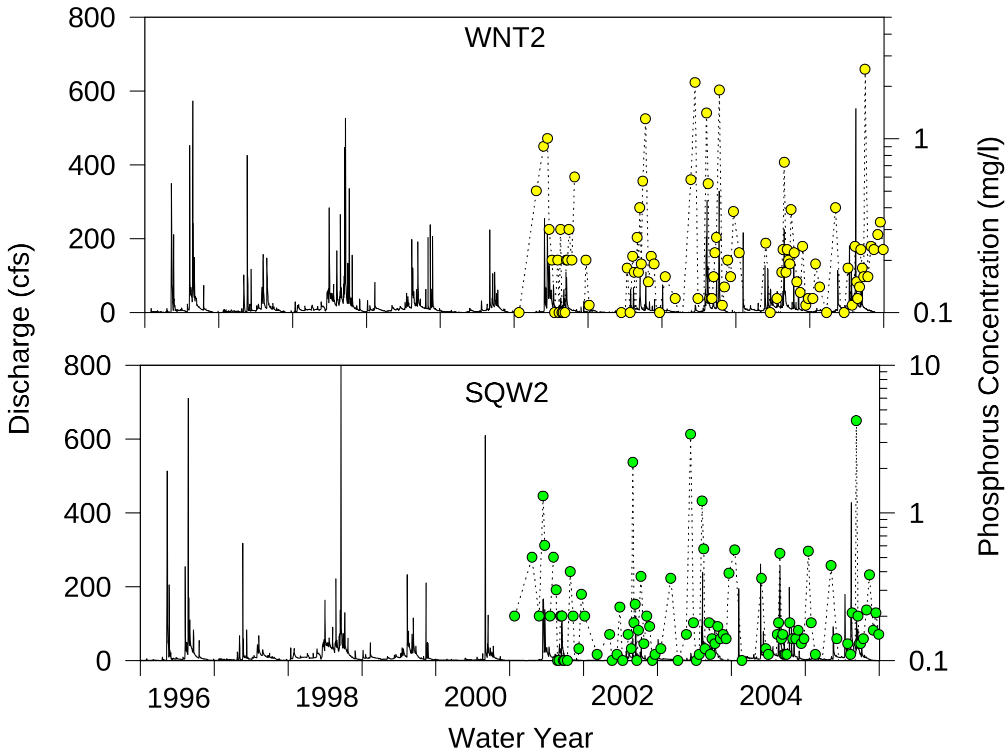


Figure 67. Relation of phosphorus concentrations to discharge at WNT2 and SQW2.

DISCUSSION

The Walnut Creek Watershed Monitoring Project was successfully completed by meeting the objectives established for the project. A comprehensive 10-year monitoring program was performed from Water Year 1995 to Water Year 2005 (Objective 1) to document water quality improvements due to land use and land management changes within the Walnut Creek watershed (Objective 2). Moreover, monitoring activities have greatly increased understanding of how land use changes in the watershed have affected short- and long-term water quality (Objective 3). By all measures, results from the Walnut Creek project met or exceeded these goals.

Project results indicated substantial differences in the detectability of NPS pollution improvements in Walnut Creek watershed. Nitrate concentrations showed greatest improvement during the monitoring program whereas improvements in other NPS pollutants, herbicides, fecal coliform, phosphorus and suspended sediment were not as apparent. Reasons for differences in the detectability of change are diverse and are related to the individual NPS pollutant and the manner by which the pollutant is delivered to streams. In the case of nitrate and suspended sediment, the rate of change and the lag time for observing change are further related to historical conditions that predate the initiation of the Neal Smith Refuge and the Walnut Creek project. The following discussion focuses on detecting change in NPS pollutants during the Walnut Creek project and the timeframe needed to observe water quality improvements.

Detecting Changes in Nitrate

Evidence from the Walnut Creek project suggests that prairie restoration in row crop-dominated watersheds can reduce nitrate

Table 31. Summary of median annual phosphorus concentrations at 10 project monitoring sites for water years 2000 to 2005. All concentrations in mg/l.

Water Year	SQW1	SQW2	SQW3	SQW4	SQW5	WNT1	WNT2	WNT3	WNT5	WNT6
2001	0.20	0.20	0.15	0.20	0.20	0.10	0.20	0.17	0.14	0.20
2002	0.15	0.15	0.11	0.09	0.20	0.11	0.17	0.10	0.16	0.19
2003	0.11	0.14	0.11	0.06	0.15	0.12	0.16	0.09	0.10	0.16
2004	0.10	0.14	0.12	0.07	0.09	0.12	0.17	0.07	0.13	0.12
2005	0.11	0.14	0.13	0.10	0.09	0.13	0.17	0.10	0.18	0.12

concentrations in streams and lower nitrate export. Nitrate concentrations decreased in the Walnut Creek watershed and subbasins in response to prairie restoration up to 3.4 mg/l over ten years, and nitrate loading rates from lower Walnut Creek watershed were one-half those of row crop areas. Data from this watershed study are consistent with investigations conducted at the plot scale that indicated significantly lower drainage from perennial grass systems (prairie, CRP) and lower nitrate concentrations in drainage water (Randall et al., 1997; Brye et al., 2001). Plot studies point to the extended growing season of perennial systems to increase annual ET, reduce accumulation of soil water and increase nitrate uptake and plant assimilation compared to corn and soybeans (Randall et al., 1997). Data from the Walnut Creek project extends the plot scale results to a watershed scale to confirm that perennial cover in watershed reduces nitrate losses. Monitoring results from the Walnut Creek project further confirm results from modeling studies that have suggested that perennial cover placed in agricultural watersheds can reduce pollutant loads (Nassauer et al., 2002; Coiner et al., 2001, Vache et al., 2002).

While clear that nitrate concentrations and loads may be reduced due to prairie restoration, it is equally evident that converting grasslands back to row crop increases nitrate concentrations in streams. In Squaw Creek watershed and upper Walnut Creek, areas formerly enrolled in CRP were converted back to row crop agriculture and stream nitrate concentrations increased up to 11.6 mg/l in ten years. Randall et al. (1997) summed up the effects of tilling grasslands in this way: "Tilling the land for crop production leads to greater mineralization of soil organic matter. Coupled with N applications (both inorganic and organic), improved subsurface drainage through tile lines, and less efficient use of water by shallower rooted, shorter growing season crops, we can expect modern farming practices to contribute to higher losses of NO_3 to both surface and groundwater" (p. 1247). Row crop agriculture increases both water loss to stream through reduced ET and increased runoff, and increases nitrate losses from mineralization of soil N and fertilizer application. The ability of grasslands (e.g., prairie) to process both water and nutrients compared to row crops thereby translates to reduced nitrate loads to streams. Results from the Walnut Creek project confirmed these effects by documenting the relation between land use conversion and the corresponding water quality trends.

The amount of change in nitrate concentrations in both watersheds (10 sites) was significantly related to the degree of change in row crop land cover that occurred from 1990 to 2005 (Figure 71). Although the comparisons of nitrate and row crop changes across time involved somewhat different time periods, the strong link between row crop land cover and nitrate concentrations was evident. While converting row crop to native prairie at the Neal Smith NWR reduced the amount of row crop in the various watershed areas and reduced stream nitrate, converting CRP grass back to row crop in Squaw Creek increased

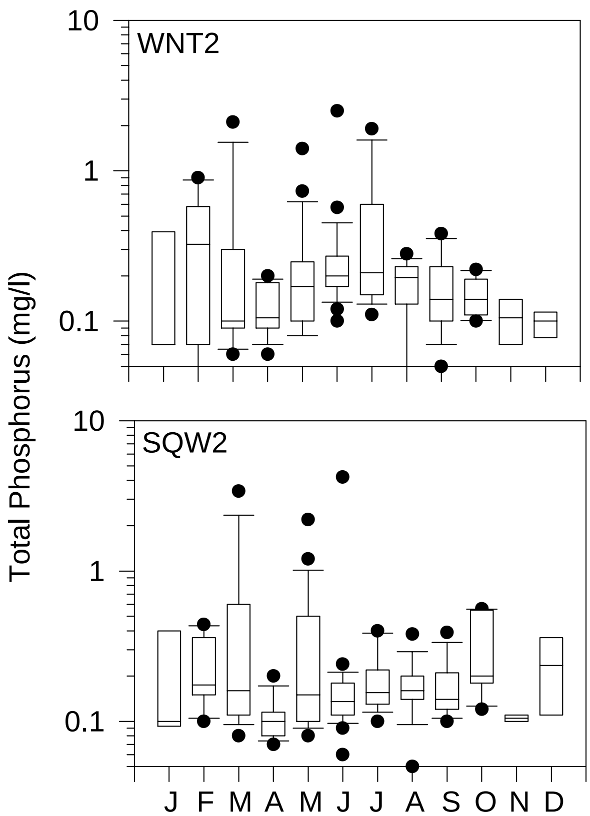


Figure 68. Box plots of phosphorus concentrations by month at WNT2 and SQW2.

the amount of row crop and greatly increased stream nitrate. The regression equation suggests that for every change in row crop of 10 percent in Walnut and Squaw Creek watersheds, a change of 3.5 mg/l nitrate may be expected to occur in a 10-year monitoring period. The slope of the relation (approximately 0.2) was higher than the relation of row crop to mean annual stream nitrate concentrations reported by Schilling and Libra (2000). They stated that mean annual nitrate concentrations in Iowa's streams could be approximated by multiplying a watershed's row crop percentage by 0.1. However, Schilling and Libra (2000) also noted that the slope of nitrate concentrations versus row crop increased with decreasing watershed size. Thus an increased slope of nitrate concentration changes to row crop changes was consistent with the previous findings because the project watershed areas are fairly small. Moreover, this project assessed the changes over time whereas the Schilling and Libra (2000) work considered long-term average values of stream nitrate. The steeper slope of stream nitrate versus row crop land use measured in this project suggests that changes in stream nitrate concentrations may respond quite rapidly to changes in row crop land cover in small watersheds.

