NUTRIENT DYNAMICS IN A SMALL AGRICULTURAL LAKE ERIE TRIBUTARY

by

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ABSTRACT

This study examined how land use, water level fluctuations of Lake Erie, and discharge all affect seasonal nutrient concentrations and delivery on Crane Creek, a small agricultural tributary of Lake Erie in Northwest Ohio. Seventeen sites were sampled in the Crane Creek watershed from May to November 2004 and April to June 2005. These sites were chosen to capture the variability of land use in the watershed and included potential point sources, catchments with a variety of land uses, sites within the Ottawa National Wildlife Refuge, and a near-shore Lake Erie site. Hydrologic measurements along with water samples were taken at each site and evaluated for nitrite-nitrate nitrogen, SRP, ammonia nitrogen, and several other water quality parameters.

There were three major findings. First, both water level fluctuations driven by Lake Erie seiches and higher discharge make the downstream sites less spatially and temporally variable than the upstream sites. The downstream sites also had higher water quality because of wetland transformation of nutrients and dilution from lake water inflow. Second, while agricultural and urban land use likely contribute nutrients from fertilizer use and urban runoff, point sources in the catchment seem to have a greater influence on water quality in Crane Creek, particularly in times of low stream discharge. The influence of varying patterns of land use was difficult to determine because homogeneity of the landscape and point sources confounded the analysis. Finally, within the lower estuary, water quality in the system was similar to water quality in surrounding diked pools. This similarity makes hydrologic reconnection of these wetlands a possibility, although physical constraints complicate the restoration process. These findings have implications for other small Lake Erie tributaries.

I. Introduction

Over the past century, increasing nutrient inputs to the Great Lakes have caused declines in water quality and adverse effects on species and food webs. Increasing nutrient inputs to Lake Erie has been of particular concern because its surrounding land area is highly agricultural and urban, and its small volume makes it particularly vulnerable to nutrient loading effects. Legislation resulting in improved wastewater treatment has helped to reduce point source nutrient inputs to Lake Erie, particularly phosphorus loading (Fraser 1987, Rosa 1987, Richards and Baker 1993). However, anthropogenic sources, including nonpoint sources such as agricultural and urban runoff, remain problematic (Richards and Baker 2002). As a result, there has been considerable research on the tributaries of Lake Erie to determine how anthropogenic sources control both loading to the Lake and nutrient concentrations in these freshwater ecosystems.

A key area of study has been the coastal wetlands at the mouths of the tributaries, particularly on the western side of the basin. The coastal wetlands are unique in that they are affected not only by short-term storm events and the discharge of the tributaries, but also by seasonal and long-term changes in lake levels (Herdendorf 1992, Keough 1999). These wetlands are important because they not only have economic and recreational value, but also serve key ecological functions and provide important reproductive habitat for Lake Erie biota (Herdendorf 1992, Prince et al. 1992). Several studies have shown that coastal wetlands reduce nutrient inputs to Lake Erie by acting as nutrient sinks or transformers (Heath 1992, Mitch 1994, Krieger 2003), though this latter process is highly variable with discharge. Due to coastal development, agricultural practices, diking of wetland units, and loss of protective barrier beaches, most of the original coastal wetlands in western Lake Erie have been destroyed and few estuarine wetland complexes remain (Herdendorf 1987, Kowalski and Wilcox 1999).

Other studies have focused on the effects of land use on the tributary watersheds. Here, a healthy stream is defined as one that meets the recreational, economic, and social needs of society and still maintains its ecological integrity (Meyer 1997). Many agricultural and urban land uses seen in the Lake Erie watershed have detrimental effects on water quality and watershed health (Smith et al. 1987, Allan 2004). Agricultural and urban lands contribute higher levels of phosphorus and nitrogen than other land uses (Tong and Chen 2002), and can also affect stream hydrology and sedimentation (Allan 2004).

While some studies indicate that urbanization is more important than agriculture in determining nutrient levels and water temperature (Osborne and Wiley 1988, LeBlanc 1997), agricultural land use plays a significant role in determining the water quality of many watersheds. The Ohio EPA integrated report of 2004, for example, named agricultural practices as a high magnitude source of degradation to Lake Erie tributaries. Although the effect of agricultural land use on water temperature is uncertain (Borman and Larson 2003), nitrate and phosphorus levels have been linked with fertilizer application and runoff (Castillo et al. 2000, Baker and Richards 2002, Boyer et al 2002). This relationship is subject to variation, since specific agricultural practices and crop rotation patterns can have varying effects (Meissner et al. 1999, Forster et al. 2000), as can application of specific herbicides (Richards et al. 1996).

Two studies by Richards and Baker highlight how nutrient concentrations in Lake Erie tributaries have changed over time. The first study was reported in 1993 and included data from several Lake Erie tributaries spanning from 1975-1990. They found that, over this period of time, phosphorus had decreased in the tributaries, most likely because of point source reduction. Nitrate, on the other hand, had increased, most likely due to agricultural land use. Richards and Baker's second study in 2002 examined many of the same tributaries over the same time period, but more directly examined the connections of water quality trends and agriculture. In this study, they determined that fertilizer and manure application rates were highly predictive of phosphorus and nitrogen. Although these two studies provide valuable information about general trends on Lake Erie tributaries, they showed little spatial variability in the sampling sites within a single system. For example, in the 2002 study, only one site was sampled on each watershed. This approach creates difficulty in determining appropriate restoration and mitigation techniques for an individual watershed. Also, the temporal intensity of data collection used in this study, three samples every day for 9-16 years, may be unrealistic for watershed managers who want to get a rapid overview of a particular system.

Although many water quality studies exist that incorporate factors such as land use, discharge, and the effects of influx of lake water, few studies examine one Lake Erie tributary as an entire system, from the headwaters to the wetland estuary complex. In this study, I examine how land use, water level fluctuations of Lake Erie, and stream discharge all affect seasonal nutrient concentrations and loading to Crane Creek, a small agricultural tributary of Lake Erie. Crane Creek was an ideal study site for several reasons. First, it directly affects important coastal wetlands on Lake Erie, one of the critical habitats of the Great Lake Basin. Also, since part of Crane Creek is on a national wildlife refuge, understanding how land use practices in the rest of the watershed will affect the wildlife of the Refuge can help determine future management practices. Finally, Crane Creek is similar to many small, Midwestern agricultural streams, and the findings could be applicable to other systems in the region.

There were three major objectives in this research. The first objective was to characterize current water quality of Crane Creek for one growing season, documenting both seasonal and spatial variation. The second objective was to characterize nutrient loading and delivery in Crane Creek, incorporating land use data (GIS and rapid assessment physical stream surveys) and discharge measurements. Finally, the last objective was to compare the seasonal chemistry of lower Crane Creek to that of the adjacent diked pools on the Ottawa National Wildlife refuge. This comparison will help determine the implications of hydrologically reconnecting the diked wetlands to the estuary of Crane Creek in terms of potential for biological diversity.

II. Methods and Materials

A. Study Site

Crane Creek is a small tributary in the western basin of Lake Erie. It flows in a northeasterly direction through Ottawa, Wood, and Lucas counties in northwestern Ohio to the Ottawa National Wildlife Refuge, located about 32 km east of Toledo. Within the Refuge, Crane Creek empties into Lake Erie through a freshwater estuary. Crane Creek itself is fed by three named tributaries: Ayers Creek, Henry Creek, and Little Crane Creek. The mainstem is approximately 32.2 km long, and the entire watershed is approximately 143.5 km^2 (Wells 2001).

Soil type is relatively homogeneous within the Crane Creek watershed, and the topography is relatively flat. Elevation ranges from approximately 175 meters above sea level at the estuary to 196 meters above sea level in the headwaters. The watershed's soils originated from glacial till and lake and beach sediments deposited during the last glaciations. This watershed was originally part of the Black Swamp that stretched over much of Northern Ohio. In the late 1800s, the combined forces of railroad construction and efficient drainage caused the area to be drained and opened up for commercial and residential development (Herdendorf 1992). The mucky soils found today are high in organic matter and clay. The distance from the soil surface to bedrock can be over one meter (Ohio DNR 1996).

Crane Creek experiences a typical Midwestern temperate climate. The average annual precipitation in the Crane Creek watershed is 84.3 cm/year, including 48.3 cm during the growing season. Air temperature in the watershed ranges from -20.6º C to 35.0º C, with an average temperature of -4.5º C in January to an average temperature of

22.8º C in July (National Weather Service Forecast Office 2005). This climate, along with the soil type, provides ideal conditions for the agricultural practices that dominate the watershed. The Crane Creek watershed's landcover is nearly 80 percent agricultural, 6 percent urban, and only 2 percent forested. Most of the agriculture surrounding Crane Creek includes farming of soybean and corn. Many agricultural fields drain into Crane Creek by way of farm tiles. Particularly in the upper reaches, where much of the stream has been straightened and converted to agricultural ditches, water in the river channel is primarily runoff from crop fields.

Residential use is also a significant land use in the watershed. In the last few decades, urbanization in the area has increased greatly (US EPA 1995). Crane Creek passes through one urban center, the small town of Milbury, and several subdivisions and neighborhoods. Correspondence with Janet Hageman from the Ohio EPA's Division of Surface Water indicated that there are several points of unsewered input to Crane Creek from surrounding towns and trailer parks (Hageman 2005). Other minor land uses in the area include cattle pastures, limestone mining, and oil drilling. Since these practices are either more scattered within the watershed or near the boundaries of the watershed, it is unclear what influence they might have on the stream.

The river returns to a more natural sinuosity in the Ottawa National Wildlife refuge, where the channel is primarily surrounded by wetlands. Some of these wetlands are hydrologically connected to Crane Creek. However, many wetland units on the Refuge are isolated from the stream by earthen dikes in order to control water levels. The water levels in the pools are occasionally drawn down to promote growth of wetland vegetation that provides food and habitat for Great Lakes waterfowl. These diked units house much of the primary productivity on the Refuge and provide habitat for migratory birds. Most of the estuary of Crane Creek is also diked, except for a small opening where it is allowed to exchange waters with Lake Erie. The wetland estuary complex represents some of the last intact wetlands on the Western Lake Erie shoreline. As mentioned earlier, these coastal wetlands provide habitat for endemic Great Lakes species, including many species of freshwater clams and fish. The managers of the Refuge use Crane Creek not only to attract waterfowl to the area, but also as a water source for the diked wetland units.

Seiches across Lake Erie also affect to the Crane Creek system. Seiches are wind-initiated fluctuations of the lake levels that can change the flow of Crane Creek and affect water quality. During a seiche event, water from Lake Erie flows into Crane Creek, backing up the river water for several miles. The lake level fluctuations typically follow a 12 to 14 hour period. Major storm events can cause these oscillations to become more frequent or more extreme. These seiches tend to be strongest in the summer months and have been known to drive water level fluctuations of up to two meters in one day (Herdendorf 1987).

Six study sites were chosen on the Refuge and Lake Erie (Figure 1, Table 1) to monitor the water quality of Crane Creek on the refuge, the associated diked pools, and of the lake. Initially, two agricultural ditches were monitored on the refuge, but when they were found to have little to no contribution of flow to Crane Creek, those sites were discarded. One site (Site 1) was chosen at the mouth of Crane Creek. Two sites on the refuge (Sites 2 and 3) were upriver of this site. Two of the diked pools (Sites A and B) were sampled to serve as a comparison for Crane Creek. Pool 2A is about 65 acres, while Pool 2B is approximately 95 acres. Water level drawdown was attempted in Pool 2A in 2004, but was difficult because of the large amounts of rain that summer. Pool 2B was drained in August 2005. Finally, a site on Lake Erie (Site C) in nearby Metzger Marsh was selected to be able to compare near-shore lake water to river water.

