EFFECTS OF DAM REMOVAL ON WATER QUALITY VARIABLES

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ABSTRACT

A monitoring study was conducted on the Sandusky River, in north-central Ohio, to characterize changes in water quality and quantity before and after the removal of a low-head dam (St. John). Short-term time series of flow and turbidity were taken during the dewatering and the removal of the dam in order to determine sediment loads. High resolution spatial data, based on vertical profiles and longitudinal surveys, were collected before and after the removal to determine changes in temperature, dissolved oxygen (DO), pH, turbidity, specific conductivity, and oxidation-reduction potential (ORP) in the 18 km reservoir. Surface water samples were collected upstream and downstream of the dam before and after removal and analyzed for nitrate, ammonia, and phosphate. After dam removal, denitrification potential was studied within the former reservoir to compare the potential in newly exposed mudflats with the riverbed and the floodplain.

During the breach of the dam, change in depth and turbidity was measured at a sampling station 200 m downstream. A small increase in volumetric flow was measured, however, time series data taken at the same sampling station during the removal showed a larger increase in flow. The discharge from the breach was not discernable at a gaging station 53 km downstream, however, the larger discharge from the removal was evident, though clearly attenuated. The removal also caused an increase in turbidity less than that
of a bankfull storm event that occurred less than one week later, however, the duration of increased turbidity during removal was shorter than the storm event.

The vertical profile of temperature and DO measured behind the dam before removal showed a decrease in temperature and DO with increasing depth from the surface. This was similar to lake stratification, not river conditions. The longitudinal surveys of temperature and DO were determined to be more variable before dam removal and more variable in the 13.7 km reservoir than in the control reach upstream of the reservoir. Longitudinal surveys of specific conductivity indicated no spatial trends. Values mimicked historical trends (1970-1973) at the St. John Dam by increasing with low discharge and decreasing with high discharge. Before and after dam removal, longitudinal surveys of turbidity varied throughout the reservoir and were affected mostly by discharge and storm events. ORP and pH were similar upstream and downstream, before and after removal, suggesting no significant differences.

Phosphate and ammonia were not detected above 0.5 mg L\(^{-1}\) upstream or downstream of the dam before or after removal. Before dam removal, nitrate concentrations upstream from the reservoir were statistically different than concentrations in the reservoir. However, this was also true of the former reservoir after the dam was removed, concluding dam removal had no affect on nitrate distribution in the Sandusky River system.
Denitrification potential was measured in the reservoir after removal. Data showed there was no significant difference in removal potential longitudinally or laterally in the river. There was, however, a significant increase in potential when samples were treated with nitrate (increased from 5 to 100 mg N\textsubscript{2}O g\textsuperscript{-1} soil h\textsuperscript{-1}). This indicated that the river system was nitrate limited. Furthermore, studies also determined that the loss of sediment-water (denitrification potential) contact from dewatering the reservoir was insignificant. Even though sediment-water contact was lost, there was still enough sediment-water contact left to adequately remove nitrate from the system.

The dam breach did not significantly affect discharge or turbidity, however, dam removal increased turbidity and discharge similar to a small storm event. Dam removal did not significantly affect longitudinal values of nutrients, pH, ORP, turbidity, or specific conductivity, however, it did decrease the variability of temperature and DO in the former reservoir. Also, dam removal did not significantly affect the river system’s ability to remove nitrate.
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CHAPTER 1

INTRODUCTION

Dam removal is a new and exciting restoration tool being utilized more and more as an alternative to dam maintenance. Over 2.5 million dams dot the landscape throughout the United States including over 2 million low-head (usually smaller than 5 meters high) dams and an estimated 77,000 large dams (The Aspen Institute, 2002) documented by the U.S. Army Corps of Engineers National Inventory of Dams. Fifty-eight percent of U.S. dams are privately owned with 36% used for recreation (The Heinz Center, 2002). Dams are distributed all across the United States with the greatest number in Texas and the oldest in Connecticut (1677). Dams are one of the largest anthropogenic structures affecting river water quality (Graf, 1999) and aquatic organisms. Approximately one-third of all fish, two-thirds of all crayfish, and three-quarters of all freshwater mussels are threatened with extinction as a result of dams blocking 600,000 miles of free flowing river (Babbitt, 1999). The creation of dams has been historic and taken place for various reasons including: recreation, flood control, irrigation, water supply, and transportation. Dam removal is much more recent and has taken place for many reasons including: safety, water quality, watershed restoration, and no longer financially profitable. Thirty percent of high-head dams are passed their life expectancy
(50 years) and by 2020, 80% of the dams will be outdated (River Alliance of Wisconsin & Trout Unlimited, 2000). Over 500 low-head and high-head dams have been removed across the United States in the past century with Wisconsin, California, Pennsylvania, and Ohio leading the way (River Alliance of Wisconsin and Trout Unlimited, 2000). To complement the ecological benefits, dam removal also has economic benefits. Dams typically cost three to five times less to remove than they do to repair (River Alliance of Wisconsin, 2004).