Recent modeling studies (Worrall and Burt, 1999, 2001) examined the effects of land use change on nitrate concentrations and wondered, "how much land use change is safe"? They used export coefficients and structural models to show that plowing up permanent pastures rapidly releases soil nitrogen to streams and that this process dominates over sequestration of soil N from converting lands back to permanent or temporary grassland. In this study, the rate of nitrate increase following grassland conversion to row crop was greater than the rate of nitrate decrease following conversion of row crop land to prairie. Despite similar degree of land use change in Walnut and Squaw creek subbasins showing the greatest nitrate concentration changes, ranging from decreasing row crop 27 to 31 percent in Walnut subbasins to increasing row crop 26 to 28 percent in Squaw subbasins, the rate of increase in nitrate concentration in Squaw Creek was more than double the rate of decrease in Walnut Creek. However, this type of comparison is complicated by several factors. Some of the difference in rate of change may be traced to more gradual implementation of prairie restoration in Walnut Creek compared to more rapid plowdown of CRP grassland in Squaw Creek. Furthermore, tile drainage contributions from areas converted back to row crop would increase the rate by which changes in water quality could be observed. In contrast, most drainage tiles located in Walnut Creek prairie restoration plots were plugged or pulled wherever encountered by refuge staff. However, both watersheds have similar hydrogeology and placements of land use changes in the watersheds were rather

Table 32. Trend tests for changes in phosphorus concentrations over time at project monitoring sites, adjusted for appropriate covariates as indicated.

Station	Covariates	Slope (log scale) (Negative = decrease)	Prob.>t on slope estimate	r^2	Decrease over 10 years (%)
WNT2	Season Log(WNT1-TP)	ns	.5161	0.60	ns
WNT1	Season Log(WNT1Q)	ns	.3274	0.26	ns
WNT3	Season Log(WNT2Q)	ns	.1735	0.15	ns
WNT5	Season Log(WNT2Q)	ns	.1069	0.23	ns
WNT6	Season Log(WNT2Q)	ns	.9122	0.26	ns
SQW2	Season Log(SQW2Q) Log(SQW1-TP)	ns	.4804	0.46	ns
SQW1	Season Log(SQW2Q)	ns	.9391	0.23	ns
SQW3	Season Log(SQW2Q)	+ 0.069	.0098	0.34	385%
SQW4	Season Log(SQW2Q)	ns	.1143	0.27	ns
SQW5	Season Log(SQW2Q)	- 0.048	<.0001	0.32	95%

Q = discharge, FC = fecal coliform concentration, ns = not statistically significant
ns = not significant above 0.1

piecemeal based on field boundaries and property ownership. Neither watershed had land use changes located for maximum water quality effect. Thus, while differences in implementation may account for some of the differences in the rate of nitrate concentration change in streams, project monitoring data support modeling that suggests nitrate concentrations more rapidly increased following conversion of grassland to crops than decreased following conversion back to grassland.

The rate of change, or the lag time needed for observing water quality change, is also governed by the hydrogeology of the watersheds. Uplands in Walnut and Squaw creek watersheds consist of loess mantling pre-Illinoian till, whereas their floodplains are comprised of mainly silty alluvium. In the absence of tile drainage, nitrate leached from soils moves with shallow groundwater to discharge to streams. In the Walnut and Squaw creek watersheds dominated by low permeability glacial materials and glacial-derived alluvium, groundwater flow velocities are slow. Hence the time needed for observing changing nitrate concentrations in streams resulting from land use change is dependent on the velocity of groundwater flow to deliver nitrate to streams.

In the uplands of Walnut Creek watershed, a groundwater flow model was recently completed of a 7.8 ha catchment located in the western portion of the Neal Smith refuge

Table 33. Description of benthic macroinvertebrate metrics.

Total Taxa Richness-Reflects the health of the community through a measurement of the variety of taxa (mutually exclusive taxa) present. Generally, there is an increase in taxon richness with increasing water quality, habitat diversity, and habitat suitability (Plafkin 1989).

Ephemeroptera, Plecoptera, Trichoptera (EPT) index-The EPT taxa metric is the number of distinct taxa within the generally pollution sensitive orders of Ephemeroptera, Plecoptera, and Trichoptera (Mayfly, Stonefly, and Caddisfly, respectively). An increasing value represents a higher number of EPT taxa and improved water quality (Plafkin 1989).

Percentage of Dominant Taxon (PDT)- This metric represents the percentage of the sample that is composed of the dominant taxon. The larger the percentage, the more common the taxon is in the sample. Higher percentages reflect an unbalanced community.

Percentage of Three Dominant Taxa (P3DT)- As with Percentage Dominant Taxon this metric represents the proportion of the community that is composed of the dominant taxa, however, instead of considering one taxon, three taxa are considered. In comparison to percent of dominant taxon this metric does not have as much variability and can provide for a more discriminatory evaluation. Higher percentages reflect a less balanced community.

(Weisbrod, 2005). In the model, the hydraulic conductivity of the upland loess, weathered till and alluvium was estimated to be 0.17 m/day from slug test and tracer test data (Weisbrod, 2005). The hydraulic conductivity of the pre-Illinoian till was estimated to be an order of magnitude lower (Schilling and Thompson, 1999). From the calibrated groundwater flow model, the average travel time for a particle entering the groundwater system, flowing downslope in the catchment and leaving the model area was estimated to be approximately 75 years (Weisbrod, 2005). The model was also run to predict the amount of time needed to reduce nitrate concentrations in upland monitoring wells due to dilution from low nitrate recharge flowing through restored prairie. Results suggested that the rate of nitrate reduction would be on the order of 1.2 to 2.2 mg/l every 10 years in the upland wells. Interestingly, this rate of nitrate reduction in upland settings is very similar to those measured in the Walnut Creek watershed and subbasins. Results from the groundwater flow model imply that decades are needed for groundwater in upland catchments to discharge to streams and for measuring groundwater nitrate concentration reductions over time.

Overall, the distance that groundwater would have flowed during the 10-year monitoring project can be estimated by

$$V = -K(dh/dl)/n$$

where v is the average linear velocity (m/s), K is the hydraulic conductivity (m/s), dh/dl is the hydraulic gradient (dimensionless) and n is the porosity. Assuming the K of the upland loess to be 0.2 m/day (0.66 feet/day), the gradient to be 0.04 and the porosity to be 0.3, the estimated groundwater flow velocity is 0.027 m/day (0.089 feet/day). The distance that groundwater would have flowed in 10 years is 98.6 m, or 323 feet. Land use changes located at a distance beyond approximately 325 feet from a stream would not be expected to have an effect on stream water quality during the 10 years of monitoring.

Table 34. Description of metrics used to calculate the Index of Biotic Integrity (IBI) for fish.

Metric	Explanation/Desired Condition
Native Fish Species Richness	More species implies greater habitat complexity and favorable stream conditions.
Proportion of Simple Lithophilic Spawners	Simple lithophilic spawners are sensitive to sedimentation, and require clean gravel/cobble substrate for reproduction. Therefore, a greater percentage of lithophilic spawners implies better benthic conditions.
Proportion of Fish as Omnivores	Omnivores consume a variety of plant and animal material, and therefore are less sensitive to environmental degradation that causes changes in the food base. A higher percentage of omnivores implies unfavorable environmental conditions.
Proportion of Benthic Invertivores	Benthic invertivores are often specialists, and therefore are sensitive to environmental degradation that causes a change in the food base. A higher percentage implies favorable stream conditions.
Proportion of Three Dominant Species	A higher percentage implies unfavorable stream conditions.
Proportion of Top Carnivores	Top carnivores are generally longer-lived fish that need stable environmental conditions and food base. A higher percentage implies favorable stream conditions.
Number of Sucker Species	Suckers are generally longer-lived species that are intolerant of degraded conditions. A greater number implies favorable stream conditions.
Number of Benthic Invertivore Species	Benthic invertivores are often specialists, and therefore are sensitive to environmental degradation that causes a change in the food base. A greater number of benthic invertivore species implies favorable stream conditions.
Number of Sensitive Species	A greater number of species sensitive to degraded environmental conditions implies favorable water quality and habitat conditions.
Fish Assemblage Tolerance Index	Each fish species receives a tolerance value, the index is based on the sum of tolerance values for a site. A higher index value implies unfavorable environmental conditions.
Adjusted Catch per Unit Effort	A greater density of fish implies more favorable stream conditions. Tolerant fish species are not included in this metric.