Outside of the Refuge, twelve study sites in the Crane Creek watershed were selected to represent the diversity of land use on the river (Figure 2, Table 1). Sites included the two main tributaries, Henry Creek and Ayers Creek, and several sites in agricultural and residential areas, including one site in the middle of Milbury (Site 8B). One site (Site 10D) was near a major interstate and truck stop to capture the effects of runoff from roads. Another site (Site 6) was chosen near a cattle enclosure to determine the effects that runoff from cattle may have on water quality. Finally, since there is a general lack of data on nutrient loading from point sources (Richards and Baker 1993), one site (Site 10C) was chosen near a large pipe discharging effluent from the town of Stony Ridge.

B. Hydrologic Methods

Two instruments were used to measure current velocity in Crane Creek. Velocity was measured at most sites on Crane Creek using the Marsh-McBirney Model 2000 portable flowmeter. Taking into account the stability of the zero measurement (±0.015 meters/second), the flowmeter has a precision of ±2% of the reading. Velocity measurements were taken at regular intervals along a horizontal cross section of the channel. The high amount of sediment and typically low flows at each site caused some difficulty in obtaining accurate discharge measurements. To measure the velocity for Crane Creek at Stange Road (Figure 1, Site 3), a standard AA current meter attached to a bridgeboard was used. Within the time frame of this study, velocity could not be determined for the mouth of Crane Creek (Figure 1, Site 1). Flow there is typically bi-directional on a 12-14 hour period. Also, flow could not be measured at Crane Creek across from Pool 2A (Figure 1, Site 2) on the refuge due to high water levels. At the other sites, when the water velocity could not be measured due to instrument failure or weather conditions, the flow was estimated using a depth-velocity relationship for that particular site.

C. Nutrient measurements

All water samples were taken as grab samples, labeled, and immediately placed in a cooler on ice. Upon returning to the lab, they were kept frozen and processed as soon as possible (typically within 1-2 weeks).

Samples were taken once to twice a month from May to November 2004 and April to June 2005. Before processing, each sample was filtered through a 0.45 µm membrane filter to reduce the effects of turbidity or suspended solids. Samples were analyzed for the following parameters: gilvin, SRP (soluble reactive phosphorus as P), ammonia nitrogen, nitrite-nitrate nitrogen, and TP (total phosphorus as P). All concentrations reported here for phosphorus and dissolved nitrogen are in units of mg L^{-1} as elemental P or N. The Hach DREL/2000 Water Quality Laboratory spectrophotometer was used to make all photometric determinations (Hach 1993). Gilvin was measured using a simple absorption scan at 440 nm. The ascorbic acid method was used to measure phosphorus as orthophosphate or soluble reactive phosphorus (SRP). Ammonia nitrogen, referred to hereafter as ammonia, was measured using the salicylate method. Nitrate-nitrite nitrogen was measured using the cadmium reduction method. If the nitrite-nitrate nitrogen levels were high (above 0.44 mg L^{-1}), the nitrate-nitrite nitrogen level was also measured using the ultraviolet nitrate scan method with the UVI Double Beam UV/Vis Spectrophotometer (Thermo Spectronic 2000). Since nitrite concentrations are expected to be low from previous studies (ex. Richards and Baker 2002), this measurement will hereafter be referred to as nitrate. Total phosphorus (TP) was measured on a single date (28 June 2005) at all of the sites to be related to concurrent turbidity, suspended sediment, and suspended solids measurements. This measurement was also made using the Hach spectrophotometer and the acid persulfate digestion method. Deionized water samples were analyzed as controls along with each of the sample sets to ensure accuracy of collected data.

D. Physical and Chemical Measurements

The following measurements were taken along with water samples at each of the sites: conductivity, temperature, and alkalinity. All measurements were made in the thalweg of the stream channel. Conductivity and temperature were measured using the Hach Sesion5 conductivity meter, which was calibrated monthly. Alkalinity was measured using sulfuric acid with a digital titrator.

In addition to monthly measurements of these three parameters, single time measurements of suspended sediment, turbidity, and suspended solids were made. As noted above, all of these measurements were made in June 2005. Suspended sediment was measured using a handheld sediment sampler to collect a water column sample. A small amount of this sample was filtered through a pre-weighed filter, dried, and reweighed to measure amount of suspended sediment (Standard Methods 1995). These samples were also measured for Formazin Turbidity Unit (FTU) by the spectrophotometer. Samples for suspended solids were collected from the thalweg of the stream. These samples were analyzed on site with the field Hach spectrophotometer using the photometric method for suspended solids (Hach 2000).

Chemical oxygen demand (COD) was measured twice, once for seasonal differences and once for spatial differences. For seasonal difference, all samples from Crane Creek at Elliston Road (Site 4) from May 2004 to June 2005 were analyzed. For spatial differences, all samples from June 2005 were analyzed. June 2005 was chosen because the demand for oxygen would be the highest when there are low flows and high temperatures. The method used for COD analysis was the reactor digestion method.

Finally, each site was assessed for substrate, riparian vegetation, and surrounding land use. This survey was adapted from the "Rapid Assessment Protocols for Use in Streams and Wadeable Rivers" physical characterization and water quality field data sheet developed by the EPA (Barbour et al. 1999). These data were collected at each of the sites outside the refuge in August 2005 and for the sites in the Refuge in September 2005.

Table 2 provides a summary of all of the measurements (hydrologic, nutrient, physical, and chemical parameters) and also provides a time frame for each measurement.

E. Land Use Data

Two land use coverages for the entire Crane Creek watershed were obtained. One coverage was created in 1994 by the Ohio Department of Natural Resources. The other was a coverage created by the Coastal Land Services in 2000. Since it was the most recent coverage, the Coastal Land Services raster cover was used for determining land use in the Crane Creek watershed.

A watershed shape file for the entire Crane Creek watershed was created using Geoprocessing Wizard of ArcView 3.3. Using a digital elevation map, watershed boundaries were also created for each water quality study site on Crane Creek in order to calculate the area drained by each site. The creation of these watersheds allowed a calculation of percentage of each type of land use within each watershed. It also allowed for a comparison of land use in the Crane Creek watershed and the encompassed site watersheds between 1994 and 2000.

F. Data Analysis

Data Desk© version 6.1 and Microsoft Excel© were used to perform all statistical analyses and create associated figures and tables. The study sites were divided up into groups for several of the tests. 'Crane Creek upstream' was defined as all of the sites above Elliston Road (Site 4), where the seiche effect was minimal. 'Crane Creek downstream' was defined as all of the sites from Elliston Road to the mouth of Crane Creek (Site 1) on Lake Erie. In these sites, the lake level could have significant effects on the water chemistry and discharge patterns.

Data analysis was divided up into five components: longitudinal trends, temporal trends, effects of discharge on water quality, effects of land use on water quality, and interrelationships among variables.

1. Longitudinal Trends

To determine the differences in all of the parameters (including physical parameters, chemical parameters, and nutrient concentrations, loads, and yields) between different parts of the study area, four major comparisons were used. First, ANOVA and Scheffe post-hoc tests were used to compare Crane Creek upstream, Crane Creek downstream, Pool 2A, Pool 2B, and Lake Erie. Henry Creek at Broadway (site 10C) was excluded from Crane Creek upstream for this test because it was a strong outlier that could mask or exaggerate the differences between the groups. Second, Crane Creek upstream mainstem, Crane Creek downstream mainstem, Henry Creek, and Ayers Creek were compared. In this analysis, ANCOVA and Scheffe post-hoc tests were used to account for discharge differences in these groups. Third, the most downstream sites on Henry Creek and on Ayers Creek were compared using a paired t-test over sample dates to determine the differences in contribution of each tributary to Crane Creek. For this comparison, data from days in which both tributaries were not sampled were excluded. Finally, boxplot distributions of each of the parameters for each of the sites on Crane Creek were compared to find significant trends moving upstream on the river.

2. Temporal Trends

Two approaches were used to determine how the parameters changed over time. First, scatterplots were created of each parameter over time to compare Crane Creek upstream, Crane Creek downstream, Lake Erie, Pool 2A, and Pool 2B. Henry Creek and Ayers Creek did not have separate groups because these tributaries should not show temporal patterns distinct from Crane Creek upstream. Henry Creek at Broadway was again excluded as an outlier. The early sample dates (May 2004 and the first sample date in June) were also excluded because all sites were not sampled on these dates. Second, ANOVA tests were performed on the Crane Creek data grouped by seasons to determine the differences between seasons, separating Crane Creek upstream and downstream.

3. Effects of Discharge on Water Quality

Two loading calculations were made to determine nutrient input rates into and out of the downstream part of the Crane Creek system. First, an estimate of yearly nutrient loading from Crane Creek at Opfer Lentz (Site 5) was determined by estimating an average daily load from all the sample dates and extrapolating that to one year. To determine yearly loads out of the refuge and into Lake Erie, discharge data from Crane Creek at Elliston (since, on an annual basis, nearly all of the stream discharge at Elliston would eventually be expected to exit to Lake Erie) was combined with nutrient concentration data from Crane Creek Across from Pool 2A (Site 2). The nutrient concentrations from the mouth of Crane Creek were not used to calculate loading out to Lake Erie because the bidirectional flow can drastically change nutrient concentration in a twelve-hour period.

Three additional approaches were used to determine the relationship of discharge to the nutrient concentrations, loadings, and yields on Crane Creek upstream. Crane Creek downstream was not included in this analysis because comprehensive discharge data were not available. First, a linear regression on nutrient concentration and either watershed area or discharge (depending on which had the better fit) was performed. Second, a linear regression of discharge against each of the nutrient loadings was used to determine the rate at which nutrient loading increased with discharge. Finally, data for a day with high flows (30 October 2004) and for a day with low flows (12 August 2004) were isolated. Graphs of nutrient concentrations, loads, and yields by site on those days were created to compare trends based on low or high discharge.

4. Effects of Land Use on Water Quality

Using multiple linear regression analysis, the relationship between land use and each of the parameters was determined, controlling for watershed area and discharge. Both percentage of agricultural land use and percentage of urban land use were used in the analysis once it was determined that the autocorrelation was not significant. Agricultural land use and urban land use were chosen because they have been shown to have a strong influence on nitrate and phosphorus concentrations (Osborne and Wiley 1988, Tong and Chen 2002) and they produced the best fit.

5. Interrelationships among Variables

This analysis was used to determine if there were potential interactions or associations between the measured parameters. The first part of this analysis consisted of examining the relationship between one-time measurements of turbidity, suspended solids, suspended sediments and total phosphorus. The second part of this analysis consisted of using a Person-Product Moment Correlation chart to look for strong pair-wise associations among all of the parameters, including the one-time measurement parameters. Nutrient loads and nutrient yields were treated separately in two different correlation charts.