Of the dams removed, little documentation exists on post-removal effects. Future studies are needed to help develop a greater knowledge of how rivers are affected by various dam removal processes. There is a general four-step process to low-head dam removal that includes pre-planning of the removal and removal process, drawdown or draining of the impoundment, removal of the structure, and post-removal environmental restoration (River Alliance of Wisconsin and Trout Unlimited, 2000), however, every dam has its own unique characteristics so pre- and post-removal investigations should be modified to fit each situation.

Dams alter the hydraulic regime and geomorphology of a river and have a significant affect on habitat and water quality. Water quality effects in dammed reservoirs include increased temperature, oxygen depletion, deposition of contaminant laced sediments, disruption of the biogeochemical cycles, and supersaturation of gases in the reservoir (The Heinz Center, 2002). Decreased mobility of gravel and cobble (i.e. bedload) and a decrease of total suspended solids may lead to changes in substrate distributions and channel morphology downstream, and thus alter habitat along the river continuum (Gosselink et al., 1974; Kondolf, 1997). This, in turn, may alter the diversity
and population structure of organisms in river systems. Dams may also impact areas of fish spawning and degrade delta wetlands by retaining sediments behind the dam (Babbitt, 1999). Water quality variables may vary over time and space above and below dams as environmental conditions fluctuate and aquatic habitats change (Stanley and Doyle, 2002). The longitudinal profile of the river bed behind the dam will change dramatically as water levels increase and sediments are deposited.

The thesis is divided into three separate articles, chapters 2, 3, and 4, each reflecting the effects of dam removal on water quality and flow. The second chapter discusses the impact of the dam breach and removal on turbidity and flow. The third chapter evaluates spatial distributions of several water quality variables in the St. John reservoir. The fourth chapter looks at denitrification potential as a means of reducing nitrate concentrations as a result of natural design or restoration.
CHAPTER 2

TURBIDITY AND DISCHARGE EFFECTS FROM THE BREACH AND DECONSTRUCTION OF A LOW-HEAD DAM

2.1 Abstract

In the spring of 2003, the 46 m wide, 2.2 m high St. John Dam on the Sandusky River, Ohio, was breached to reduce the water level in the reservoir. In the autumn of the same year, the dam was removed in an effort to restore aquatic habitat and health of the riparian zone. During both the breach and the removal, high resolution time series data of water depth and turbidity were monitored 200 m downstream of the dam. Though discharge and turbidity increased during the breach, the values were minor compared to those during the removal. Release of the impounded water resulted in a peak turbidity of 113 NTU and a peak discharge of 46 m³ s⁻¹ in the first 2 hours of the removal. Turbidity and flow returned to background levels within 7 hours of the removal. While these levels of turbidity and discharge were comparable to a subsequent flow event, the transport of suspended sediment was ten fold greater for the storm event. A USGS gage, 53 km downstream of the dam, captured the passage of the flood wave that corresponded to the dam removal. This flow was attenuated by 63% over a 30 hour period, illustrating the diffusive nature of the dam release.
2.2 Introduction

Dam removal has increasingly become a means of restoring habitat and natural flows to riverine systems. Dewatering of reservoirs behind dams, either by breaching (partial removal to decrease the volume of the reservoir) of a dam or low level outlets, is a key step in removal of the dam. Usually this is a controlled deconstruction, which is quite different from the unpredictability of dam breaches that occur during natural disasters.

Dam removals have been given a great deal of attention in a variety of reports (e.g. American Rivers et al., 1999; American Rivers and Trout Unlimited, 2002; The Heinz Center, 2002; Landers, 2004) and trade journals (Bioscience, 2002; Francisco, 2004). Few studies have monitored the hydraulics of pre- to post-dam removals to quantify the downstream effects of the breach and removal on the peak discharge and sediment transport. This chapter will describe the flow and suspended sediment discharge during the breach and deconstruction of the dam.