However, tile drainage can greatly accelerate the transfer of water quality effects to streams following land cover change. In upland areas of Walnut Creek, prairie restoration has largely occurred in areas where tile drainage is minimal or has been removed. Tiles existing in these areas are not maintained and have been allowed to become blocked with sediment and vegetation. Hence, transferring water quality benefits from upland prairie

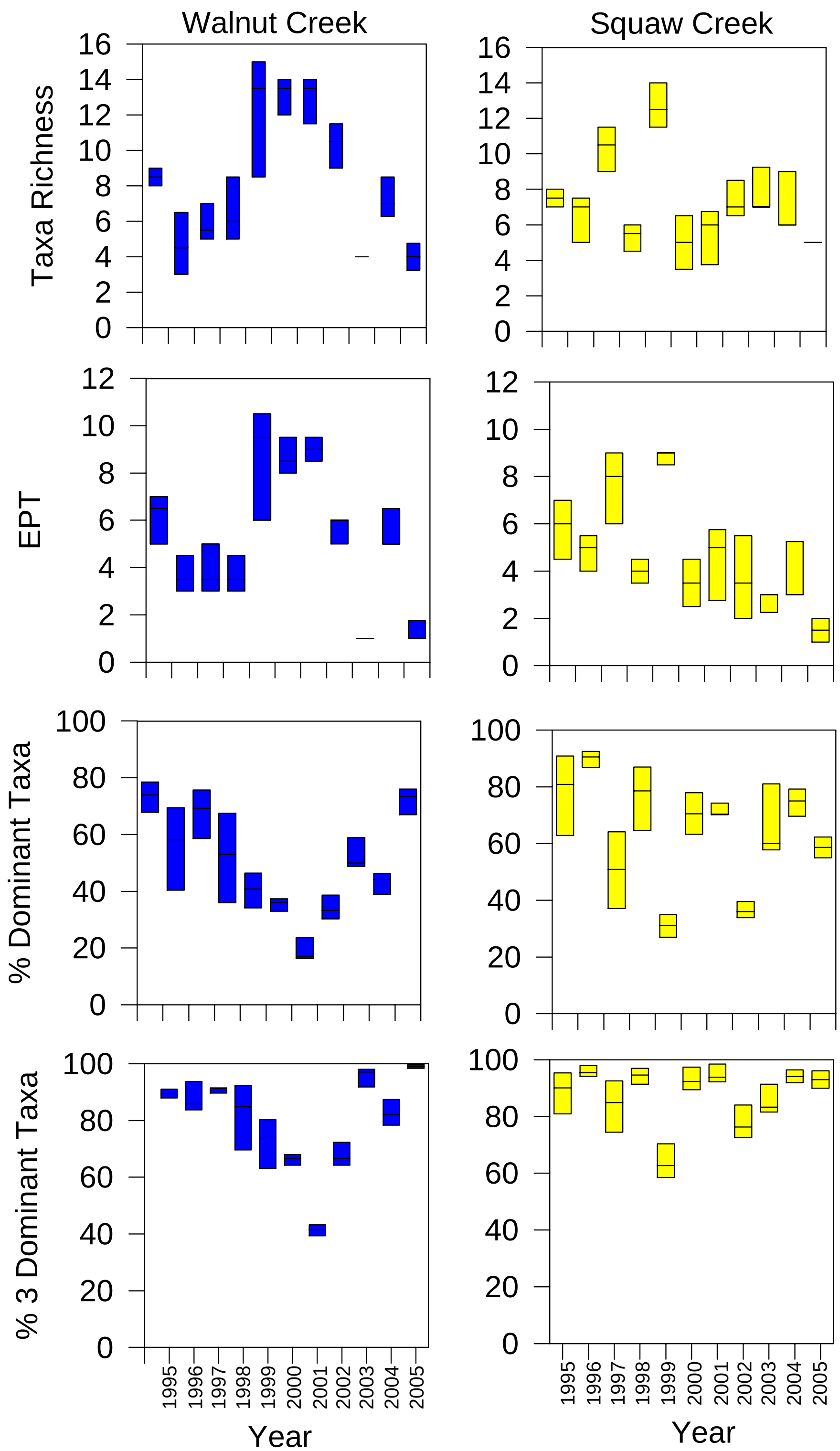


Figure 69. Box plots of annual benthic macroinvertebrate metrics in Walnut and Squaw creek watersheds.

Table 35. Walnut Creek mean metric values-ANOVA between year comparisons (years with no letters in common are significantly different).

Calander Year	Total Taxa	EPT Taxa	Percent of dominant taxon	Percent of three dominant taxa
2005	4.0(B)	1.3(B)	71.7(AB)	99.0(B)
2004	7.3(AB)	5.7(AB)	42.7(AB)	82.8(AB)
2003	8.0(AB)	2.0(AB)	51.0(AB)	95.1(AB)
2002	10.3 (AB)	5.5 (AB)	34.4 (AB)	68.3 (A)
2001	12.75 (A)	9.0 (A)	34.2 (B)	70.6 (AB)
2000	13.0 (A)	8.75 (A)	35.1 (AB)	66.1 (A)
1999	11.75 (AB)	8.25 (A)	40.2 (AB)	71.7 (AB)
1998	6.75 (AB)	3.75 (AB)	51.8 (AB)	81.0 (AB)
1997	6.0 (AB)	4.0 (AB)	67.1 (AB)	90.5 (AB)
1996	4.75 (AB)	3.75 (AB)	54.9 (AB)	88.8 (AB)
1995	8.5 (AB)	6.0 (AB)	73.1 (A)	89.4 (AB)

restoration to streams will likely be governed by groundwater velocity. In contrast, headwater regions of both Walnut and Squaw Creek watersheds are tile-drained. Most stream initiation points in both watersheds occur as tile outlets from headwater catchments, with first-order streams often beginning at road crossings with tile drainage discharging into a road culvert. In these tile-drained upland areas, land cover can have a proportionally large effect on water quality since subsurface water bypasses slow groundwater transport and is rapidly directed to streams via tiles. The effects of headwater contributions on stream water quality were particularly evident in Walnut Creek watershed. Nitrate concentrations in surface water generally start elevated in tile-drained headwater regions and remain elevated, with some dilution occurring as water flows through the watershed, generally dominating the water contributions from the lower portion of the watershed containing the restored prairie. In Squaw Creek, the rapid change in stream nitrate concentrations measured at subbains SQW4 and SQW5 may have been accelerated by tile drainage, although the actual extent of tile drainage in these subbasins is unknown.

In the floodplain, a similar assessment of travel distance can be made, though with less certainty due to stratigraphy. Investigations conducted in the riparian zone of Walnut Creek indicated that the hydraulic conductivity of the silty alluvium (Camp Creek, Roberts Creek and Gunder members) ranged from 5.6 x 10^{-5} m/s for the Camp Creek Member (post settlement material) to 1.7 x 10^{-6} m/s for the Roberts

Table 36. Data metric results (FIBI scores), unadjusted FIBI scores and adjusted FIBI scores for Walnut Creek.

Metric	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Native Fish Species Richness	8 (3.7)	17 (7.7)	12 (5.5)	13 (5.9)	14 (6.4)	15 (6.9)	11 (5.0)	12 (5.5)	15 (6.9)	10 (4.6)	13 (5.6)
Number of Sucker Species	2 (4.7)	3 (7.0)	4 (9.4)	3 (7.0)	3 (7.0)	2 (4.7)	1 (2.3)	1 (2.3)	4 (9.4)	1 (2.3)	1 (2.3)
Number of Sensitive Species	0 (0)	0 (0)	0 (0)	1 (1.3)	1 (1.4)	0 (0)	0 (0)	1 (1.3)	0 (0)	0 (0)	0 (0)
Number of Benthic Invertivore Species	1 (1.2)	1 (1.3)	1 (1.3)	3 (3.8)	3 (3.8)	3 (3.8)	2 (2.5)	3 (3.8)	2 (2.5)	2 (2.5)	3 (3.8)
Proportion of Three Dominant Species (%)	78.5 (4.2)	51.4 (9.5)	72.3 (5.4)	67.3 (6.4)	79.7 (4.0)	68.9 (6.1)	69.6 (6.0)	59.5 (7.5)	67.5 (6.3)	75.4 (4.8)	69.8 (5.9)
Proportion of Fish as Benthic Invertivores (%)	0.6 (0.2)	2.3 (0.7)	3.5 (1.1)	2.4 (0.8)	3.6 (1.1)	12.1 (3.9)	12.5 (4.0)	14.3 (4.6)	0.7 (0.2)	1.6 (0.5)	8.1 (2.6)
Proportion of Fish as Omnivores (%)	75 (0.7)	29.2 (7.3)	16.3 (9.1)	40.9 (5.6)	72.1 (1.1)	12.6 (9.6)	29.3 (7.2)	33.3 (6.7)	45.8 (4.9)	25.4 (7.8)	49.6 (4.3)
Proportion of Fish as Top Carnivores (%)	0 (0)	0.6 (3.6)	0 (0)	0 (0)	0.4 (3.0)	0 (0)	0 (0)	0 (0)	0.3 (2.7)	0 (0)	0 (0)
Proportion of Fish as Simple Lithophilus Spawners (%)	0 (0)	2.3 (1.5)	0 (0)	2.4 (1.5)	2.8 (1.8)	10.7 (6.8)	12.1 (7.7)	9.5 (6.1)	0.3 (0.2)	0.8 (0.5)	7.4 (4.7)
Fish Assemblage Tolerance Index	9.4 (0.9)	7.9 (3.3)	8.5 (2.3)	7.6 (3.9)	9.3 (1.1)	7.9 (3.4)	8.5 (2.4)	6.8 (5.1)	8.9 (1.8)	9.5 (0.8)	9.2 (1.4)
Adjusted CPUE	10 (1)	15.8 (1.6)	9.1 (0.9)	22.2 (2.2)	7.3 (0.7)	19.3 (1.9)	20.2 (2.0)	10.4 (1.0)	27.2 (2.7)	2.6 (0.3)	8.8 (0.9)
Unadjusted FIBI Score:	15	40	32	35	29	43	36	40	34	22	29
Proportion (%) of Fish with DELTs (Adjustment)	0 (0)	0 (0)	0 (0)	0.5 (0)	0.8 (0)	4.9 (-10)	0.7 (0)	0 (0)	0 (0)	1.6 (0)	0 (0)
Low Numbers of Fish Adjustment	No	No	No	No	No	No	No	No	No	No	No
Adjusted FIBI Score:	15	40	32	35	29	33	36	40	34	22	29
DNR Rating*	Poor	Fair	Fair	Fair	Fair	Fair	Fair	Fair	Fair	Poor	Fair