III. Results

A. Current Water Quality Status of Crane Creek

1. Site Characteristics for Crane Creek

Crane Creek is a highly impacted catchment in terms of conductivity, alkalinity, and chemical oxygen demand (Table 3). Water temperature of Crane Creek ranged from 1.3º C to 32.8º C, with an average of 19.4º C. Water temperature followed a typical pattern for a small agricultural stream, peaking in the summer and dropping to near freezing in the fall. Conductivity ranged from 106-2490 µs cm^{-1}, with a mean conductivity of 738 µs cm^{-1}. Alkalinity ranged from 50-370 mg L^{-1}, with an average of 168 mg L^{-1}. Gilvin ranged from below detection to 0.47 abs cm^{-1}, with an average gilvin of 0.02 abs cm^{-1}. Chemical oxygen demand measurements (data not shown) taken at Site 4 ranged mostly from 15-30 mg L^{-1}, with one measurement above 60 mg L^{-1} in October, and showed no major temporal trend.

Crane Creek also has elevated nutrient concentrations. Table 4 summarizes the mean values and ranges of nitrate-nitrite nitrogen (nitrate), soluble reactive phosphorus (SRP), and ammonia nitrogen (ammonia). Nitrate concentrations ranged from 0.01 to 4.26 mg L^{-1}, with a mean of 0.49 mg L^{-1}. SRP ranged from below detection to 1.70 mg L^{-1}, with a mean SRP value of 0.12 mg L^{-1}. Ammonia concentrations ranged from below detection to 13.2 mg L^{-1}, with a mean of 0.34 mg L^{-1}.

A few significant correlations were detected among physical parameters, chemical parameters, and nutrient concentrations, and removing outlier measurements from Henry Creek at Broadway made these relationships more apparent. Alkalinity and conductivity were strongly correlated (R^2=0.76), as were gilvin and COD (R^2=0.67). SRP and ammonia concentrations were also somewhat correlated with each other (R^2=0.50), but not with nitrate. Also, SRP and total phosphorus (TP) had a strong linear relationship in the Crane Creek watershed (R^2=0.96) yielding the following regression equation:

$$TP = (1.06*SRP) + 0.04$$

Although this relationship likely is an underestimate since measurements were taken at low stream discharge when particulate material would be minimal, it is important to note because SRP can often be taken up by algae and converted to TP (Krieger 2003) or locked up in particulates in the sediment (Richards and Baker 1993). On the date it was examined, total phosphorus was also strongly correlated with conductivity, alkalinity, and gilvin, but had little to no correlation with measurements of suspended sediment or turbidity. Since only a few measurements were taken of FTU and turbidity, these results must be treated with some caution.

2. Longitudinal Trends

Crane Creek is more degraded upstream than downstream, as evidenced by higher values of all water quality parameters at upstream sites as compared with downstream sites (Tables 3 and 4). ANOVA and Scheffe post-hoc comparisons of Crane Creek upstream and Crane Creek downstream that included the diked pools and the lake site showed that the upstream sites were significantly higher in mean conductivity, alkalinity, and SRP than the downstream sites (Table 5). Upstream and downstream mainstem comparisons (without the tributaries) also showed significant differences in ammonia loading.

Henry Creek was the most degraded part of Crane Creek for several water quality parameters. Results from the ANCOVA and Scheffe post-hoc tests for the four parts of

Crane Creek (mainstem upstream, mainstem downstream, Henry Creek, and Ayers Creek) indicated that Henry Creek had significantly higher mean conductivity, significantly higher mean ammonia concentrations, and strongly higher mean alkalinity than the other three streams (Table 6). It should be noted here that ammonia concentrations were significantly influenced by tributary rather than discharge. Henry Creek also had higher mean SRP concentrations than two out of three of the other streams.

However, Henry Creek and Ayers Creek were nearly equivalent in terms of loading contributions to Crane Creek. The paired t-test over sample dates found significant differences only in temperature (df=9, $p \leq 0.001$) and gilvin (df=10, p=0.027) between the tributaries. Ayers Creek had higher mean water temperature, and Henry had higher mean gilvin. There were also strong but not significant differences in conductivity (df= 10 p=0.180) and loading of SRP (df=8, p=0.129). Henry Creek had both higher mean conductivity and higher SRP loading. The sites on Henry Creek and Ayers Creek (Sites 7a, 9c, and 10c) had the highest values of most of the nutrient, physical, and chemical parameters. In particular, Henry Creek at Broadway (Site 10c) seemed to be a major contributor for all of the nutrients (Figure 3).

Across all sites, trends could be observed with increasing watershed area. Average conductivity, alkalinity, and COD all decreased with increased watershed area. SRP and ammonia loading increased with increasing discharge and watershed area, but nitrate loading showed more interesting variation. Nitrate concentration, loading, and yield all were markedly higher at Crane Creek at Elliston (Site 4) and then dropped dramatically once Crane Creek entered the Refuge (Figure 4). Since nutrient loading often increases as watershed area increases, observed decrease in nutrient loading may indicate lake level and wetland effects on the downstream sites.

3. Seasonal Trends of nutrient concentrations

Over time, the downstream sites on Crane Creek were less variable in water quality parameters than the upstream sites (Table 7). The downstream sites only showed significant seasonal differences for gilvin, which was higher in spring 2005 than other times of the year. The upstream sites, on the other hand, showed significant seasonal differences for conductivity, alkalinity, and SRP. Conductivity increased throughout the summer and peaked in September. Alkalinity also increased through the summer but peaked in November. SRP concentrations were low in the spring and then increased in the summer and fall. Both Crane Creek upstream and downstream showed a weak trend of increased nitrate concentrations in the spring.

Because sites 4 and 10c were outliers in several analyses, they were selected to look for site-specific temporal trends to determine when they would be of greatest concern. Extreme outliers in these sites made it difficult to determine temporal trends with certainty. For Site 4, nitrate, ammonia, and phosphorus concentrations all peaked in mid to late summer, when stream discharge would be low and primary productivity would be high. For Site 10c, nitrate concentrations peaked in the summer, while phosphorus and ammonia concentrations peaked in early fall.

B. Nutrient Loading and Delivery on Crane Creek

Annual loading estimates into and out of the downstream system showed that more nutrients entered the wetland-estuary complex than exited the refuge to Lake Erie. Using the average of all of the nutrient loading measurements from Crane Creek at Opfer

Lentz (Site 5), it was estimated that approximately 2094 kg/year of DIN (1846 kg/year of nitrite-nitrate nitrogen and 248 kg/year of ammonia nitrogen) and 498 kg/year of SRP entered Crane Creek at Elliston Road over the study period. Comparatively, approximately 1270 kg/year/km2 of DIN (792 kg/year of nitrite-nitrate nitrogen and 479 kg/year of ammonia nitrogen) and 100 kg/year of SRP, apparently exited the refuge and entered Lake Erie over the study period. In estimating TP loading from the TP: SRP ratios at Opfer Lentz (0.38/0.30=1.27) and Crane Creek across from Pool 2A (0.13/0.03=4.33), 632 kg/year of TP would enter Crane Creek at Elliston Road and 433 kg/year of TP would exit the refuge and enter Lake Erie.

In addition to wetlands and lake levels, discharge and watershed area also had strong relationships to nutrient loading and delivery. In the upstream sites, nitrate behaved very differently from ammonia and SRP. All three of the nutrient concentrations had a poor correlation with discharge or watershed area (in the case of nitrate concentration, discharge was a better fit than watershed area), but some patterns could be seen (Figure 5). As discharge increased, nitrate concentrations increased. As watershed area increased, ammonia and SRP concentrations decreased. In linear regression analysis (Figure 5), nitrate loading increased more sharply with discharge than either ammonia loading or SRP loading. It should also be noted here that ammonia loading had a low correlation with discharge.

The varying levels of stream flow also affected nutrient concentrations, loadings, and yields (Figures 6-8). As expected, the high flow day was higher in overall nutrient concentrations, loads, and yields than the low flow day. However, the sites with the highest values changed when stream flows were low. On low flow days, many of the sites on the tributaries and headwaters (particularly sites 7a, 10c, and 11d) had the highest nutrient concentrations, loads, and yields of all of the sites. Interestingly, ammonia concentrations, loads, and yields were highest on site 10c for both high and low flow days (Figure 8).

Unlike discharge and watershed area, the regressions between nutrient concentrations and land use had poor R^2 values and showed few significant relationships (Table 8). In multiple regressions of nutrient concentration with percentage of agricultural land, percentage of urbanized land, discharge, and watershed area, percentage of agricultural land showed a significant negative effect on SRP and ammonia. Percentage of urbanized land also showed a significant negative effect on SRP. Neither percentages of agricultural nor urbanized land showed significant correlations to nitrate concentrations.

Conductivity was the only other water quality parameter that had a strong relationship to percent agricultural land, watershed area, and discharge. Conductivity increased as percentage of agricultural land increased and watershed area and discharge decreased.

C. Water quality of the diked pools

The pools most closely resembled near-shore Lake Erie and the downstream sites on Crane Creek in water quality. There were no significant differences between the lake site, Pool 2A, and Pool 2B for all of the parameters, and the downstream sites were only significantly different from the pools in conductivity (Table 5). In comparing Pool 2A to Pool 2B, the levels of nutrients and other parameters were generally higher in Pool 2A than Pool 2B (Tables 3 and 4) and Pool 2A was more temporally variable than Pool 2B.

The nutrient concentrations tended to peak in the summer for Pool 2A. Since Pool 2B did not have as many sample dates, it was difficult to determine when its nutrient concentrations would peak, though the available data suggest peak values in late fall and early spring.

IV. Discussion

A. Current Water Quality of Crane Creek

Although Crane Creek is a degraded system compared to many streams in other parts of the U.S., Crane Creek's water quality is comparable to other Lake Erie tributaries (Table 9). With the exception of Grand River, the mean nitrate concentration in Crane Creek was lower than in other catchments. Mean SRP concentration, on the other hand, was higher than other catchments. Comparison of Crane Creek to Old Woman Creek (OWC), both similar to Crane Creek in size and located in northern Ohio, likewise shows that Crane Creek has lower nitrate concentrations but higher SRP concentrations than OWC (Table 10). Also, while ammonia concentrations remained nearly the same through the wetlands of Crane Creek, the ammonia concentrations of OWC increase through its wetlands, apparently due to resuspension of anaerobic sediments (Krieger 2003).

When current and historical Crane Creek data are compared (Table 11), long-term trends in nutrient concentrations become evident. Nitrate concentrations decreased in the 1990s in Crane Creek, but have again increased in the last decade. These changes in nitrate may be due to changes in atmospheric deposition of nitrate, changes in fertilizer use, or changes in crops (Richards and Baker 1993). Average phosphorus concentrations, on the other hand, have clearly decreased. This result agrees both with observed reduced phosphorus loading to Lake Erie from tributaries (Fraser 1987) and with general Lake Erie tributary trends (Richards and Baker 1993). Finally, ammonia has also decreased significantly. These historical trends, while promising, must be interpreted with some caution because historical data may not be representative of the watershed. EPA monitoring often targets sites known to have low water quality, rather than sampling sites throughout the watershed (Ohio EPA 1993 1 and 2). As a result, comparing historical water quality data to more representative values may be misleading.

Furthermore, simply looking at average nutrient concentrations over all of Crane Creek or at one particular site ignores inherent spatial and temporal variation. It is clear that the upstream sites and the downstream sites behave differently, a trend that has been documented in other Lake Erie tributaries (Krieger 2003). The upstream sites on Crane Creek are much more spatially and temporally variable than the downstream sites due to localized land use, small channel size, and lack of baseflow. Henry Creek is the highest area of concern for Crane Creek although, in terms of loading and delivery, Ayers Creek is equally problematic. These tributaries, which seem to be heavily impacted by point source nutrient inputs, should be targeted for restoration.