2.3 Methods

2.3.1 Site Description

The St. John Dam was a low-head dam located in the Sandusky River Basin (Figure 2.1), a coastal watershed in north central Ohio with a hydrologic unit code (HUC) of 0400011. The Sandusky River flows north into Lake Erie at Sandusky Bay and has been designated a State Scenic River for half of its 208 km length because it has a natural riparian zone with low development. The St. John dam was a concrete dam, 46 m wide and 2.2 m high, that was anchored in a bedrock (limestone) substrate. The dam was
located in the upper watershed of Sandusky River at river mile 50, south of Tiffin, Ohio (Figure 2.1). The reservoir extended approximately 13.7 km upstream with a longitudinal bed slope of 0.00012. During base flow conditions the total reservoir area was approximately 0.59 km² with a storage of $5.6 \times 10^5$ m³. Sediments accumulated in the reservoir were composed of fine gravels, sands, clays, and silts, however, estimates of the amount of impounded sediment did not exist. The dam was built in the middle 1930's and functioned as a water supply reservoir for the city of Tiffin, Ohio. Because the dam was obsolete, the Scenic Rivers Program, Ohio Department of Natural Resources (ODNR), approached the dam owner to initiate removal of the aging structure.

2.3.2 Gage Description

Two real time gage stations operated by the U.S. Geological Survey on the Sandusky River were used to monitor discharge upstream and downstream of the dam. The upstream gage station (USGS 04196500) at Upper Sandusky, Ohio, is located 46.5 km from the dam at river mile 79. The downstream gage station (USGS 04198000), near Fremont, Ohio, is 53 km from the dam at river mile 17. Discharge at the upstream gage was a nearly continuous period of record from 1921-current, while the downstream gage was a nearly continuous period of record from 1923-current. Both gage stations calculated discharge and gage height every 15 minutes. All data were grouped to 30 minute averages.

2.3.3 Instrumentation and Analysis

A fixed elevation 6920 YSI water quality sonde (Yellow Spring Instruments, Inc.) was deployed at a sampling site 200 m downstream of the dam during breach and removal. The sonde was equipped with probes to measure temperature, conductivity, pH,
oxidation-reduction potential, and dissolved oxygen. In addition to water quality probes, the sonde was equipped with a 6026 turbidity probe to measure turbidity of suspended sediments and a pressure sensor to record the relative depth (i.e. water level). Before deployments of the sonde, turbidity was calibrated (in NTU) against a laboratory instrument and standards. A 3-point standard method was used in March and a 1-point method in November (The one point method was used since one of the three standards was unreliable). The sonde was placed in a 15.2 cm diameter PVC tube that was open at the bottom. The sonde rested on a bolt placed through the lower end of the tube. The tube was clamped to a galvanized steel pole (5.08 cm diameter, 2.5 meters long) that was driven into the substrate and the top of the tube was capped with a locking mechanism (Fondriest Environmental, Inc.).

Time series data of turbidity and depth were collected every 5 minutes during the breach (from March 17, 2003 until March 24, 2003). During the removal (from November 15, 2003 until November 21, 2003) the sensor was redeployed but sampled less frequently (every 10 minutes) to conserve battery power. Sensor depth was converted to total depth by adding the height from the pressure sensor to the bed, noting that no deposition or erosion occurred at the sampling station. Total depth at the sampling station was calibrated to flow by interpolating from a flow-depth (i.e. Q-H) rating curve for data from the Upper Sandusky and Fremont gages taken during the period centered on the breach and dam removal and weighted by watershed area. It should be noted that the flow passing the sampling station represents 75% of the watershed monitored by the downstream gage. Flow and turbidity values were averaged to 30 minute points to coincide with gage station data. Historical data on sediment
concentration at the downstream gage (2002-current) were obtained from the Water Quality Laboratory, Heidelberg College. Pre- versus post-dam removal sediment loading was statistically compared by testing the difference in slope and elevation of trends in loading versus discharge (Zarr, 1984).

2.4 Results

2.4.1 Dam Breach (March 18, 2003)

Deconstruction of the St. John Dam began at 9:00 a.m. on March 18, 2003, when the dam was breached with a notch on the west side to gradually reduce water levels in the reservoir (Figure 2.2A). The notch, approximately 4 m wide and 0.5 m high, was chipped-out of the dam by a track hoe, however, a breakdown of the track hoe interrupted work for several hours. By 5:00 p.m. the same day, the breach was completed leaving the dam with a notch approximately 8 m wide and 1 m high (Figure 2.2B). Prior to the breach on March 18, natural riverine discharge decreased from nearly 27 m$^3$ s$^{-1}$ to 18 m$^3$ s$^{-1}$ (Figure 2.3A). After the release of impounded water during the breach (signified by the arrows in Figure 2.3A, 2.3B), discharge leveled off at 18 m$^3$ s$^{-1}$ until 6:00 p.m., then continued to decrease. The nearly constant reduction in discharge was confirmed at the upstream gage station, which served as a control since it was located upstream of the reservoir (Figure 2.3B). As a result of the breach, the increase in discharge at the sampling station 200 m downstream was 4 m$^3$ s$^{-1}$. The volume of water released during the breach can be estimated from the hydrograph as