CPUE = catch per unit effort; DELT = deformities, eroded fins, lesions, or tumors
*0-25 = poor, 26-50 = fair, 51-75 = good, 76-100 = excellent

Creek Member (Schilling et al., 2004). The hydraulic gradient in the floodplain also varies in the floodplain and increases substantially near incised Walnut Creek due to the effects of channel incision. The average hydraulic gradient across the entire floodplain is approximately 0.02 (Schilling et al., 2004). Assuming a porosity of 0.3, the average linear groundwater flow velocity within the Camp Creek member in the floodplain was estimated to be 1.04 ft/day, whereas the average linear groundwater flow velocity in the Roberts Creek member was considerably less (0.03 ft/day). The distance that groundwater would have flowed in the floodplain in 10 years ranges between 3796 feet in the Camp Creek Member to 110 feet in the Roberts Creek Member. Considering that the water table often drops below the Camp Creek unit during much of the year, the estimated travel distance based solely on the Camp Creek Member is probably high. The actual travel distance likely ranges between 110 to 3800 feet and will vary according to the stratigraphy of the floodplain sediments. Thus, effects of floodplain land use changes on stream water quality are difficult to quantify, but with the width of the Walnut Creek floodplain generally varying between 600 and

Table 37. Data metric results (FIBI scores), unadjusted FIBI scores and adjusted FIBI scores for Squaw Creek.

Metric	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Native Fish Species Richness	14 (6.8)	9 (4.4)	11 (5.3)	15 (7.3)	11 (5.3)	10 (4.8)	13 (6.3)	12 (5.8)	10 (4.8)	13 (6.3)	13 (6.3)
Number of Sucker Species	2 (5.0)	0 (0)	1 (2.5)	3 (7.4)	1 (2.5)	1 (2.5)	1 (2.5)	1 (2.5)	1 (2.5)	1 (2.5)	2 (5.0)
Number of Sensitive Species	0 (0)	0 (0)	0 (0)	2 (2.8)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Number of Benthic Invertivore Species	1 (1.4)	1 (1.4)	2 (2.7)	3 (4.0)	1 (1.4)	1 (1.4)	2 (2.7)	1 (1.4)	1 (1.4)	1 (1.4)	1 (1.4)
Proportion of Three Dominant Species (%)	64.8 (7.3)	62.3 (7.8)	48.7 (10)	49.7 (10.0)	60.9 (8.1)	76.9 (4.8)	52.1 (9.9)	59.3 (8.4)	67.1 (6.8)	54.1 (9.5)	62.4 (7.8)
Proportion of Fish as Benthic Invertivores (%)	3.3 (1.1)	5.8 (2.0)	14.9 (5.0)	2.2 (0.7)	2.3 (0.8)	2.0 (0.7)	1.5 (0.5)	3.0 (1.0)	11.5 (3.9)	2.9 (1.0)	6.6 (2.2)
Proportion of Fish as Omnivores (%)	50.6 (4.3)	17.4 (9.2)	18.6 (9.1)	60.1 (2.9)	27.8 (7.7)	47.8(4.8)	30.9 (7.3)	32.6 (7.0)	41.9 (5.6)	26.4 (7.9)	28.9 (7.5)
Proportion of Fish as Top Carnivores (%)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Proportion of Fish as Simple Lithophilus Spawners (%)	0 (0)	0 (0)	0.8 (0.5)	1.1 (0.7)	0 (0)	0 (0)	0.2 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)
Fish Assemblage Tolerance Index	8.8 (1.9)	8.4 (2.5)	7.5 (3.9)	7.9 (3.3)	8.1 (3.0)	8.4 (2.6)	8.2 (2.8)	8.2 (2.9)	8.6 (2.2)	7.6 (3.8)	7.8 (3.5)
Adjusted CPUE	44.7 (4.5)	10.1 (1.0)	30.6 (3.1)	13.2 (1.3)	12 (1.2)	19.1 (1.9)	45.6 (4.6)	23.1 (2.3)	20.9 (2.1)	51.4 (5.1)	21.4 (2.1)
Unadjusted FIBI Score:	29	26	38	37	27	21	33	28	27	37	33
Proportion (%) of Fish with DELTs (Adjustment)	0 (0)	0 (0)	0 (0)	2.7 (-5)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Low Numbers of Fish Adjustment	No	No	No	No	No	No	No	No	No	No	No
Adjusted FIBI Score:	29	26	38	32	27	21	33	28	27	34	33
DNR Rating*	Fair	Fair	Fair	Fair	Fair	Poor	Fair	Fair	Fair	Fair	Fair

CPUE = catch per unit effort; DELT = deformities, eroded fins, lesions, or tumors
*0-25 = poor, 26-50 = fair, 51-75 = good, 76-100 = excellent

1200 feet, it is suspected that water quality improvements from all but the most recent prairie plantings occurring on the Walnut Creek floodplain have probably arrived at the stream and are impacting watershed water quality. This would be consistent with dilution of upstream nitrate concentrations occurring as stream water moves through the lower portion of the Walnut Creek watershed.

In Walnut Creek, tile drainage appears to be less of a factor for floodplain settings than upland areas. Schilling and Wolter (2000) mapped the main channel of Walnut Creek between the two gaging stations and found 52 drainage tiles entering the channel, although not all were flowing. In May 1999, nitrate loads from 19 of the tiles were measured and tile contributions were found to contribute to four percent of the daily nitrate export from the watershed (Schilling and Wolter, 2001). Nitrate concentrations were less than 1 mg/l in samples collected from tiles draining restored prairie areas, but were greater than 10 mg/l in tiles draining row crop lands.

Overall, in the absence of tile drainage, land use changes occurring in the floodplain are more likely to have an impact on short term water quality than those associated with upland

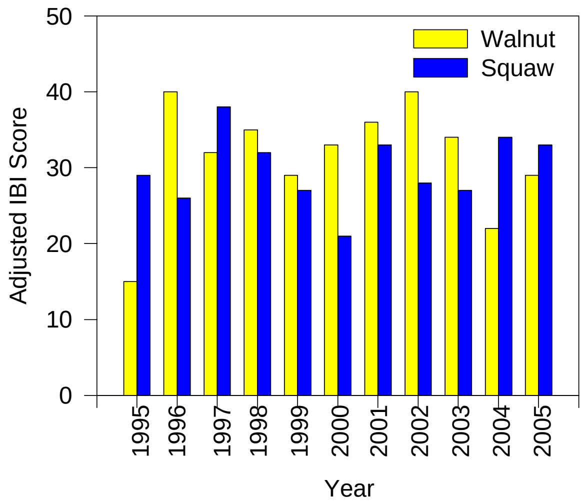


Figure 70. Summary of annual Walnut and Squaw creek IBI scores.

settings. This is consistent with the results of the 2001 synoptic survey that concluded that prairie restoration or other similar BMPs concentrated along riparian corridors can play a role in improving water quality at a watershed scale under baseflow conditions (Schilling, 2002).

One final note on the lag time for observing changes in groundwater delivery of nitrate to streams is considered. The mean residence time for groundwater in a groundwatershed (the average amount of time needed for groundwater to "turn over" in a groundwater catchment area) can be approximated by (Haitjema, 1995):

$$F(T) = nH/N$$

where n is the aquifer porosity, H is the saturated aquifer thickness, and N is the areal recharge rate due to precipitation. For Walnut Creek watershed, n is assumed to be 0.3, H is estimated to be 20 feet and N is equal to the long-term average baseflow in Walnut Creek (0.43 feet or 5.1 inches). The mean residence time for groundwater in Walnut Creek is estimated to be approximately 14 years. Considerable uncertainty lies in the estimate of H, but it was derived as an approximation of the thickness of saturated loess and oxidized till in upland settings, and a midrange estimate of saturated alluvium in the floodplains. In actuality, H could be less than 10 feet in sloping bluffs where pre-Illinoian till outcrops, or greater than 40 feet in some floodplain settings. The residence time is essentially the mean of a cumulative frequency distribution of travel times in the watershed and would imply that, on average, 14 years is needed for groundwater to drain from the watershed, with some groundwater draining faster to streams and some draining much slower. As discussed above, it is likely that the travel time for groundwater in floodplains is less than the 14-year average, but the travel time for groundwater located in uplands is probably much greater than 14 years. Hence, the amount of time needed to detect the water quality changes due to groundwater from all areas of the watershed is ultimately on the order of several decades.