In addition to higher discharge, there are three probable causes of the higher water quality, temporal consistency, and spatial consistency of the downstream sites. One is the stalled flows observed in Crane Creek at Elliston Road (Site 4), which could allow phosphorus to be absorbed into the sediments. The wetlands, which begin near US Route 2 on the Refuge, could be another cause of the consistently lower values in the downstream sites. Wetlands both transform and accumulate nutrients (Mitsch et al. 1994), dependent on water levels, amount of inflow from the lake, soil conditions, and existing biota (Wang and Mitsch 1998). SRP is often taken up by biogeochemical processes such as uptake in phytoplankton and bacterioplankton and converted to TP (Heath 1992, Mitsch et al 1994, Krieger 2003). Phosphorus is also transformed by both abiotic geochemical reactions and physical sedimentation (Heath 1992, Mitsch and Wang 2000).

Nitrogen entering wetlands in the form of organic nitrogen is subject to ammonification, assimilation by plankton, and reduction to atmospheric nitrogen (Heath 1992). Finally, lake water inflow could be causing a dilution effect in the downstream sites. Since lake water is often of higher water quality than the tributaries, the influx of lake water dilutes nutrients and other dissolved materials (Krieger 2003). Restoration in these downstream sites would be difficult unless the dynamics of the low discharge and lake water inflow were well understood.

B. Nutrient Loading and Delivery

1. Nonpoint sources of Nutrients

Nitrate seems to enter the Crane Creek watershed primarily from nonpoint sources. Nitrate concentrations increased with increasing discharge (Figure 5), indicating more nitrate being washed off the land during increased flows (Castillo et al. 2000). Also, in a linear regression with discharge (Figure 5), the coefficient for nitrate loading was greater than 1, indicating that the loading was increasing with increasing runoff.

The major nonpoint source in Crane Creek seems to be runoff from both agricultural fields and residential lawns. The stream survey showed that many of the study sites are surrounded by lawns or agricultural fields, specifically soybean fields, and a local resident mentioned heavy pesticide spraying in agricultural fields. During storm events, these lawns and fields could have runoff of fertilizers and pesticides high in nutrients, sending high amounts of nutrients into the watershed (Osborne and Wiley 1988). From stream survey observations, other nonpoint sources could include road runoff from places like the truck stop at Crane Creek at Warns and runoff from the cow pasture at Rieman, though no direct discharges were observed at these sites.

Some of these sites were buffered by a riparian zone of trees or tall grasses, which might help mitigate the nutrient influx and promote higher in-stream biological diversity (Stauffer et al. 2000). However, since these riparian zones were often only 10-50 feet wide and the grasses are seasonal, the potential for unmitigated runoff is significant. Particularly in highly urbanized locations and several of the soybean fields, there was little to no riparian vegetation, which could allow runoff from the roads and the fields to directly enter the channel. Riparian areas can be of limited importance if upstream sites are left unbuffered (Osborne and Wiley 1988).

2. Point sources for nutrients

Unlike nitrate, point sources seemed to be the dominant source for SRP and ammonia. Both SRP and ammonia concentrations decreased as discharge increased (Figure 5), indicating dilution. Also, the loading coefficients for SRP and ammonia were both less than one (Figure 5), indicating that the nutrient concentration input was fairly constant and highly subject to dilution. The dilution of SRP in high flow periods has been well documented (Osborne and Wiley 1988, Castillo et al. 2000). Still, this result is somewhat surprising, since previous studies have suggested that point sources are not a major source of phosphorus in Lake Erie tributaries and often constitute less than 25% of the annual loading (Richards and Baker 1993, Castillo et al. 2000).

The five major point sources on Crane Creek were noted through both the stream survey and communications with Janet Hageman of the Ohio EPA. One is Crane Creek at Williston Road and Wildacre Road, which receives storm discharge from the town of Curtice. The second is a rundown trailer park at Billman Road and Young Road, which contributes raw sewage to Ayers Creek and another small tributary of Crane Creek. The third point source is from the town of Milbury in the middle of the Crane Creek watershed. Milbury is sewered, but experiences combined sewer overflows during heavy rainfall events, the last recorded of which was in May of 2005. The fourth source is discharge from the town of Lemoyne at a point close to the intersection of Lemoyne Road and Truman Road. Finally, the fifth point source documented both by the EPA and the stream survey was a discharge pipe emptying into Henry Creek at Broadway Road. This pipe, which is responsible for many of the elevated nutrient levels in this study, is untreated discharge from the town of Stony Ridge (Hageman 2005, Ohio EPA 1993 (1) and (2)). This discharge pipe should be targeted for enforcement because it has detrimental effects on Crane Creek's water quality, particularly when flows are low.

The importance of point sources in Crane Creek is indicated by the differences in nutrient concentrations, loads, and yields during high and low discharge. On a high flow day (Figure 6), nitrate was highest in the largest sites. SRP and ammonia, while high in some smaller sites, were also high in sites with large watershed area on high flow days (Figures 7 and 8). However, on a low flow day, nitrate, SRP, and ammonia were highest in the tributaries and at the most upstream sites (Figures 6-8), especially in Henry Creek at Broadway (Site 10c). For ammonia, Henry Creek at Broadway has a major influence during both high and low flows. This trend indicates that point sources are of some concern on high flow days, but of particular concern during low flow days. Since Crane Creek typically experiences extremely low flows during the summer, it is important to understand that point sources control water quality during those periods.

3. The effects of land use on water quality

The land use regression showed a poor relationship of the nutrient concentrations to land use percentage. In many cases, discharge or watershed area was a better predictor of nutrient concentrations than land use. For example, in the case of nitrate, discharge and watershed area were significant factors in predicting discharge while percentage of agricultural land was not. In cases where land use was a significant factor in determining nutrient concentrations, increasingly urban and agricultural land use percentages resulted in decreasing nutrient levels. This result is counter to the observations made within the watershed and previously mentioned studies on land use and water quality, which suggest that urban and agricultural land use increase nutrient concentrations.

This analysis does not indicate that land use is unimportant in determining nutrient concentrations. The high nutrient concentrations in the stream cannot be from point sources alone, and agricultural lands have been shown to have a strong association with both nitrate and phosphorus (Tong and Chen 2002). However, there are other factors to consider in the Crane Creek watershed. First, strong effects of land use on variation of nutrient concentrations are not possible because most of the sites are dominated by agriculture and there is little variation of land use. As mentioned earlier, nearly the entire watershed is agricultural, with fairly uniform distribution of urban areas. Second, since discharge is such a dominant factor, the influence of land use is likely not detectable statistically. Third, the number of point sources scattered throughout the watershed with similar land use distribution may confound the analysis.

Comparing the 1994 land use coverage from the Ohio DNR to the 2000 land use coverage from Coastal Land Services, it is evident the watershed is becoming increasingly urbanized. Agricultural and forested lands have been replaced by residential and commercial lands due to urban sprawl from Toledo and other surrounding cities. Particularly, the central and upper reaches of Crane Creek show significant increase in urbanization from 1994, and can be projected to be even higher today (Ohio DNR 1994 and Coastal Land Services 2000). Many studies have shown that urbanization can have even more detrimental effects on water quality than agricultural land use (Osborne and Wiley 1988, Tong and Chen 2002). Thus, understanding the effects of increased urbanization will be essential in monitoring Crane Creek's water quality.

C. Comparison of Crane Creek to the diked pools

Pool 2A was higher than Pool 2B in every water quality parameter, although none of these differences were statistically significant. The differences in drawdown timings and dilution probably explain the elevated nutrient concentrations of Pool 2A. Pool 2A was drained during the summer of 2004 and was at low levels for most of the study period. Pool 2B was drained towards the end of the study and was at high water levels for most of the study period.

In comparing both pools to Crane Creek, it appeared that while Crane Creek upstream was significantly different from the pools in nutrient concentrations and other water quality parameters, Crane Creek downstream was not that different from the diked pools. If the pools were hydrologically reconnected to the creek to allow for more natural water level fluctuations, the data suggest that no significant change in nutrient concentrations or the physical parameters would occur. Particularly since the pools are currently filled with both rainwater and water pumped from Crane Creek downstream, there is little likelihood of changes in nutrient concentration or of potential threats to biological diversity.

However, conversations with the manager of the Ottawa National Wildlife Refuge, Doug Brewer, indicate that reconnecting some of the units hydrologically may not actually reconnect them to Crane Creek wetlands due to elevation and water levels. Pools 2A and 2B have been diked for so long that they have filled with sediment and are at a higher elevation than the channel. By taking out the dikes, the pools may lose their water to Crane Creek and become moist land, an ideal habitat for invasive species such as phragmites, purple loostrife, and rush. According to Brewer, while reconnection is ideal, it is currently impractical because it may favor expansion of invasive species that could thrive on the refuge (Brewer and Mason 2005).

A compromise might be found in creating a small point of connectivity from the diked pools to the wetlands, as has been done in other Lake Erie tributaries (Kowalski and Wilcox 1999). This connection would allow the waters to fluctuate more naturally with the dynamics of Crane Creek and Lake Erie, while continuing to promote the primary productivity of the diked pools, which is currently not reproducible in the channel of Crane Creek.

As discussed above, the downstream sites (and the diked pools) have higher water quality than the upstream sites on Crane Creek. This difference in water quality is at least partially due to the transforming effects of the wetlands and the dilution by lake water inflow. The loading calculations comparing nutrient yields into the downstream sites and out of the refuge (page 20) showed a decline in DIN and SRP loading before the water enters Lake Erie. DIN decreased approximately 824 kg/year (a 39% decrease) and

SRP decreased 398 kg/year (an 80% decrease), even though watershed area increased downstream. Estimating total phosphorus loading using the TP:SRP ratios, total phosphorus loading would decrease from 199 kg/year, a decrease of 31%. This is less than the 80% predicted by SRP, indicating that some of the SRP is being converted to other forms of P before entering Lake Erie. The estuary is acting as a processor of nutrients and it seems that the wetlands still connected are having a mitigating effect on the water quality of this highly impacted stream.

Clearly, these calculations are underestimates because they do not account for strong storm events, extremely low lake levels, organic forms of nitrogen, and total phosphorus transport at high discharge. Particularly, since discharge data was not available for the mouth of Crane Creek, these numbers must be taken with some caution. Nevertheless, they provide a useful point of comparison and indicate potential influence of lake level fluctuations and mitigating wetland effects.

V. Conclusion

The purpose of this study was to examine how land use, water level fluctuations of Lake Erie, and discharge all affect seasonal nutrient concentrations and delivery on Crane Creek. There were three major findings. First, both water level fluctuations driven by Lake Erie seiches and higher discharge make the downstream sites less spatially and temporally variable than the upstream sites. The downstream sites also had higher water quality than the upstream sites because of wetland transformation of nutrients and dilution from lake water inflow. Second, while agricultural and urban land use likely contribute nutrients from fertilizer use and urban runoff, point sources in the catchment seem to have a greater influence on water quality in Crane Creek, particularly in times of low stream discharge. The influence of varying patterns of land use was difficult to determine because homogeneity of the landscape and point sources confounded the analysis. Finally, within the lower estuary, water quality in the system was similar to water quality in surrounding diked pools. This similarity makes hydrologic reconnection of these wetlands a possibility, although physical constraints complicate the restoration process. These findings have implications for other small Lake Erie tributaries, particularly streams in agricultural catchments.