$$V_{res} = \int (Q_{total} - Q_{bg}) dt$$  \hspace{1cm} (1)
where the total flow, \( Q_{\text{total}} \), encompasses from 0 \( m^3 \) s\(^{-1}\) to the maximum flow during the breach. The background flow, \( Q_{\text{bg}} \), represents the background river flow, estimated from the trend from pre- to post-breach flows. Based on a total breach time of 9 hours (from 9:00 a.m. to 6 p.m.), the volume of reservoir discharged during the breach was \( V_{\text{Res}} = 1.69 \times 10^4 \) m\(^3\). The Froude number, based on a rectangular channel at the sampling station where the cross-section is width \( x \) depth \( (w \times d) \), is given as

\[
F = \frac{u_p}{\sqrt{gd_p}}
\]  

(2)

where the peak velocity, \( u_p \), is given as \( Q_{\text{total}}/(w \times d_p) \), which was calculated from flow and depth measured 200 m downstream where the width was 20 m. For a measured peak depth of 1.1 m, the peak velocity was 0.86 m s\(^{-1}\) \( (27 \text{ m}^3 \text{ s}^{-1}/[20 \text{ m} \times 1.1 \text{ m}] \) and \( F = 0.26 \). This low Froude number is indicative of a mild, bed slope (i.e. \(<10^{-3}\)) at the sampling station. The discharge from the breach was not evident at the downstream gage. This was most likely a result of the wave disturbance being attenuated before reaching the gage. An increase in discharge from a storm event on March 21 was captured at both gages as well as at the sampling station. From March 18 to 21, 2003, turbidity at the sampling station decreased from 210 NTU to 65 NTU, then increased to 110 NTU by midnight on March 23, 2003. No increase in turbidity was evident during the breach at the 200 m downstream sampling location.

2.4.2 Dam Removal (November 17, 2003)

The deconstruction schedule originally stipulated the complete removal of the dam 1-2 weeks subsequent to the breach, but was postponed as a result of unusually high water levels in April, 2003. Deconstruction was further postponed during the fish
spawning season in late spring. In early summer, 2003, the elimination of the Civilian Conservation Corp (ODNR), as a result of State budget cuts, put the project in jeopardy since this group was operating the heavy machinery for the removal. Substitution of a private contractor was too expensive since the State had reduced the operating budget for the Scenic Rivers Program. In mid-summer, the state Department of Transportation offered financial assistance for fully removing the dam in exchange for mitigation credit for restoring the riparian zone. A bid process ensued, a new construction crew was hired and briefed, and the entire structure was removed from the river bed on November 17, 2003.

The removal began at 8:00 a.m. when two track hoes were positioned on bedrock below the dam and began removing a log jam and other debris on the east side of the dam crest. This operation caused the deteriorating concrete on the east bank to fail at approximately 8:30 a.m. Although the plan was to remove the dam over several hours, the structure was demolished in less than 2.5 hours (Figure 2.4A, 2.4B). On November 16, 2003, one day prior to the removal, discharge was decreasing at the 200 m downstream sampling site (Figure 2.5A) and at the upstream and downstream gage stations (Figure 2.5B). The collapse of the eastern dam structure produced an abrupt increase in discharge (signified by the arrows in Figure 2.5A, 2.5B). Within two hours, discharge peaked at 46 m$^3$ s$^{-1}$ then returned to pre-removal flow conditions after 7 hours (i.e. by 3:30 p.m the same day), releasing a volume of $V_{Rev} = 4.1 \times 10^5$ m$^3$. The peak in discharge at the sampling station was only the result of draining the reservoir, confirmed by the fact that the upstream gage showed no increase in flow during the duration of the removal, and no other discharges were evident between the sampling station and the
upstream gage. It appeared that flow was reduced at the upstream gage within 2 hours after the removal, possibly the result of a negative wave that propagated upstream. The Froude number for the peak discharge during removal was $F = 0.39$ (used for slope determination), based on a measured, peak depth of $d_p = 1.5 \text{ m}$ and a peak velocity of $u_p = 1.53 \text{ m s}^{-1}$, again indicative of the mild slope of the reservoir typography. Dividing $V_{Res}$ by the nominal cross-sectional area in the reservoir of $30 \text{ m}^2$ ($20 \text{ m} \times 1.5 \text{ m}$) gives the total distance of 13.7 km, which is close to the measured length of 12.8 km of reservoir during normal flow conditions.