Detecting Changes in Runoff NPS Pollutants

While nitrate concentrations and loads showed improvements in Walnut Creek watershed, herbicides, fecal coliform and phosphorus did not show consistent evidence for water quality changes. It is suspected that the main reason for this difference can be traced to the manner in which NPS pollutants are delivered to streams. Unlike nitrate that is primarily discharged to streams with groundwater seepage and tile drainage, herbicides, fecal coliform and phosphorus are primarily delivered via surface runoff. Whereas nitrate concentration patterns vary according to a near normal distribution in a year, annual concentration patterns of runoff driven pollutants are highly skewed and often vary several orders of magnitude between stormflow and baseflow. Because of this variability, sampling strategy becomes critical for detecting water quality changes in surface water. Water sampling on a fixed interval basis, as used during this project, did not routinely monitor storm

Table 38. Mean score (0 to 10 possible) and 95% Confidence Interval (CI) for the Index of Biotic Integrity (FIBI) and 11 metrics scores for 10 reference sites (31 sampling events) and 12 randomly selected REMAP sites (12 sampling events) in Ecoregion 47F lacking stable riffles and abundant

	Reference Sites		REMAP Sites	
Metric	**Mean Score**	**95% CI**	**Mean Score**	**95% CI**
Native Fish Species Richness	6.5	5.9 - 7.1	5.7	4.8 - 6.7
Number of Sucker Species	4.4	3.5 - 5.4	2.6	0.9 - 4.3
Number of Sensitive Species	1.4	1.0 - 1.9	1.0	0.3 - 1.7
Number of Benthic Invertivore Species	4.6	3.9 - 5.3	3.1	2.1 - 4.1
Proportion of Three Dominant Species (%)	6.1	5.3 - 6.9	4.7	3.3 - 6.1
Proportion of Fish as Benthic Invertivores (%)	3.1	2.2 - 4.1	2.2	0.5 - 3.8
Proportion of Fish as Omnivores (%)	6.1	5.1 - 7.2	6.8	5.2 - 8.4
Proportion of Fish as Top Carnivores (%)	2.8	1.8 - 3.9	1.8	0.5 - 3.0
Proportion of Fish as Simple Lithophilus Spawners (%)	2.1	1.4 - 2.8	0.9	0.3 - 1.4
Fish Assemblage Tolerance Index	3.7	3.0 - 4.4	3.9	2.3 - 5.6
Adjusted CPUE	3.1	2.2 - 4.1	2.7	1.4 - 3.9
Overall FIBI score	39.9	35.4 - 44.5	32.1	25.4 - 38.8

CPUE = catch per unit effort

events when the majority of runoff-pollutants are delivered to streams. Since baseflow constitutes a majority of total streamflow, a fixed sampling interval will be biased toward collecting baseflow water samples. During the few sampling days that corresponded to runoff events, high concentrations of herbicides and phosphorus were detected. Several high flow events sampled during this study occurred during the last years of the project that, in all likelihood, skewed the time-series trend analyses and load estimation.

This was particularly reflected in atrazine concentration patterns. Despite atrazine restrictions on refuge-owned lands that reduced atrazine applications by an estimated 28 percent, atrazine continued to be detected in Walnut Creek and subbasins at frequencies and concentrations no different than Squaw Creek. And yet, a baseflow synoptic sampling completed in 1999 revealed large differences in atrazine losses within Walnut Creek refuge that were not observable by the fixed interval monitoring. It seems unlikely that the large reduction in atrazine applications that occurred in Walnut Creek watershed, compared to a probably increase in applications in Squaw Creek associated with increased row crop, would not have resulted in a statistically significant decrease in herbicide concentrations in Walnut Creek. Thus, changes in surface water concentrations of atrazine and other herbicides may have occurred but they could have been missed by the sampling design.

The lack of significant changes in fecal coliform concentrations at the watershed outlets WNT2 and SQW2 may point to poor sampling design or to an insufficient amount of change

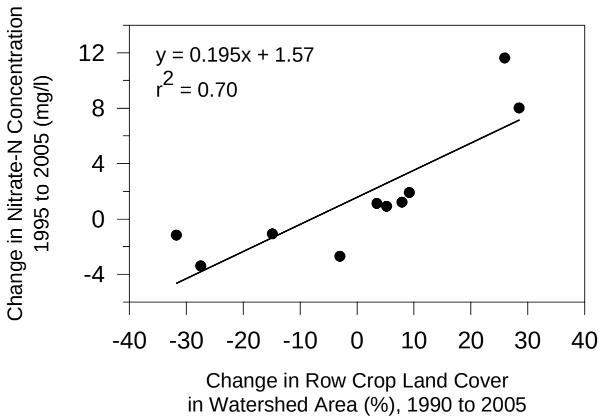


Figure 71. Relation of change in nitrate in stream nitrate concentrations (as determined by statistical methods) with change in percentage of land cover in row crops in watersheds and subbasins.

occurring in the watersheds. Elevated fecal coliform detections occurred throughout the monitoring project, with elevated fecal coliform detections associated with both rainfall runoff and late summer low flow periods. High fecal coliform counts during high flow may be due to both contributions from distal sources and local sources, but low flow peaks in fecal coliform imply that sources are close to the monitoring sites. The best monitoring strategy would probably be a combination of an event-based sampling design with fixed interval sampling to detect watershed-scale fecal coliform trends.

However, it is quite possible the monitoring program implemented during the project was sufficient to detect changes in fecal coliform if they in fact occurred. Statistically significant changes were observed in smaller subbasins where changes in pasture intensity were evident. Increases in fecal coliform in Walnut Creek subbasins may be due to bison grazing whereas a decreasing trend in upstream Walnut Creek and one Squaw Creek subbasin may be due to decreasing grazing activity. Overall, at a watershed scale, minimal changes occurred in grazing intensity or manure applications in either Walnut or Squaw creek watersheds to expect measurement of statistically significant changes at their watershed outlets. Moreover, in the case of WNT2, a large pasture with cattle access to Walnut Creek is located within a half-mile of the downstream gaging station that probably limited the detectability of changes in fecal coliform concentrations in the Walnut Creek watershed.

Other studies have had success in documenting the effectiveness of BMPs to reduce sediment, nutrients and bacteria runoff from unrestricted livestock grazing (Meals, 2001; Line and Jennings, 2002; McNeil, et al., 2003). However, detecting changes in these NPS pollutants has typically required a sampling effort specifically designed to detect the change, including event-based sampling and before/after treatment comparisons. Thus, should future targeting of BMPs for pasture sites in Walnut and Squaw creek watersheds be considered, monitoring activities should be focused on isolating the change to a smaller area with an emphasis placed on designing an appropriate monitoring strategy.

In terms of phosphorus, data from the Walnut Creek project provides much needed information on concentration ranges and temporal patterns of P in small agricultural watersheds. However, like fecal coliform, a sampling strategy designed to capture P variability during both runoff events and baseflow would probably be needed to detect changes in P over time. Furthermore, additional analysis of total P and dissolved P (orthophosphate) in stream water would aid interpretation of P patterns and sources.

Detecting Changes in Biological Indices

Despite changes in terrestrial land cover during the project, changes in biological monitoring indices over time were not evident. In regards to benthic macroinvertebrates, it was likely that local hydrologic and habitat factors largely governed the macroinvertebrate community structure. Many plains streams are frequently dominated by collectors and have low taxonomic diversity (Harris et al., 1999). However, Walnut Creek has shown the ability to support many taxa of aquatic macroinvertebrates, with a total of 108 mutually exclusive taxa collected over the course of the monitoring project. This suggests that local (e.g. macro/microhabitat) factors tended to influence community structure more than landscape changes occurring in Walnut Creek watershed. Research has noted that changes in local habitat quality may influence community structure change more than water quality changes (Minshall 1978, Richards et al. 1993), although changes in stream hydrology coupled with instream productivity changes may influence trophic structure (e.g. influencing shifts from collectors/gatherers to scrapers). Walnut Creek has shown low levels of primary productivity in comparison with regional streams (Don Huggins-Kansas Biological Survey, personal communication). Thus, the possibility exists that as water quality/clarity improves in Walnut Creek over time, the stream may shift more towards an autochthonous versus an allochthonous stream with a concomitant change in community structure.

Variability in the fish communities and FIBI scores for both Walnut and Squaw creek sites may be attributed to several factors, such as fluctuating flow regimes or differences in watersheds. Walnut Creek discharges into Lake Red Rock, an impoundment of the Des Moines River, whereas Squaw Creek discharges into the South Skunk River. Because the fish community structure of the Lake Red Rock reservoir is likely different than community structure of the South Skunk River, this might offer a possible explanation as to why more and different species were found in Walnut Creek. Although no improvement in FIBI scores was observed over time or between creeks, this fish community data provides baseline data for future comparisons, when improvements in the fish communities might become more pronounced.