Rehabilitation of Crane Creek presents several possibilities. The main priority should be reduction and better regulation of the point sources, particularly sewage outfalls from surrounding urban centers such as seen in Henry Creek at Broadway. It is possible that, similar to other Lake Erie tributaries that have a significant number of point sources (Richards and Baker 1993), removal of point sources will reduce nutrient levels. The other major priority should be better management of agricultural land. While regulating agricultural land use may be more effective in controlling nutrient loading than controlling nutrient concentrations (Osborne and Wiley 1988), better agricultural practices like conservation tillage and reduction of fertilizer use could improve the water quality significantly, particularly in terms of nitrate and phosphorus (Fraser 1987, Richards and Baker 2002). For example, one study found that if fertilizer was simply applied parallel to crop rows in the spring instead of widely distributed the fall, total phosphorus in watersheds could be reduced (Richards and Baker 1993). Finally, as a preventative measure, increasing riparian vegetation will help buffer the stream from runoff from agricultural and urban areas. Whatever measures are taken, the focus should be on restoring ecosystem functions such as nutrient cycling, rather than creating short-term solutions (Moerke and Lamberti 2004).

Due to time and financial constraints, the scope of this study was limited. More sample dates, particularly to capture storm events and daily fluctuations in the watershed, would have created a more complete picture of nutrient variability on the watershed. In particular, storm events must be better studied because precipitation can contribute over 20% of water volume to the channel under low flow conditions. Also, since streams like Crane Creek are highly variable from year to year because of storm events and weather variability, this type of study should ideally be continued over several years (Richard et al. 1996, Krieger 2003). Finally, reliable stream discharge data on the downstream sites influenced by the lake would more accurately depict delivery to Lake Erie (Krieger 2003).

Figure 1: Digital orthoquad of the Ottawa National Wildlife Refuge. Enumerated sites (1-3) are sample sites on Crane Creek. Sites 'A' and 'B' indicate diked pools 2A and 2B sampled on the refuge. Site 'C' indicates sample location on Lake Erie by Metzger Marsh (original image created by Kurt Kowalski at USGS based on 1997 NAPP aerial photographs).

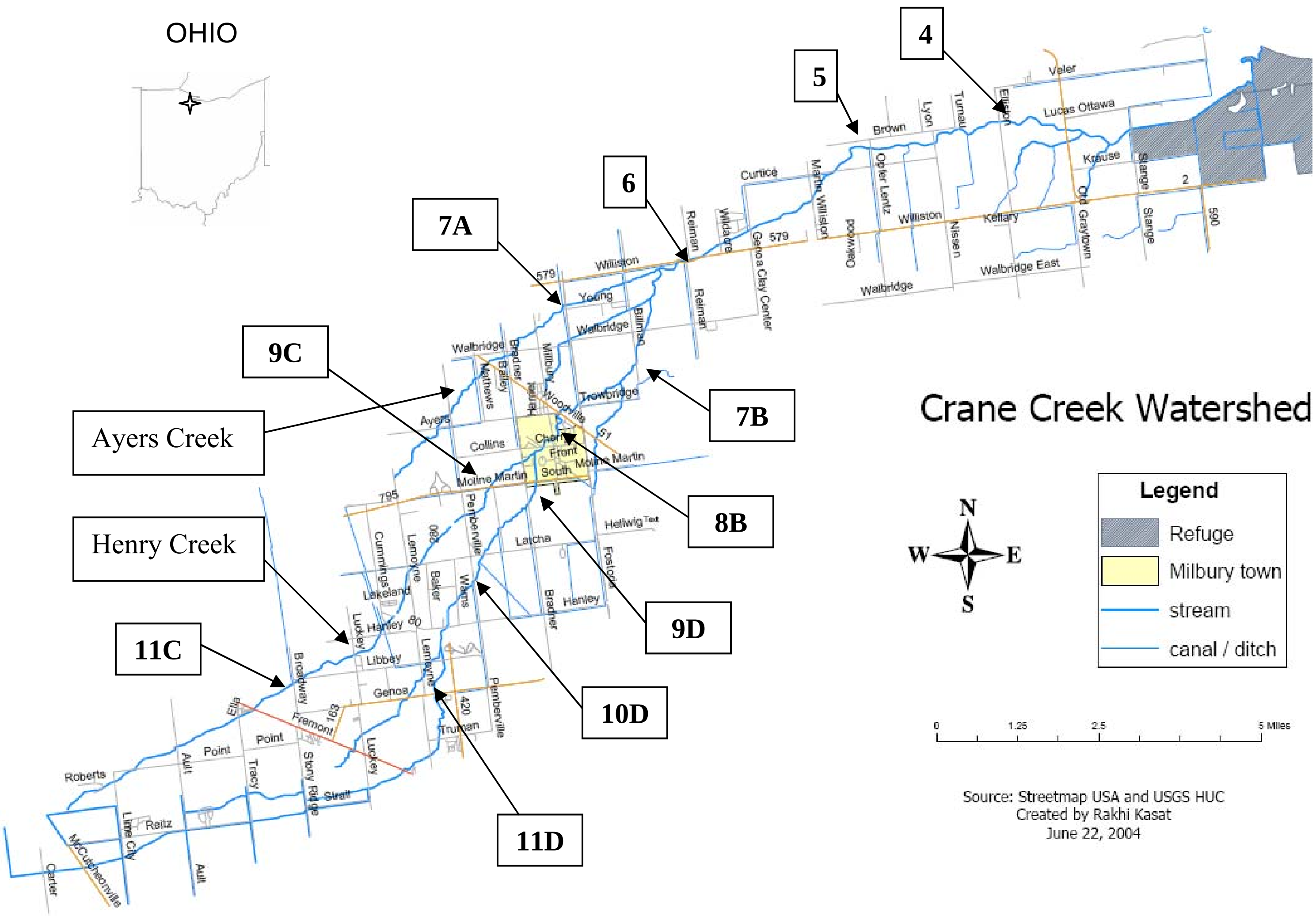


Figure 2: Map of the Crane Creek watershed. Eleven locations on Crane Creek outside of the refuge, identified numerically, were sampled from 2004 to 2005. Ayers Creek and Henry Creek are the two main tributaries of Crane Creek.

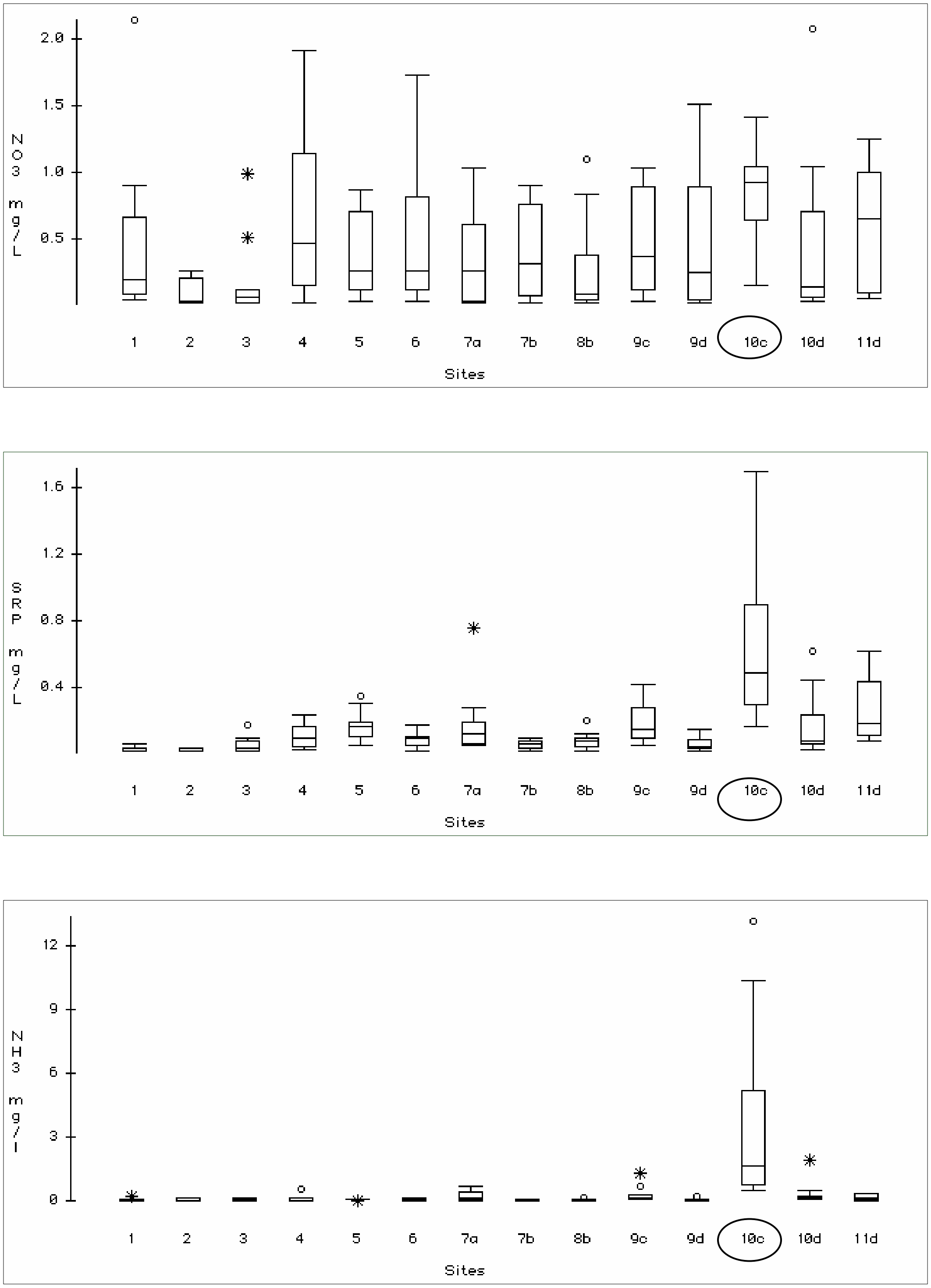


Figure 3: Boxplot distribution comparison of Henry Creek at Broadway (Site 10C) concentrations of nitrate, SRP, and ammonia to concentrations in the rest of Crane Creek from 2004-2005. For every nutrient, Site 10c has a higher median concentration than any other site. ‘º’ represents extreme values while ‘*’ represents very extreme values.