The downstream gage recorded the passage of the reservoir discharge 9.5 hours after the initial release, however, the flow was highly attenuated (much different than the flood wave captured at the 200 m downstream sampling site). The peak discharge at the downstream gage was $17 \text{ m}^3 \text{ s}^{-1}$ recorded at midnight on November 18, 2003, some 15.5 hours after the removal. The tail of the distribution was recorded nearly 30 h after the initial removal. The attenuation of the flood wave illustrates that the disturbance in discharge (and water level) that resulted from the removal of the dam did not propagate as a surge but rather as a diffuse wave, which is expected given the low Froude number and mild channel slope (Julien, 2002).

Two days after the removal, discharge at the sampling station increased from $4 \text{ m}^3 \text{ s}^{-1}$, on November 19, 2003, to $53 \text{ m}^3 \text{ s}^{-1}$, on November 21, 2003, coinciding with runoff from an area wide storm with a local intensity of 2 cm in 24 hours. This runoff event was also captured by the upstream and downstream gages with time lags of roughly $-15$ hours for upstream and $+15$ hours for downstream gages, relative to the 200 m sampling station. Turbidity resulting from the suspended sediment load was directly related to
flow conditions at the sampling station. Turbidity reached a peak value of 114 NTU during the removal compared to 113 NTU during the subsequent storm event on November 21, 2003.

2.5 Conclusions

Breaching of the St. John dam in the spring of 2003 caused a minor increase in volumetric flow that was not evident 53 km downstream. In contrast, the removal of the dam in late autumn 2003, had flow and depth characteristics that were similar to the subsequent storm event on November 21, 2003. The attenuation of the flood wave for the removal is well predicted by the solution of the Saint Venant equation in the dam breach model of Ponce et al. (2003). Model results can be compared to the St. John data using a Froude number for dam breach (Ponce et al., 2003) calculated as

$$F_{db} = \frac{q_p}{\sqrt{gV_{res}}}$$  \hspace{1cm} (3)

where $q_p$ is the peak flow and $g$ is the gravitational acceleration. For the breach in March 2003, the volume of the reservoir drained ($V_{res} = 1.69 \times 10^4$ m$^3$) and the peak discharge per unit width of $q_p = 0.95$ m$^2$ s$^{-1}$ (19 m$^3$ s$^{-1}$/20 m) give $F_{db} = 2.3 \times 10^3$ (0.95 m$^2$ s$^{-1}$ x [9.8 m s$^{-2}$ x $1.69 \times 10^4$ m$^3$])$^{-0.5}$. Ponce et al. (2003) show that this value of $F_{db}$ on a mild slope ($10^4$) for a small volume release ($2.7 \times 10^3$ m$^3$) gives a dimensionless distance of $X/L_0 = 3 \times 10^{-2}$ (see Figure 4, Ponce et al., 2003) where $L_0 = d_0/S_0$ and $d_0$ is normal depth. For the downstream station, $L_0 = 2.5 \times 10^3$ m (0.3 m / $1.2 \times 10^4$) thus, $X = 75$ m ($3 \times 10^{-2}$ x $2.5 \times 10^3$ m), consequently, the flood wave had attenuated before the 200 m sampling station. In contrast, for the removal in November 2003, the volume of the reservoir drained
\( V_{\text{Res}} = 4.1 \times 10^5 \text{ m}^3 \) and the peak discharge per unit width of \( q_p = 2.3 \text{ m}^2 \text{ s}^{-1} (46 \text{ m}^3 \text{ s}^{-1/20m}) \) give \( F_{ds} = 1.1 \times 10^{-3} \) \( (2.3 \text{ m}^2 \text{ s}^{-1} /[9.8 \text{ m} \text{ s}^{-2} \times 4.1 \times 10^5 \text{ m}^3]^{0.5}) \). Based on this dam-break Froude number, a mild slope, and a larger volume release \( (1.08 \times 10^5 \text{ m}^3) \), \( X/L_0 = 1 \) and \( X = 2.5 \times 10^3 \text{ m} \). Thus, the flood wave during the breach attenuated after the sampling station but well before the downstream gage at Fremont and before the City of Tiffin. Further, given the ratio of peak to background discharges at the sampling station of \( q_p/q_r = 3.8 \) \( (46 \text{ m}^3 \text{ s}^{-1}/12 \text{ m}^3 \text{ s}^{-1}) \) and a dimensionless downstream distance of \( X/L_0 = 24 \) \( (5.9 \times 10^4 \text{ m} / 2.5 \times 10^3 \text{ m}) \) for the Fremont (i.e. downstream) gaging station, the percent of attenuation should have been 96.7% for a mild slope \( (10^{-4} \text{ in Figure 3c, Ponce et al. 2003}) \) and 41% for a moderate slope \( (10^{-3} \text{ in Figure 3c, Ponce et al. 2003}) \). The actual attenuation of 63% \( (100 \times [1 - 17 \text{ m}^3 \text{ s}^{-1}/46 \text{ m}^3 \text{ s}^{-1}]) \) reflects the greater slope of 0.0015 in the lower potion of the watershed.