Detecting Changes in Suspended Sediment

Sediment Erosion Model

The Revised Universal Soil Loss Equation (RUSLE) was used to estimate the changes in gross sediment erosion occurring in Walnut and Squaw Creek watersheds from 1990 to 2005. RUSLE model is a method for calculating the average sheet and rill erosion under specified conditions and was developed by the U.S. Department of Agriculture. GIS software (Arcview spatial analyst) was used to calculate the erosion rate for a 30-meter grid cell using the 1990 and 2005 landcover grids, the ISPAID soil grid and the NRCS Rainfall Erosion Index. Results from the RUSLE model represent total annual gross erosion from the watershed and does not account for actual sediment delivery to a stream. However, the model was improved in the case of Walnut and Squaw creek watersheds by incorporating the effects of conservation practices mapped by CLU into the K factor and slope length determinations (erodibility factors). Except for prairie plantings, all conservation practices mapped in 2005 were assumed to be present at the same locations in 1990. With inclusion of conservation practices, results from the RUSLE model are presumed to represent a close approximation of the total gross sheet and rill erosion from the landscape.

Sediment erosion modeling suggests that land use changes in upland areas of Walnut and Squaw creek watersheds have affected sediment erosion potential. In Walnut Creek watershed, based on the 1990 land cover, the RUSLE model predicted that 35,871 tons of sediment were eroded from the landscape in 1990. In 2005, following conversion of 23.5 percent of the landscape to native prairie, total gross erosion in the watershed was reduced to 22,591 tons. Thus, prairie restoration reduced gross sediment erosion by 13,279 tons, or 37 percent between 1990 and 2005. A 15 percent decrease in row crop acreage from 1990 to

2005 translated to a 37 percent decrease in sediment erosion.

In Squaw Creek watershed, the RUSLE model predicted that 25,078 tons of sediment were eroded in 1990 and 28,857 tons were eroded in 2005, an increase of 15 percent (3,779 tons). Thus, a 9 percent increase in row crop acreage in Squaw Creek translated to a 15 percent increase in total gross erosion. This is noteworthy since much of the land that was converted to row crop from CRP in Squaw Creek was located in highly erodible areas. Results from the RUSLE modeling suggest that sediment erosion in Walnut Creek was reduced by 46 percent compared to Squaw Creek. This would probably be considered a success in other watershed monitoring projects without collection of actual sediment monitoring data.

Sediment Sources

However, suspended sediment concentrations and loads measured at the two watershed outlets did not show significant differences from 1995 to 2005 despite the land use changes. Daily and seasonal patterns of sediment discharge were similar in both Walnut and Squaw creek watersheds, and both watersheds showed a strong linear relation of annual discharge to sediment loss. Indeed, the regression annual discharge versus sediment loss was nearly identical for both watersheds (Figure 26). It is hypothesized that sediment sources are different in Walnut compared to Squaw, with streambank erosion playing a significantly greater role in sediment delivery in Walnut Creek watershed. Research has shown that the source of sediment in incised channels in Mississippi and Tennessee was dominated by streambeds and banks (Shields et al., 1995; Simon, 1989). Two mapping projects implemented during the project provide evidence to suggest that streambank erosion is also an important process in Walnut Creek watershed.

In October 1998, channel features and morphology of Walnut Creek were described in detail by traversing the stream channel located between the two USGS stream gauges (Schilling and Wolter, 2000). While streambank recession rates varied considerably in different channel reaches, little evidence for bank erosion was observed in straightened segments of the channel but severe bank erosion occurred at many outside meander bends, debris dams or cattle access points. Severe bank erosion on many outside meander bends tended to be located immediately downstream of straightened reaches. Average left and right bank erosion rates were similar (0.135 ft/yr and 0.143 ft/yr, respectively) and yielded a watershed erosion rate of 0.278 ft/yr. Based on the stream survey data, the mass of sediment eroded from Walnut Creek streambanks was estimated to be 7,091 tons per year for the portion of channel located between the two USGS gauging stations. Considering that the average annual sediment load at the downstream Walnut Creek USGS gaging station for the 1996 to 1998 period (time corresponding to the stream survey) ranged 9,399 to 18,367 tons, the percentage of total annual suspended sediment load derived from streambank erosion may range from 39 to 75 percent. It should be noted that discharge and sediment export for water years 1996 and 1998 were the highest measured during the project so that while the relative percentage of suspended sediment load derived from streambank erosion may be consistent across years, the tons of sediment derived from bank erosion may vary considerably according to precipitation and discharge patterns. If the 1998 bank erosion estimate of 7,091 tons is applied to the 10-year average sediment export from Walnut Creek (8,384 tons), the percentage of total sediment load from streambank erosion is estimated to be approximately 85 percent.

In 2004, a second steam mapping program was conducted in the main channels of Walnut and Squaw creeks in partnership with researchers from Iowa State University. Similar to methodology used in October 1998, the stream channels were traversed and GPS was used to identify and map severely eroding streambanks in both watersheds. Results from this investigation are pending further analyses, but preliminary findings indicate that severe streambank erosion is more prevalent in Walnut Creek compared to Squaw Creek. In Walnut Creek watershed, approximately 19,200 feet of severely eroding streambanks were mapped between the two gages (total stream length of 33,473 feet), or 57.4 percent of the stream channel consisted of severely eroding banks. In Squaw Creek, the same relative main channel stream length between SQW1 and SQW2 (28,320 feet) had 4,784 feet of severely eroding banks, or 16.9 percent of the total stream length. Thus, Walnut Creek has over three times more severe streambank erosion occurring than Squaw Creek.

In both watersheds, streambanks are particularly susceptible to erosion. Channel incision has occurred primarily through post-settlement materials and Holocene alluvium, which lack the cohesive strength of the underlying pre-Illinoian till. In particular, Historical post-settlement alluvium (Camp Creek Member) would be easily remobilized by streambank erosion because these materials lack internal structure provided by buried soil horizons developed during the Holocene (Bettis and Littke, 1987; Kreznor et al., 1990; Beach, 1994).

A temporal comparison between 1998 and 2004 in Walnut Creek suggests less bank erosion occurring in 2004 than 1998. Total sediment export from bank erosion in 2004 was estimated to be 4,265 tons based on the length of eroding channel segment mapped in 2004 and an average erosion rate of 0.28 feet/year (same value as used in 1998). This total represents approximately 51 percent of the average annual sediment export. Considering the low-flow conditions measured during several post-2000 years, vegetation may have re-armored some streambanks that were observed to be severely eroding in 1998. The percentage of annual sediment load from streambank erosion in Walnut Creek is consistent with data from larger Iowa rivers where it has been estimated that 45% of the total sediment load leaving the state was from in-stream bank erosion. (Odgaard, 1984).

A similar calculation of streambank sediment export for Squaw Creek can be made. Assuming an average erosion rate of 0.28 feet/year (same value as Walnut) and a bank height of 10 feet, approximately 1,127 tons of sediment is estimated to be derived from bank erosion in Squaw Creek. With an average sediment export of 8,044 tons, it is estimated that bank erosion contributes approximately 14 percent of the annual sediment load in Squaw Creek. This is substantially less than the contribution of streambank erosion in Walnut Creek.

Thus, the 2004 stream mapping data suggest that sources of sediment to Walnut and Squaw creeks are different. Streambank erosion appears to play a much more dominant role in sediment erosion in Walnut Creek compared to Squaw Creek. However, with annual sediment export nearly the same from the two watersheds, by difference, the source of sediment in Squaw Creek must be something other than bank erosion. It is hypothesized that overland sheet and rill erosion is primarily responsible for sediment export in Squaw Creek. This is consistent with row crops comprising a much greater proportion of land cover in Squaw Creek watershed than Walnut Creek. Field reconnaissance and land cover mapping of the riparian zone of Squaw Creek further indicates that row crops are often planted to the channel edge with little or no riparian buffers present along the stream corridor. Thus soil erosion from overland flow may be quickly routed to surface water. In contrast, much of the riparian corridor in Walnut Creek is buffered with warm and cool season grass or forest. Although modeling suggested that sheet and rill erosion was reduced by more than 20 percent in Walnut Creek, the fact that streamflow did not change during the monitoring project indicates that the power of the stream to erode sediment was not diminished. Thus, hungry water (that is, water containing less sediment than its carrying capacity) discharged to Walnut Creek may have eroded sediment from other sources, including the streambanks and streambed.

Differences in stream morphology may have also contributed to differences in sediment sources between Walnut and Squaw creeks. Although Squaw Creek is only marginally less sinuous than Walnut Creek (1.14 compared to 1.16, respectively), there is little evidence of natural stream meanders remaining in Squaw Creek watershed. In contrast, several reaches of Walnut Creek are heavily meandered and these segments are integrated throughout the main channel length. Stream mapping documented that streambank erosion in the meandered segments of Walnut Creek was much greater than channelized segments and often associated with debris dams and cattle access points (see discussion below). Squaw Creek watershed contains little riparian forest to contribute to debris dams and few pastures. Squaw Creek also has greater basin relief, relative relief, and steeper main channel slope than Walnut Creek. Thus, streamflow in Squaw Creek may be more rapidly directed through the straightened stream network with little disruption of channel flow. Indeed, discharge and sediment data indicates flashier conditions in Squaw Creek than Walnut Creek.