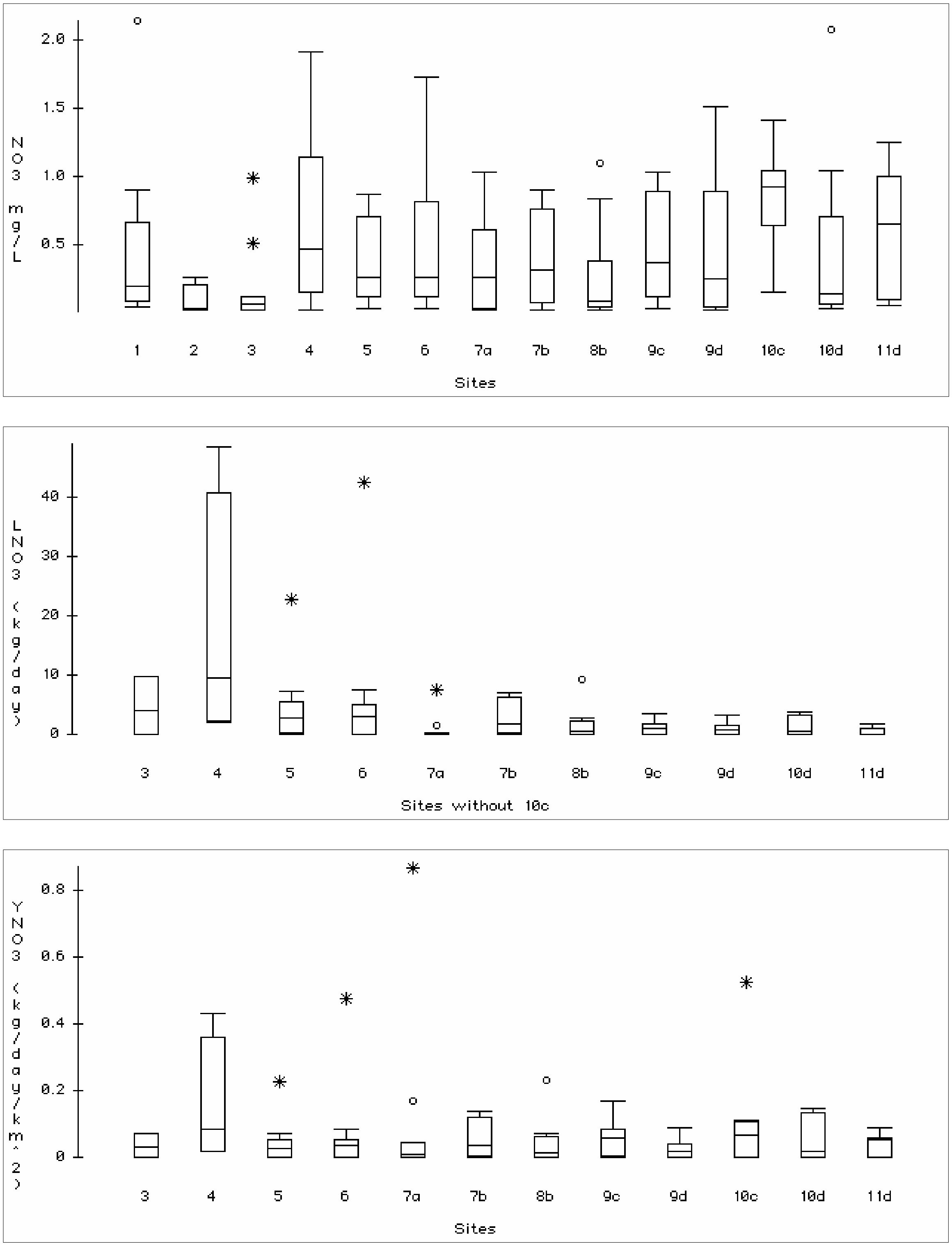


Figure 4: Upstream trends of nitrate concentration, loading, and yield on Crane Creek. Note the sharp decline in values downstream from Elliston Road (Site 4) for all three boxplots, possibly indicating a lake level effect. ‘º’ represents extreme values while ‘*’ represents very extreme values.

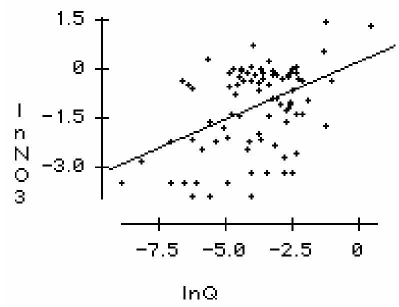
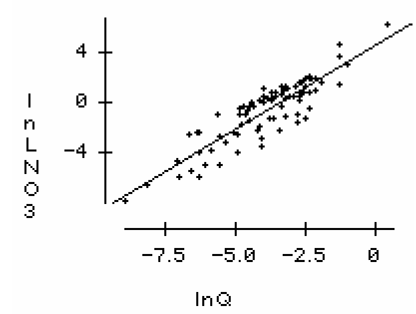
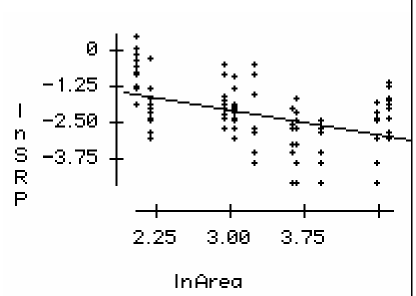
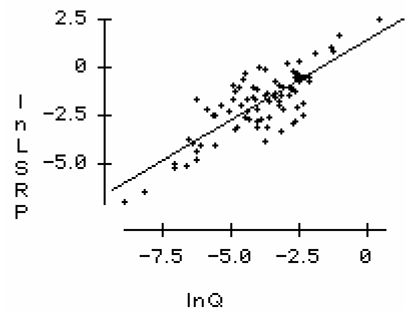
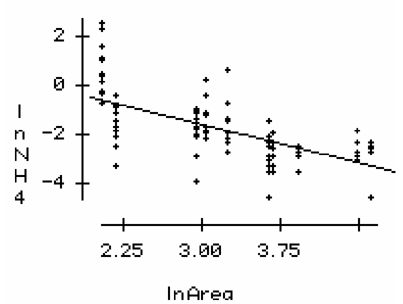
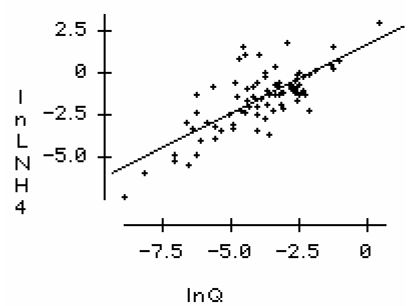
Nutrient Concentrations	Nutrient Loading
 $y=0.36x+0.23$ $R^2=0.19$	 $y=1.36x+4.70$ $R^2=0.78$
 $y=-0.58x-0.31$ $R^2=0.20$	 $y= 0.84x+1.45$ $R^2=0.66$
 $y=-1.06x+1.63$ $R^2=0.43$	 $y=0.83x+1.73$ $R^2=0.57$

Figure 5: Regression of nutrient concentrations and loadings with discharge or watershed area on Crane Creek upstream (Sites 5 to 11d). For nitrate concentration, discharge was used instead of watershed area because watershed area had a very poor fit. For nutrient concentration regression analysis, the significance level for all three nutrients was $p \leq 0.001$, standard error ranged from 0.08-0.12, and degrees of freedom ranged from 81-83. For nutrient loading regression analysis, the significance level of all three nutrients was $p \leq 0.001$, standard error ranged from 0.07-0.08, and degrees of freedom ranged from 81-83.

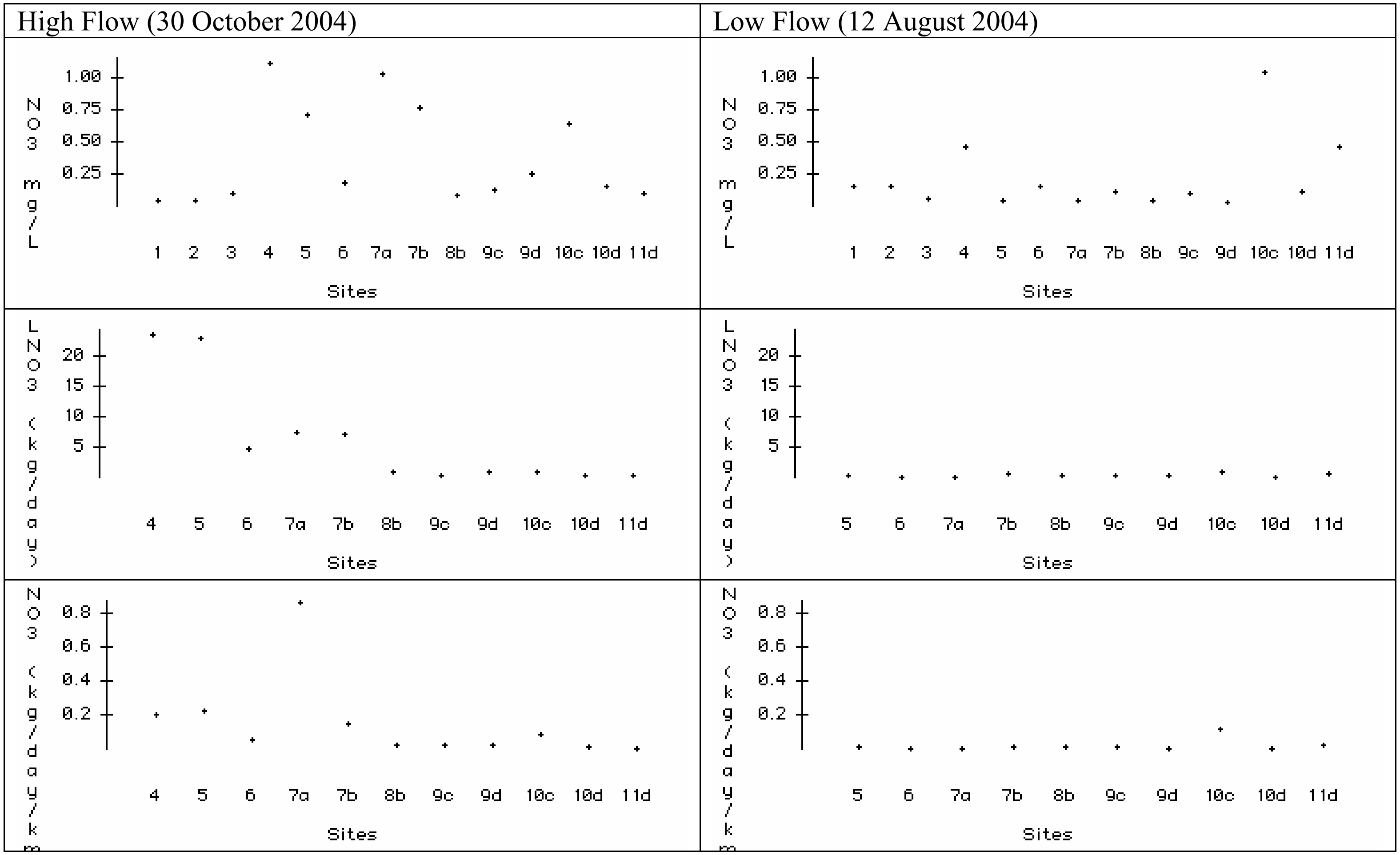


Figure 6: Comparison of nitrate concentrations, loadings, and yields for high and low discharge days on Crane Creek. Steam flow data was not available for sites 1-3. Site 4 is missing from the loading and yield on the low flow day because discharge was zero.