Another important aspect of the dam removal was the transport of the reservoir sediment. The suspended sediment load can be calculated as the product of the flow \( (Q) \) and suspended sediment concentration \( (C_{ss}) \) given a relationship exists between suspended sediment and the turbidity. Based on independent grab samples, turbidity (in NTU) was linearly related to the suspended sediment by the regression equation \( C_{ss} \) \( (\text{mg L}^{-1}) = 0.439 \text{ NTU} + 5.3199 \) \( (n = 66, r^2 = 0.9996) \). Since the peaks in suspended sediments concentration \( (55 \text{ mg L}^{-1}) \) and water discharge \( (46 \text{ m}^3 \text{ s}^{-1}) \) were similar for both the removal and the subsequent storm event, the peak loadings were similar at 2.5 \text{ kg s}^{-1}. Because this storm occurred shortly after the dam removal, it is not a good comparison of turbidity for a typical storm since it is likely sediment once impounded by the dam was mobilized. To verify the effects of the removal on sediment transport, historic sediment
loading at the sampling station was calculated from flow and suspended sediment data from the downstream gaging station, weighted to the watershed area of the sampling station. When loading for storm events (i.e. $Q > 30 \text{ m}^3 \text{ s}^{-1}$) one year prior to the removal and 10 months after the removal were plotted versus discharge (Figure 2.6), no statistical differences were found in either the slopes ($\alpha(2) = 0.05$, df = 36, $t = 0.364$, $p = 0.921$) or the elevations ($\alpha(2) = 0.05$, df = 37, $t = 0.0814$, $p = 0.923$). Consequently, the overall trend in loading was best presented by a line of intermediate elevation (thin regression line, Figure 2.6). The scatter about the overall regression line was not explained by seasonal variability in flow ($\alpha(1) = 0.05$, df = 2, 35, $F = 1.37$, $p > 0.25$). The suspended sediment loading rate was low during the removal (grey square, Figure 2.6) compared to historical data for this watershed, and other watersheds globally (Eisma, 1994). Further, the loading derived from a 20 year median (grey circle, Figure 2.6; Richards et al., 2001) was within the error of the overall regression. Given the bankfull flow (140 $\text{ m}^3 \text{ s}^{-1}$) was similar to the 20 year median, the bankfull sediment loading rate was high compared with that of the removal. Thus, removal of the St. John Dam caused minor increases in channel storage and sediment transported downstream of the reservoir compared to a bankfull event. Finally, since the removal was a short lived event compared to the duration of large storms, less total mass was transported during the removal. As an example, for the storm discharge on November 21, 2003, the total mass of suspended sediment, determined as $L_{ss} = \int Q_{total} C_{ss} dt$, was a factor of 10 greater for the storm event ($2.6 \times 10^5 \text{ kg}$) compared to the removal ($2.7 \times 10^4 \text{ kg}$). These results, which are based on suspended load and wash load but not bed load, give a more quantitative assessment of
the hydraulics of dam removal, however, they may not be typical for all low-head dam removals if channel slope and sediment load differ greatly.

2.6 Notation

\( C_{SS} \) = suspended sediment concentration

\( d_0 \) = normal depth

\( d_p \) = peak depth

\( F \) = Froude number

\( F_{db} \) = Froude number for dam-breach and removal

\( g \) = gravitational acceleration

\( H \) = water level

\( L_0 \) = reference channel length

\( L_{SS} \) = suspended sediment load

\( q_p \) = unit peak discharge during the breach and removal

\( q_r \) = unit discharge, pre-breach and pre-removal

\( Q \) = volumetric discharge

\( Q_{total} \) = total volumetric discharge

\( Q_{bg} \) = volumetric discharge, pre-breach and pre-removal

\( S_0 \) = longitudinal bed slope

\( u_p \) = peak downstream velocity

\( V_{resv} \) = volume of the reservoir discharged

\( w \) = width of the channel

\( X \) = downstream distance from the dam
2.7 References


*Biosence*. 2002. Special section on dam removal and river restoration. 52(2).