In contrast to bank erosion, downcutting through the streambed appears to be a less significant source of sediment than suggested by other studies. In Walnut Creek, the longitudinal profile of the streambed measured in 2005 exhibited a fairly consistent slope, averaging 7.6 feet per mile (Figure 72). However, differences in channel slope at downstream and upstream locations suggest that the stream channel may not have reached equilibrium. The channel gradient was lower in the downstream portion of the watershed and steeper in the upstream area (Figure 72). This suggests that the downstream channel is aggrading whereas the upstream channel is still eroding. A sharp change in channel gradient measured at approximately mile 3 above WNT2 may be indicative of a knickpoint in the channel, but considering that the channel slope did not vary

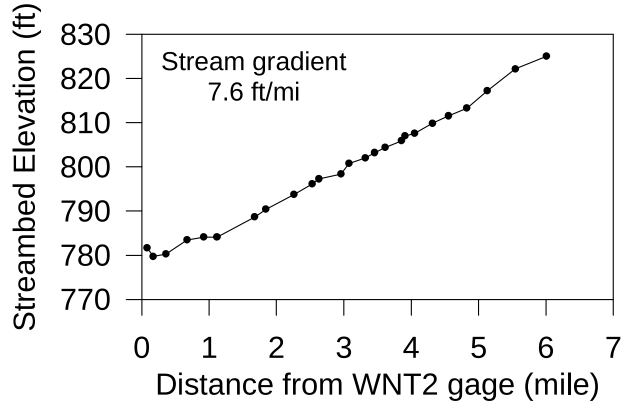


Figure 72. Longitudinal profile of streambed elevation from stream gage sites WNT2 to WNT1 in Walnut Creek watershed.

on either side of this area, it is suspected that this point is a measurement error. The area between the two measurement points is a bridge crossing that may have affected the channel gradient.

The channel longitudinal profile data is consistent with field mapping data from 1998 that showed evidence for higher stream gradients in several straightened reaches of Walnut Creek. Channelized segments in the upper reaches of the watershed exhibited a narrow, V-shaped channel cross section and a channel bottom consisting of bare or thinly mantled substrate (Schilling and Wolter, 2000). However, stream mapping in Walnut Creek showed little or no streambank erosion occurring in these straightened reaches. On the contrary, downcutting appears to have been slowed by a resistant substrate bottom (pre-Illinoian till or Roberts Creek Member alluvium) that allowed streambanks to become relatively stable and well vegetated. These conditions typified much of the Squaw Creek channel as well. Streambanks in channelized segments showed little sign of recent mass wasting or undercutting, whereas most severe streambank erosion in Walnut Creek was concentrated in active meandering areas, debris dam areas and cattle access points. Substantial accumulation of streambed sediment was measured in downstream Walnut Creek channel.

The mass of sediment stored on the channel bottom and the amount of time needed to remove it from the Walnut Creek watershed was estimated (Schilling and Wolter, 2000). Based on the stream survey data, the average channel bottom width of Walnut Creek was 13.24 feet and the average bed thickness was 0.41 feet. Considering that the channel length between the two gaging stations was 32,203 feet, the volume of sediment stored in the channel was estimated to be 174,810 cubic feet. Assuming a specific gravity of 2.70 for the silty streambed material (Spangler and Handy, 1982), the mass of stored sediment in the channel was approximately 14,726 tons. Based on the mean annual flow rate and annual suspended sediment concentration measured at the downstream USGS station for the 10-year monitoring period between 1996 and 2005 (14.8 cfs and 104.1 mg/l, respectively), flushing the amount of sediment stored in the channel bottom from the watershed would require approximately 9.7 years, assuming no additional inputs. Hence, eroded sediment that reaches the main channels of Walnut or Squaw creeks does not necessarily exit the watersheds immediately. Sediment that accumulates in the channel behind logs, debris dams or other impediments can provide temporary base levels in the stream and temporary storage sites of sediment for long periods (Mosley, 1981; Trimble 1983). When these temporary storage sites are destroyed or disrupted, stored sediment becomes available again for transport and eventually flushed from the watershed. Thus, streambed sediment stored in the channel bottom can also provide a continuing source of sediment downstream.

Lag Time for Detecting Changes in Sediment Export

Several factors conspire to make it difficult to assess the timeframe needed to observe reductions in suspended sediment loads in Walnut Creek watershed. First, the variability and flashy behavior of discharge and sediment transport in the watershed is problematic for detecting changes in sediment export. Although samples were collected on a daily basis, it is unknown whether this sampling resolution adequately captured variations in sediment concentrations and loads that occurred on an interval less than this (i.e., hourly). While discharge and control covariates are beneficial for detecting change in the treatment watershed (Walnut Creek), these factors only explain a portion of the variance. For example, in Ohio, a six percent reduction in sediment loads over a 10-year period was not detectable despite collection of daily discharge and sediment data and having a control watershed (Richards and Grabow, 2003). Richards and Grabow (2003) recommended that additional sampling be implemented to reduce autocorrelation of daily samples and improve the detectability of change.

Secondly, often a long term monitoring record is needed to factor out influences of climate and historical sediment storage. Potter (1991) looked at a 47-year period of record in southwestern Wisconsin to factor out climatic factors in order to detect changes in watershed hydrology resulting from adoption of conservation measures. Others, such as Trimble (1983), Knox (1977) and Beach (1994), examined historical records dating as far back as the 1830’s to prepare sediment budgets needed to estimate changes in sediment transport at watershed scales. Trimble and Lund (1982) looked at more than 100 years of land use and sedimentation records and noted a lag time (or “hysteresis”) of 10 years between 1930 and 1940 in the Coon Creek watershed in Wisconsin before improvements in land use resulted in decreases in erosion and sedimentation. In the Buffalo River watershed in west-central Wisconsin, Faulkner and McIntyre (1996) investigated the persistence of high sediment yields despite decades of erosion control and land use changes. They found that channel incision migrating into tributary streams increased conveyance capacities of sediment downstream.

In Iowa, the longest period of record available for a similar size watershed is the Sny Magill/Bloody Run paired watershed study in northeast Iowa (Fields et al., 2005). The stream morphology and substrate materials of the Sny Magill/Bloody Run paired watershed (Seigley, et al., 1994, 1996) are substantially different than the Walnut/Squaw paired watersheds. The Sny Magill/Bloody Run watersheds are located in the Paleozoic Plateau landform region where the landscape is characterized by thin loess and till overlying bedrock formations (Prior, 1991; Seigley et al., 1994). However, both Sny Magill/ Bloody Run and Walnut/Squaw Creek watershed pairs move the majority of sediment during intermittent short-term events. In 1994, for example, seven-days (not consecutive) accounted for 68 to 85% of the total annual sediment load in the Sny Magill watershed (Seigley et al., 1996). Despite 70-80% participation from landowners and the installation of nearly 270,000 feet of terraces and other sediment control structures, changes in discharge and suspended sediment were not detected due to the flashy behavior of the streams (Fields et al., 2005).

Thirdly, the fact that sediment may be delivered to Walnut and Squaw creeks differently (streambanks vs. overland flow) adds a further complication to pairing watersheds to detect change. Modeling suggested a 46 percent reduction in sediment erosion in Walnut Creek compared to Squaw Creek but no changes were measured in sediment yields. Indeed, sediment yields were remarkably similar between the two watersheds despite very different sediment sources. Thus, assessing the lag time for measuring watershed improvements in watershed monitoring projects is complicated by varying sediment sources contributing to annual sediment yields. Comparing sediment export across watershed regions may not be measuring, in essence, “apples to apples.”

The implication suggested by this project and other studies is that there may a significant time lag before changes in land use in the Walnut Creek watershed translate to reduced discharge and suspended sediment loads downstream. Several factors contribute to the difficulty of detecting short-term change in sediment loads in the watershed. Climate variability is a primary reason other watershed monitoring projects require decades to observe sediment reductions in streams. Climatic effects, including variable location and intensity of precipitation, can completely overwhelm and mask any reductions in discharge and sediment loads for many years. Short-term monitoring projects (for example less than 10 years) have little to no chance of detecting reductions in sediment loads to streams in a humid climate. Additionally, historical land use in the Walnut and Squaw Creek watersheds has resulted in variable amounts and distribution of sediment stored on the floodplain. Considering that streambank erosion and gullies can contribute significantly to annual sediment loads, land use changes in the uplands may be difficult to detect above contributions from these sediment sources. Most of the sediment available for sediment migration in channels and gullies is derived from historical sediment storage. Finally, while conversion of additional row crop lands in Walnut Creek watershed to prairie will continue to reduce the contribution of sediment from sheet and rill erosion to the channel, channel discharge will continue to erode streambank materials unless it is slowed or reduced. Reducing discharge in Walnut Creek is likely the only way sediment export from the watershed will be reduced over time. Prairie restoration will assist with this, but contributions from tile-drained headwater areas and runoff from the steeply sloping pre-Illinoian glacial landscape will continue to deliver water to the incised channel at rates that will continue to erode streambanks severely. Once water is delivered to the channel, the incised banks are particularly vulnerable to collapse with a rapidly rising and falling stream stage and poor soil structure in the exposed banks.