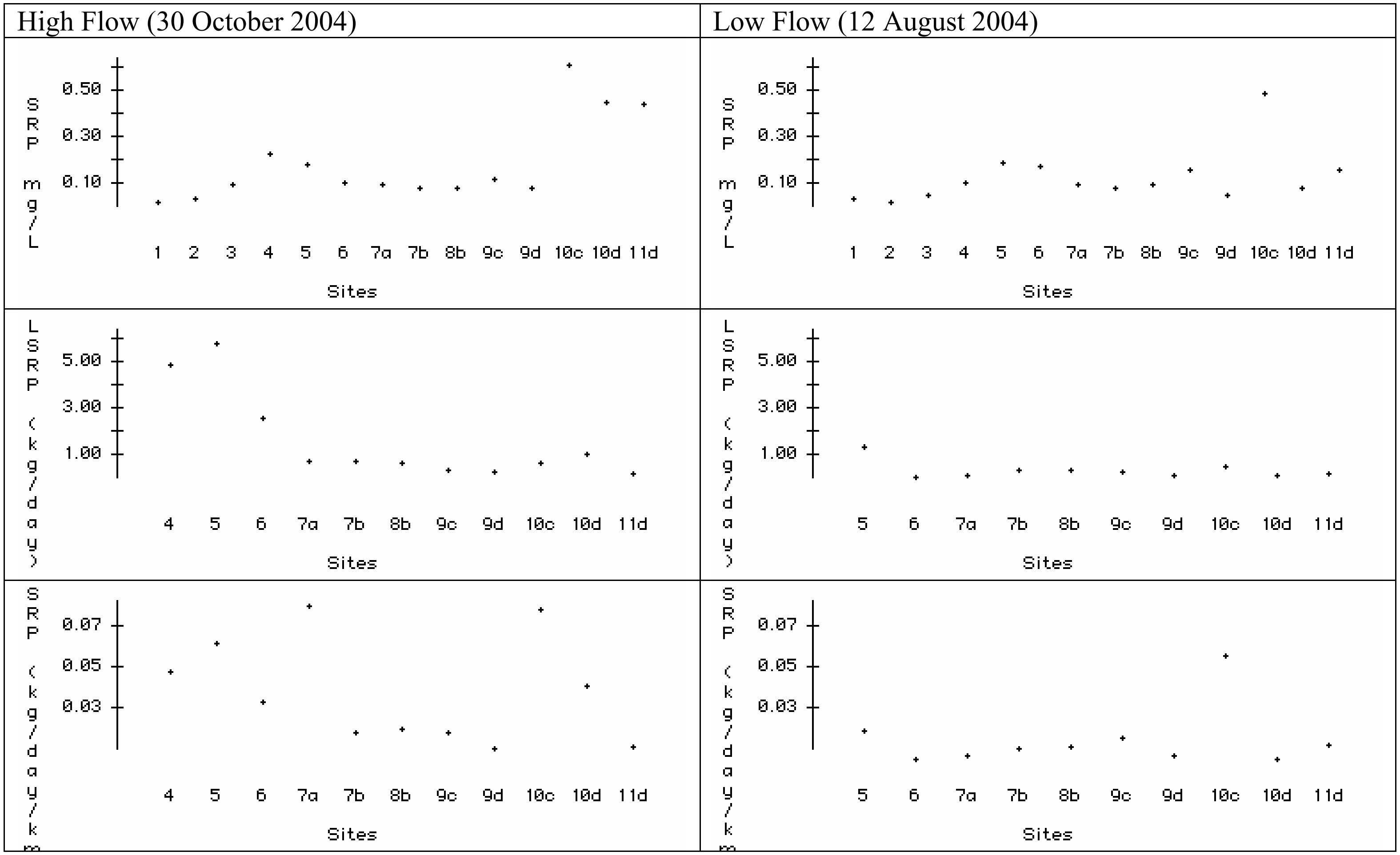


Figure 7: Comparison of SRP concentrations, loadings, and yields for high and low discharge days on Crane Creek. Stream flow data was not available on sites 1-3. Site 4 is missing from the loading and yield on the low flow day because discharge was zero.

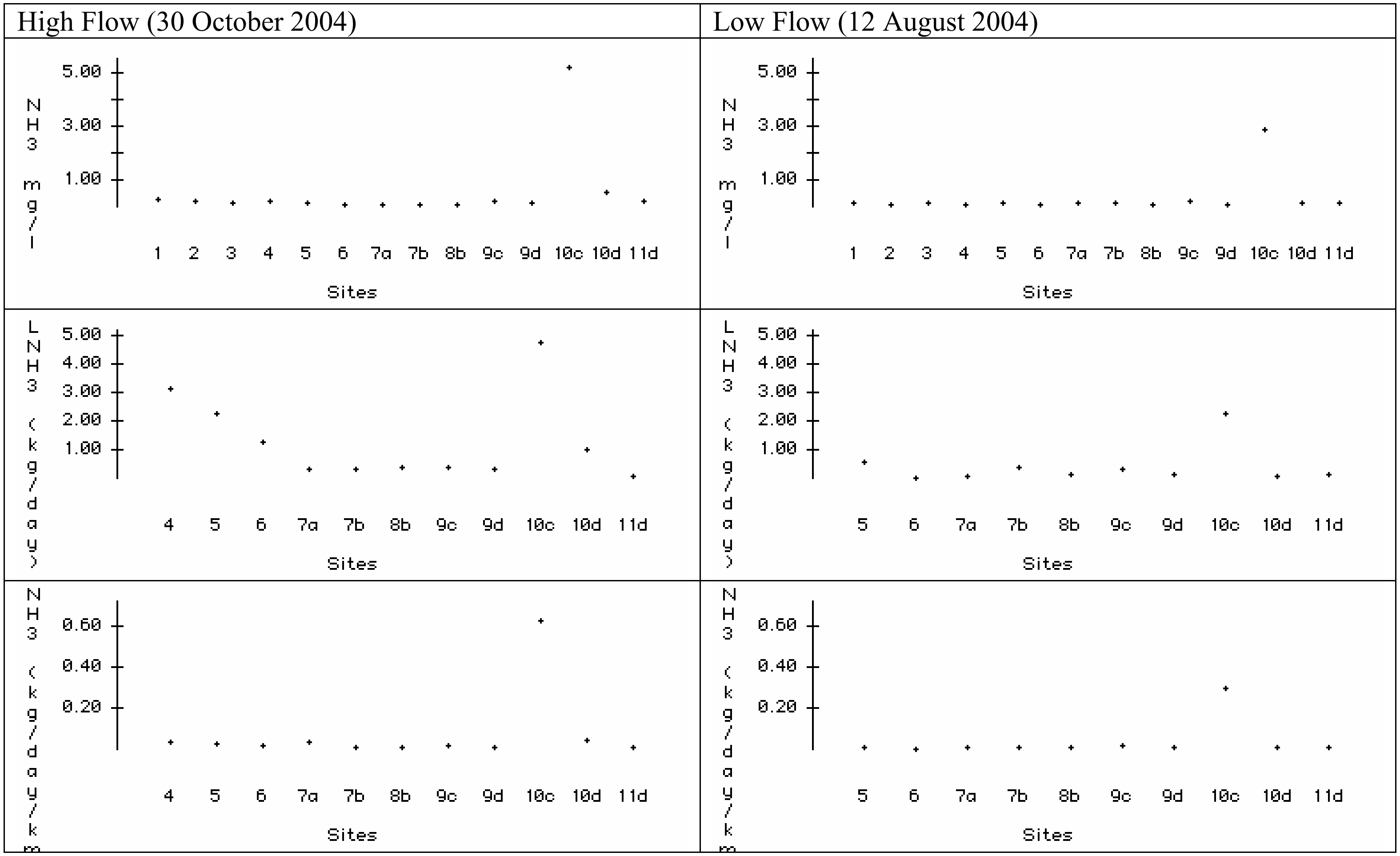


Figure 8: Comparison of ammonia concentrations, loadings, and yields for high and low discharge days on Crane Creek. Stream flow data was not available on sites 1-3. Site 4 is missing from the loading and yield on the low flow day because discharge was zero.

Table 1: Sites on Crane Creek, locations, and GPS Coordinates.

Site number	Site location	GPS Coordinates
1	Mouth of Crane Creek	41.65074 N, 83.19799 W
2	Crane Creek across from Pool 2A	41. 62128 N, 83.21477 W
3	Crane Creek at Stange Road	41.63191 N, 83.19792 W
4	Crane Creek at Elliston	41.60270 N, 83.37676 W
5	Crane Creek at Opfer Lentz	41.60270 N, 83.37676 W
6	Crane Creek at Reiman Road	41.60271 N, 83.37676 W
7a	Ayers Creek at Fostoria	41.59610 N, 83.41599 W
7b	Crane Creek at Billman	41.56707 N, 83.42502 W
8b	Crane Creek at Milbury	41.56708 N, 83.42502 W
9d	Crane Creek at Moline	41. 55855 N, 83.43113W
9c	Henry Creek at Moline Martin	41.55891 N, 83.44487 W
10c	Henry Creek at Broadway	41.51519 N, 83.46850 W
10d	Crane Creek at Warns	41.51519 N, 83.46850 W
11d	Crane Creek at Genoa	42.26271 N, 83.46840 W
A	Pool 2A	41.61981 N, 83. 21899 W
B	Pool 2B	41.62250 N, 83.21107 W
C	Nearshore Lake Erie at Metzger Marsh	41.65073 N, 83.23822 W

Table 2: Timeline for data collection on Crane Creek. Darkened boxes indicate that measurements were taken for that parameter during the indicated month.

Category	Parameter		May	June	July	Aug	Sept	Oct	Nov		April	May	June	July	Aug	Sept
Hydrologic																
	discharge		■	■	■	■	■	■	■		■	■	■			
Nutrient																
	SRP		■	■	■	■	■	■	■		■	■	■			
	TP												■			
	Ammonia		■	■	■	■	■	■	■		■	■	■			
	Nitrate		■	■	■	■	■	■	■		■	■	■			
Physical/																
Chemical																
	gilvin		■	■	■	■	■	■	■		■	■	■			
	temperature		■	■	■	■	■	■	■		■	■	■			
	conductivity		■	■	■	■	■	■	■		■	■	■			
	alkalinity		■	■	■	■	■	■	■		■	■	■			
	suspended sediment												■			
	suspended solids												■			
	FTU turbidity												■			
	COD (all sites)												■			
	COD (Elliston)		■	■	■	■	■	■	■		■	■	■			
	stream survey														■	■

Table 3: Physical and chemical water quality parameters of Crane Creek for the 2004-2005 sample period. Overall water quality for Crane Creek is in bold. Italicized are Crane Creek upstream and Crane Creek downstream, which are further broken down by site. The site on Lake Erie and the two diked pools, also in bold, are included for comparison.

		Water Temp (°C)		**Conductivity (µs cm^{-1})**		**Alkalinity (mg L^{-1})**		**Gilvin (abs cm^{-1})**	
Group	**Site**	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Lake Erie	**C**	**19.1**	**6.2-27.8**	**299**	**228-430**	**89**	**50-125**	**0.01**	**0.00-0.03**
Crane Creek		**19.4**	**1.3-32.8**	**738**	**106-2490**	**168**	**50-370**	**0.02**	**0.00-0.47**
down		*20.2*	*5.6-32.8*	*540*	*106-1004*	*134*	*70-216*	*0.03*	*0.00-0.47*
	1	19.4	5.6-25.8	334	230-505	94	70-126	0.01	0.00-0.03
	2	20	6-31.6	407	106-614	116	85-157	0.02	0.01-0.06
	3	19.9	5.6-32.8	630	359-948	155	72-199	0.02	0.01-0.04
	4	20.3	7.0-32.0	816	533-1004	181	106-216	0.02	0.01-0.47
up		*18.8*	*1.3-30.0*	*954*	*517-2490*	*201*	*105-370*	*0.03*	*0.01-0.3*
	5	17.7	5-26.5	869	762-1013	182	110-272	0.02	0.01-0.03
	6	18.7	4.6-26.2	871	608-1130	184	118-251	0.05	0.01-0.31
	7a	19.4	4.3-28.4	893	517-1104	204	119-304	0.02	0.01-0.03
	7b	17.9	3.8-26.0	878	766-1110	184	134-249	0.02	0.01-0.03
	8b	17.9	3.6-24.9	931	760-1282	194	151-242	0.02	0.01-0.02
	9c	18.5	3.3-26.0	1030	711-1665	207	121-265	0.02	0.01-0.03
	9d	21.3	3.4-30.0	817	643-975	184	105-276	0.02	0.01-0.03
	10c	20.0	1.3-29.0	1461	885-2490	258	177-370	0.02	0.01-0.03
	10d	18.7	3.0-26.8	938	744-1244	205	109-314	0.02	0.01-0.03
	11d	18.7	2.5-25.6	903	655-1227	216	161-294	0.02	0.01-0.04
Pool 2A	**A**	**20.6**	**9.3-29.2**	**349**	**200-627**	**120**	**65-195**	**0.01**	**0.01-0.02**
Pool 2B	**B**	**19.3**	**6.4-32.8**	**290**	**209-340**	**114**	**65-140**	**0.01**	**0.01-0.02**

Table 4: Nutrient concentrations in Crane Creek for the 2004-2005 sample period. Overall nutrient concentrations for Crane Creek are in bold. Italicized are Crane Creek upstream and Crane Creek downstream, which are further broken down by site. The site on Lake Erie and the two diked pools, also in bold, are included for comparison.