Figure 2.1 - A map of the Sandusky River Watershed with an inset of the U.S.A. The locations of the U.S.G.S. gaging stations are indicated by filled squares and the location of the St. John Dam with a bar.
Figure 2.2 - Photographs of: (A) the St. John dam being breached at 8:30 h on March 18, 2003 and (B) the structure after the breach on April 24, 2003
Figure 2.3 - Time series of: (A) discharge and turbidity 200 m downstream of the dam and (B) gage station discharge in March 2003. Arrows indicated the incitation of the breach.
Figure 2.4 - Photographs of: (A) the removal at 8:30 h on November 17, 2003 and (B) a view of the same site 2 hours after the removal
Figure 2.5 - Time series: (A) discharge and turbidity 200 m downstream of the dam and (B) gage station discharge in November 2003. Arrows indicate the incitation of the removal.
Figure 2.6 - A plot of sediment loading rate versus discharge for storm events greater than 30 m$^3$/s for one year prior to dam removal (open squares) and 10 months after removal (solid squares). Also plotted is the 20 year median loading value (gray circle) and the loading for the removal of the St. John Dam.
CHAPTER 3

SPATIAL AND TEMPORAL TRENDS IN WATER QUALITY DATA BEFORE AND AFTER DAM REMOVAL

3.1 Abstract

Water quality and quantity parameters were measured pre- and post dam removal at the St. John Dam on the Sandusky River in north central Ohio. A vertical profile of dissolved oxygen (DO) and temperature in the dammed reservoir found lake-like characteristics with temperature and dissolved oxygen decreasing over depth prior to dam removal. After removal, lake-like stratification was eliminated. No detectable levels of ammonia (<0.5 mg/L) or phosphate (<0.1 mg/L) were found in the waters above or below the dam for the pre- or post-removal periods. Nitrate distributions above and below the dam were unaffected by removal but were found to be a function of seasonal discharge. There were no statistically significant differences in turbidity, pH, oxidation-reduction potential (ORP), or specific conductivity before or after dam removal. Variability of turbidity values within the reservoir appeared to be a result of fluctuating discharge. Surveys of DO and temperature in the surface waters above the dam did show longitudinal differences. Variability within the reservoir before the removal was larger than in the control reach upstream of the reservoir. Longitudinal variability of both DO and temperature in the reservoir also decreased after the dam was removed.
3.2 Introduction

3.2.1 Background

Dams create reservoirs with slower moving waters, higher temperatures, and lower DO concentrations. Nutrients may remain trapped behind dams, turbidity may increase as a result of algal blooms, and ORP may change. One study determined that ORP may reach as high as 500 mV in oxygenated lakes (Bostrom et al., 1982), while another study concluded that ORP may drop as low as -200 mV in the reducing zone of the substrate (Moore and Reddy, 1994). Studies performed on streams in Tennessee indicated aquatic ecosystems need multiple chemical parameters analyzed to correctly assess in-stream dynamics of aquatic systems during and after recovery of disturbances (Adams et al., 2002). Such parameters may include pH, nutrients, DO, ORP, and turbidity. Nutrient cycling plays an important role in aquatic ecosystems. Dams create reservoirs which serve as sinks for sediments and nutrients adsorbed to the sediments. The most significant source of nutrients to reservoirs is direct runoff (e.g. overland flow), drainage sources (e.g. culverts, drainage ditches, groundwater inflow, and drainage tiles), and small stream inputs (Alam-Imteaz et al., 2003). Removing a dam allows temperatures to decrease and DO levels to increase allowing native fish populations to thrive. Once dams are removed, reservoirs may serve as transient or long term sources of nutrients to downstream reaches. In river systems with high nitrogen concentrations, nitrate (NO$_3^-$) may account for over 50% of the total nitrogen (Hedin et al., 1995; Goolsby et al., 1999). Nitrate received from drainage sources may have median concentration values of 12 to 27 mg L$^{-1}$ (Fausey et al., 1995). Total Kjeldahl Nitrogen (TKN) (ammonia and organic nitrogen) makes up most of the remaining nitrogen with
organic nitrogen usually being higher than ammonia concentrations (Richards et al., 2001).

Ft. Edwards Dam was removed on the Hudson River, New York, in the summer of 2000. Sediments behind the dam were laced with PCBs. As the dam was removed, sediments moved downstream and transported the PCBs with them (Shuman, 1995). Sediments were removed downstream of the dam two months later (Chatterjee, 1997). The removal of the PCBs resulted in increased water quality in the former impoundment and downstream, increasing the number of benthic macroinvertebrates by 50 fold. Even though the removal was not science based, Ft. Edwards has set a precedent for dam removal as a river restoration tool. The dam became the first of its kind to be deconstructed against the wishes of its owner by orders given by The Federal Energy Regulatory Commission, FERC (American Rivers, 2002).