LESSONS LEARNED

The following are some of the lessons learned from the Walnut Creek monitoring project:

- Prairie restoration can be an effective BMP to reduce nitrate concentrations and loads in agricultural watersheds. A nitrate reduction of 0.7 to 3.4 mg/l/10 years was measured in Walnut Creek watershed.
- As demonstrated by other studies, row crop land cover is significantly related to stream nitrate concentrations. Converting row crop to native prairie at the Neal Smith NWR reduced the amount of row crop in the various watershed areas and reduced stream nitrate, whereas converting CRP grass back to row crop in Squaw Creek increased the amount of row crop and greatly increased stream nitrate.
- The rate of nitrate concentration reduction measured in streams will be dependent upon the rate of groundwater flow to transport nitrate water to streams. In the Walnut Creek watershed, slow groundwater flow velocities suggest that nitrate reductions from upland prairie restoration plots will take many decades to be measured in streams. Land use changes occurring in the floodplain are more likely to have an impact on short term water quality than those associated with upland settings. Tile drainage accelerates the movement of subsurface water through soils and can possibly accelerate detection of concentration changes through time.
- Headwater regions of watersheds exert a proportionally large effect on watershed NPS export. In Walnut Creek watershed, statistical analyses and synoptic surveys indicate that much of the downstream concentrations of NPS pollutants in Walnut and Squaw Creek watersheds can be explained by upstream contributions. Once the pollutant is discharged into the incised stream network from row crop dominated headwater regions, concentrations remain elevated in the stream. Prairie restoration placed in the core of a watershed served to dilute concentrations from upstream sources.
- It was easier to detect changes occurring in NPS pollutants over time in smaller subbasins than the larger project watersheds. When areas of land use change were isolated at the subbasin scale, substantially greater water quality changes were observed.
- An event-based sampling protocol rather than a set sampling schedule would have been more appropriate to detect changes in herbicides, fecal coliform and phosphorus concentrations over time. A set sampling schedule was useful to characterize concentration ranges and long-term variability, but was not effective in capturing changes in NPS pollutants delivered primarily with runoff.
- Biological monitoring of benthic macroinvertebrates and fish was not sufficiently sensitive to detect any changes in water quality occurring in Walnut Creek watershed. Difficulties included obtaining sufficient colonization in the flashy incised streams, and accounting for the effects of downstream fish populations on measured populations. Biological monitoring may be more appropriate to assess water quality patterns across spatial scales rather than temporal scales less than 10 years.
- Suspended sediment concentrations and loads are difficult to characterize in incised streams that transport most of their sediment loads during infrequent high flows. Event-based monitoring is needed to supplement fixed monitoring to fully characterize sediment transport in these incised streams.
- Characterizing sediment reductions in watershed projects using sediment erosion models does not accurately reflect reduced sediment export. Sediment sources vary in watersheds and streambank erosion can contribute significantly to watershed sediment loads.
- Reducing upland sheet and rill erosion in watersheds without reducing water discharge from these areas will likely shift sediment sources from upland sources to instream sources such as streambanks and streambed.

• A lag time of decades is likely needed to measure changes in sediment export in order to overcome variable climate and historical sediment storage. Paired watershed studies assist in detecting change but consideration must be given to account for differences in sediment sources and delivery to streams.

• Long term monitoring is needed to capture changes in water quality due to implementation (or abandonment) of conservation practices. If benefits of conservation practices on water quality are to be fully assessed, a combination of intensive monitoring and modeling is recommended.

CONCLUSIONS

The Walnut Creek Monitoring Project began with an ambitious goal to implement a water quality program to document water quality improvements resulting from large-scale watershed restoration and management. Project results indicate that prairie restoration in an agricultural watershed can improve water quality with regards to nitrate concentrations and loads. Replacement of 17 percent of row crop lands in Walnut Creek watershed with native prairie (23 percent of the watershed planted in prairie) resulted in a reduction of nitrate of approximately 1.1 mg/l over 10 years. While this reduction cannot be considered overly substantial, it does shed light on the difficulty of detecting nitrate concentration changes in an agricultural setting where natural and anthropogenic nitrogen sources are ubiquitous. Upstream contributions from tile-drained, upland row crop areas had a significant effect on downstream water quality such that prairie restoration occurring in the core of the watershed primarily had the effect of diluting upstream nitrate contributions.

Nonetheless, native prairie restoration should be viewed as a viable conservation strategy for improving water quality in streams. As demonstrated by this study, and also by other plot studies and modeling efforts, reintroduction of perennial grasses in the agricultural landscape can serve to reduce water flux and nitrate delivery to streams. Project results highlighted the close relation of stream nitrate concentrations to land use change from row crops to grasslands. In Walnut Creek, converting row crop to grass reduced nitrate concentrations over time, but in Squaw Creek, stream nitrate concentrations rapidly increased when grasslands were converted back to row crop. This situation is analogous to historical conditions in Iowa that demonstrated how baseflow and stream nitrate concentrations increased in the 20th century as row crop acreage increased. Thus, it must be emphasized that grasslands placed in agricultural settings for water quality benefits should be part of a long-term solution to water quality problems if the water quality benefits are to be fully realized.

Early in the project, Schilling and Thompson (1999) wondered if the size of the Walnut Creek watershed was too large to detect water quality changes. Results suggest that water quality changes were greater and much easier to detect in small subbasins compared to the watershed outlets. Considering that Walnut Creek is a rather small 12-digit HUC in Iowa, project results should be kept in mind when expectations are raised for detecting water quality improvements from changing land use in larger watersheds. However, since all subbasins comprise part of larger and larger watershed areas, perhaps documenting improvements in stream water quality from conservation practices should be focused on small subbasins where changes can be detectable in shorter time frames. Detecting water quality improvements in larger watersheds will likely require a dedicated long-term monitoring effort on the order of several decades.

Project conclusions derived from herbicides, fecal coliform, phosphorus and biological sampling data were less straightforward. In the case of herbicides, fecal coliform and phosphorus, water quality changes may have been easier to detect if samples were collected on an event-basis rather than on a fixed interval schedule. Given the magnitude of reduction in herbicide applications in Walnut Creek watershed, it is likely that changes in stream herbicide concentrations occurred but were missed by sampling design. Statistically significant changes in fecal coliform concentrations over time were evident in some subbasins that may be related to land use change, but overall, lack of widespread changes in fecal coliform source areas in the watersheds prevented detection of changes at the watershed scale. Phosphorus monitoring provided insight on typical concentrations and temporal patterns of phosphorus transport in streams but was not monitored for the full duration of the project. Future monitoring for this compound should consider analyzing for the various chemical forms of phosphorus to develop an understanding of P sources and delivery processes. Ten years of biological monitoring also served to increase our knowledge of populations, abundances and variability of benthic organisms and fish in degraded Southern Iowa streams. However, annual biological monitoring was not sensitive enough to detect any subtle changes in water quality that may have occurred in Walnut Creek from prairie restoration.

Deciphering patterns of transport and sources of suspended sediment during the Walnut Creek project added greatly to the understanding of sediment erosion processes that occur in incised southern Iowa watersheds. Project results indicated that discharge and suspended sediment transport was very flashy and that much of the sediment export occurred during occasional events primarily in May and June of each year. Sediment sources can vary markedly between watersheds, as shown by this study, with streambank erosion dominating sediment sources in Walnut Creek and sheet and rill erosion primarily responsible for sediment loads in Squaw Creek. Sediment erosion modeling suggested that sheet and rill erosion in Walnut Creek was reduced greatly due to prairie restoration compared to 1990 levels and conditions in Squaw Creek. However, sediment export in both watersheds was nearly identical and related significantly with discharge. Thus, project results suggest that for other watershed projects, the first step needed to reduce sediment loads should be directed at reducing streamflow and stream power.

Finally, results from the Walnut Creek Monitoring Project attest to the necessity of conducting long-term monitoring to evaluate the effects of land use change and conservation practices on water quality. Lag times for observing water quality improvements are rarely less than several years long, and in the case of sediment, lag times of decades are the norm rather than the exception. In Walnut Creek watershed, it took more than three years of monitoring before the first statistically significant changes in nitrate concentrations were detected. After 10 years, some water quality changes became more pronounced whereas other changes highlighted in earlier reports were less significant now due to climatic effects and increased sample variance. Long term monitoring is needed to factor out the effects of climate and account for possible improvements in water quality that may take many years to show up in a stream. With water quality monitoring finished in Walnut Creek watershed, land restoration occurring throughout the watershed will continue to improve stream water quality over time but these improvements will go unmeasured and unreported. Hence, the opportunity to fill the data gap in establishing the long-term water quality benefits from grassland introduction into an agricultural watershed may be lost.

Similarly, other watershed conservation projects funded to improve water quality in streams often claim success but lack accountability for measuring actual water quality benefits. Without monitoring, potential water quality improvements from watershed projects will go unnoticed and unappreciated as the public debates the costs and benefits of programs designed to reduce NPS pollution in streams. Moreover, as the Walnut Creek project results demonstrated, monitoring will also establish realistic expectations for success to educate the public that solutions to NPS pollution problems are not easy and are not quick.

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Iowa Department of Natural Resources
Geological Survey
109 Trowbridge Hall
Iowa City, Iowa 52242-1319
(319) 335-1575
www.igsb.uiowa.edu