		N03-N (mg L^{-1})		SRP (mg L^{-1})		NH4-N (mg L^{-1})	
Group	**Site**	Mean	Range	Mean	Range	Mean	Range
Near-shore Lake Erie	**C**	**0.44**	**0.05-1.57**	**0.02**	**0.01-0.03**	**0.04**	**0.01-0.13**
Crane Creek		**0.49**	**0.01-4.26**	**0.12**	**0.00-1.70**	**0.34**	**0.00-13.20**
down		*0.51*	*0.02-3.19*	*0.05*	*0.01-0.23*	*0.08*	*0.00-0.55*
	1	0.47	0.04-2.15	0.02	0.01-0.06	0.06	0.00-0.25
	2	0.09	0.02-0.26	0.02	0.01-0.03	0.08	0.00-0.20
	3	0.18	0.02-1.00	0.05	0.01-0.17	0.08	0.00-0.20
	4	0.93	0.02-3.19	0.10	0.02-0.23	0.13	0.01-0.55
up		*0.56*	*0.02-4.26*	*0.19*	*0.01-1.70*	*0.55*	*0.00-13.20*
	5	0.37	0.03-0.87	0.17	0.05-0.34	0.07	0.01-0.10
	6	0.49	0.03-1.74	0.08	0.01-0.17	0.07	0.00-0.16
	7a	0.37	0.02-1.04	0.18	0.05-0.76	0.25	0.00-0.70
	7b	0.38	0.020.91	0.05	0.01-0.09	0.05	0.03-0.09
	8b	0.31	0.02-1.11	0.08	0.01-0.20	0.07	0.03-0.15
	9c	0.47	0.03-1.04	0.17	0.05-0.42	0.34	0.12-1.30
	9d	0.49	0.02-1.52	0.05	0.01-0.14	0.08	0.01-0.24
	10c	0.88	0.15-1.41	0.65	0.16-1.70	3.76	0.50-13.2
	10d	0.53	0.03-2.08	0.18	0.02-0.62	0.39	0.07-1.95
	11d	0.59	0.05-1.26	0.26	0.07-0.62	0.21	0.02-0.39
Pool 2A	**A**	**0.04**	**0.01-0.12**	**0.02**	**0.00-0.06**	**0.09**	**0.00-0.22**
Pool 2B	**B**	**0.02**	**0.01-0.02**	**0.01**	**0.00-0.01**	**0.02**	**0.00-0.03**

Table 5: Results from ANOVA and post-Scheffe tests for differences between Crane Creek upstream (CC up), Crane Creek downstream (CC down), Pool 2A, Pool 2B, and Lake Erie. Italicized values indicate strong but not significant differences (p-value of above 0.05 but less than 0.20)

Parameter	Group p-value (df = 4)	Pair differences	Differences p-values
Conductivity	≤0.001	CC up vs. CC down CC up vs. Lake Erie CC up vs. Pool 2A CC up vs. Pool 2B CC down vs. Pool 2A CC down vs. Pool 2B CC down vs. Lake Erie	≤0.001 ≤0.001 ≤0.001 ≤0.001 ≤0.05 ≤0.05 ≤0.01
Alkalinity	≤0.001	CC up vs. CC down CC up vs. Lake Erie CC up vs. Pool 2A CC up vs. Pool 2B *CC down vs. Lake Erie*	≤0.001 ≤0.001 ≤0.001 ≤0.001 *0.061*
Gilvin	0.647	None	
NO3-N	*0.053*	None	
SRP	≤0.001	CC up vs. CC down CC up vs. Lake Erie CC up vs. Pool 2A *CC up vs. Pool 2B*	≤0.001 ≤0.05 ≤0.05 *0.070*
NH4-N	*0.0238*	*CC up vs. CC down*	*0.188*

Table 6: Results from ANCOVA and post-Scheffe tests for differences between Crane Creek upstream (CC up), Crane Creek downstream (CC down), Henry Creek, and Ayers Creek. Italicized values indicate strong but not significant differences (p-value of above 0.05 but less than 0.20)

Parameter	Trib p-value (df = 3)	Q p-value (df = 1)	Pair differences	Differences p-value
Conductivity	≤0.001	0.001	Henry vs Ayers Henry vs CC up Henry vs. CC down	≤0.001 ≤0.001 ≤0.001
Alkalinity	0.026	0.059	*Henry vs. Ayers* *Henry vs. CC up* *Henry vs. CC down*	*0.080* *0.071* *0.172*
Gilvin	0.930	0.012	None	
NO3-N	0.122	≤0.001	None	
SRP	≤0.001	0.035	Henry vs. Ayers Henry vs. CC up *Henry vs. CC down*	0.005 ≤0.001 *0.060*
NH4-N	≤0.001	0.454	Henry vs. Ayers Henry vs. CC up Henry vs. CC down	≤0.001 ≤0.001 0.002
LNO3-N	0.887	0.003	None	
LSRP	0.471	≤0.001	None	
LNH4-N	0.039	≤0.001	CC up vs. CC down	0.071
YNO3-N	0.009	≤0.001	Ayers vs. CC up Ayers vs. CC down *Ayers vs. Henry*	0.014 0.041 *0.126*
YSRP	≤0.001	≤0.001	Ayers vs. CC up Ayers vs. CC down Henry vs. CC up *Henry vs. CC down*	0.002 0.013 0.010 *0.052*
YNH4-N	≤0.001	0.006	*Ayers vs. CC up* Henry vs. CC up Henry vs. CC down	*0.173* ≤0.001 0.015

Table 7: Results of ANOVA test of seasonal differences in water quality parameters in Crane Creek upstream and Crane Creek downstream. '*' indicates strong seasonal differences of the parameters ($0.05 \leq p \leq 0.20$) and '**' indicates significant ($p \leq 0.05$) differences between seasons.

Parameter	Crane Creek upstream p-value	Crane Creek downstream p-value	Which has more significant seasonal differences?
Conductivity	≤0.001 (df = 81)**	0.219 (df = 39)	Upstream
Alkalinity	≤0.001 (df = 81)**	0.175 (df = 38)*	Upstream
Gilvin	0.379 (df = 81)	≤0.001 (df = 39)**	Downstream
NO3-N	0.157 (df = 81)*	0.730 (df = 39)	Upstream
SRP	0.005 (df = 81)**	0.235 (df = 39)	Upstream
NH4-N	0.397 (df = 81)	0.089 (df = 39)*	Downstream

Table 8: Summary of multiple linear regression analysis of nutrient concentrations and %agricultural land, %urban land, stream discharge, and watershed area. '*' indicates $p \leq 0.05$, '**' indicates $p \leq 0.01$, and '***'indicates $p \leq 0.001$.

Nutrient	R^2	ln %urban	ln %ag	ln Discharge	ln Watershed Area
ln (NO3-N) (df=93)	0.28	-0.73	-1.02	0.46***	-0.66***
ln (SRP) (df=93)	0.29	-2.74***	-11.77**	-0.09	≤0.01
ln (NH4-N) (df=91)	0.35	-1.74*	-4.68	0.09	-0.60***

Table 9: Comparison of nutrient concentration in Crane Creek to concentrations in other Lake Erie tributaries. Comparison data is taken from the online database of the Heidelberg College Water Quality Laboratory for 1 May 2003 to 30 June 2004. Since comparison data was taken at sites close to the lake but outside of flow reversals, Crane Creek at Opfer Lentz (Site 5) was chosen for comparison. Ammonia nitrogen measurements not included because comparison values were not available.

Watershed	Drainage Area (above sampling station) km^2	NO3-N mean (mg L^{-1})	NO3-N range (mg L^{-1})	SRP mean (mg L^{-1})	SRP range (mg L^{-1})
Crane Creek (Site 5)	**101**	**0.37**	**0.03-0.87**	**0.17**	**0.05-0.34**
River Raisin	2699	4.74	0.04-13.46	0.03	ND-0.23
Vermilion	679	1.91	ND**-7.98	0.03	ND-0.41
Maumee	16395	4.62	ND-14.89	0.08	ND-0.23
Honey Creek*	386	5.13	1.38-17.01	0.07	ND-1.07
Sandusky	3245	5.05	ND-16.42	0.07	ND-0.35
Cuyahoga	1834	1.67	0.40-4.20	0.05	ND-0.22
Grand	1647	0.37	ND-1.22	0.02	ND-0.07

*Honey Creek is part of the Sandusky watershed
** ND= not detected

Table 10: Comparison of Crane Creek (CC) nutrient concentrations to nutrient concentrations in Old Woman Creek (OWC) at specific locations. Data for Old Woman Creek taken from Krieger 2003. Crane Creek at Opfer Lentz (Site 5) was used for “CC before lake effect” and Crane Creek across from Pool 2A (Site 2) was used for “CC mouth.” The nutrient concentrations from the mouth of Crane Creek (Site 1) was not used because the bidirectional flow can drastically change nutrient concentration in a twelve-hour period. Dissolved inorganic nitrogen (DIN) is nitrite-nitrate nitrogen and ammonia nitrogen combined. All nutrient concentrations are reported in mg L^{-1}.

	CC before lake effect (Site 5)	OWC before lake effect	CC mouth (Site 2)	OWC mouth
Watershed area (km^2)	112	57	145.65	68.9
NO3-N mean	0.37	3.94	0.09	0.96
NO3-N range	0.03-0.87	ND*-65	0.02-0.26	ND-17.2
SRP mean	0.17	0.01	0.02	0.01
SRP range	0.05-0.34	ND-0.25	0.01-0.03	ND-0.12
NH4-N mean	0.07	0.07	0.08	0.15
NH4-N range	0.01-0.10	ND-14.60	ND-0.20	ND-1.33
DIN mean	0.44	4.01	0.17	1.11
DIN range	0.04-0.97	ND-76.60	0.02-0.46	ND-18.53

* ND= not detected

Table 11: Nutrient concentrations on the upstream Crane Creek sites (Sites 5 to 11d) over time. Historical data was provided by the Ohio EPA in spreadsheet form and in two water quality reports. All nutrient concentrations are reported in mg L^{-1}.

Parameter	1980-1990	1991-2000	Present (2004-2005)
Nitrate mean	1.78	0.48	0.56
Nitrate range	ND*-4.04	ND-1.08	ND-4.26
TP mean	1.55	0.70	0.24**
TP range	0.12-6.2	0.16-3.79	0.05-1.81
Ammonia mean	2.75	2.16	0.55
Ammonia range	0.10-15	ND- 7.53	ND-13.2

* ND= not detected

** TP calculated from SRP values based on linear regression. Probably an underestimate.

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