Dams and dam removals have had a significant effect on water quality variables, which in turn affect the biotic community. Recovery of ecosystems can be determined using characteristics of aquatic organisms such as biomass, abundance, diversity, richness, or a combination of biological indices (Ford, 1989), however, more than one trophic level should be analyzed since some organisms take longer to return to their natural state than others (Depledge, 1999). Studies performed during the removal of Hellberg’s Dam in Lancaster County, Pennsylvania, showed an increase in benthic macroinvertebrate taxa from 16 families to 30 families after the dam was removed. The dominant families increased from pollution tolerant families to pollution sensitive families, indicating an increase in water quality (Calaman and Ferreri, 2004). Studies performed before Burgomillodo and Granby dams were removed determined benthic
macroinvertebrate richness increased downstream of the dams as diversity decreased (Camargo and Voelz, 1998). The temperature changes downstream from the Granby Dam may have led to taxanomic changes of the macroinvertebrates and spatial concentrations of oxygen, while short-term flow fluctuations may have caused the macroinvertebrate responses below the Burgomillodo Dam. It was determined that flow regulation alters turbidity, water temperature, and other variables affecting downstream benthos (Stevens et al., 1997). Two studies documenting post-removal effects determined that fish and macroinvertebrates were able to recover within a matter of months (Stanley et al., 2002) to years (Kanehl et al., 1997). Studies performed on five former impoundments in Wisconsin concluded that following dam removal exposed mudflats re-vegetated with non-native wetland species followed by xeric species as the flats began to dry up (Lehnart, 2000). Two of the sites took longer than 40 years to recover native species. Table 3.1 lists case studies documenting effects of dams and dam removals on various rivers throughout the United States. As dam removal becomes a more popular river restoration tool, more studies will be needed to document the effects of dam removal on ecological conditions via monitoring well into the future (Babbitt, 2002).

3.2.2 Objective of the Study

The main objective was to statistically determine water quality changes upstream and downstream of the St. John Dam for pre- and post-removal conditions.
3.2.3 *Hypothesis*

It was hypothesized that the St. John Dam removal would significantly reduce spatial (longitudinally, vertically, and transversely) variability of nutrients ($\text{NO}_3^-$, $\text{PO}_4^{3-}$, $\text{NH}_3$), water temperature, turbidity, DO, ORP, specific conductivity, and pH.

3.3 *Methods*

3.3.1 *Study Area - St. John Dam*

St. John Dam was a concrete low-head dam located on the Sandusky River in north central Ohio. At base flows, the reservoir impounded $5.6 \times 10^5 \text{ m}^3$ of water. The river drains approximately 3,680 km$^2$ of the Sandusky River watershed (Figure 3.1) and flows north to Lake Erie. Eighty-nine percent of the watershed is located in four counties: Sandusky, Seneca, Wyandot, and Crawford, which consist of 27 major cities or villages. Table 3.2 shows the different types of land use designated in the watershed. Along with the St. John Dam, the Sandusky River has four other dams. The St. John Dam was constructed in the 1930s to serve as a water supply for the city of Tiffin. It was originally 2.2 m high, 46 m wide, and was located at river mile 50 (SRWC, 2002). It was breached on March 18, 2003, by notching the dam on the west bank. The dam was then removed on November 17, 2003.

Late Wisconsin-age glacial till and glacial lake deposited sediments are the primary soils near the dam. These soils include: limestone (prone to dissolution) and beach ridges. Near the dam, parts of stream are exposed to the bedrock, which consists mainly of a Silurian Tymochtee and “Monroe” Dolostone (SRWC, 2002).
The Sandusky River Watershed receives its highest precipitation in July with as much as four inches of rain, and February receives the smallest rainfall events with as little as two inches of rain. On average, 36.6 inches of rain falls on the watershed every year with approximately 30% draining directly into surface waters (SRWC, 2002).

3.3.2 Sampling Design and Methodology

Sandusky River water was sampled and analyzed for nutrients (NO$_3^-$, PO$_4^{3-}$, NH$_3$) on a monthly basis from September of 2003 to August of 2004. In November, two sets of samples were taken; one before dam removal and one after dam removal. Samples were unable to be collected during January, February, and March because the river was frozen. No sample was taken in May as a result of high river conditions and safety reasons.

Samples were collected at four sites downstream from the dam and four sites upstream. Sites were determined based on location and accessibility. The upstream sites were located approximately 0.05 km (directly behind the dam, Figure 3.5), 6.4 km (Heck's Bridge, Figure 3.4), 17.7 km (county road 35, Figure 3.3), and 24.1 km (county road 16, Figure 3.2). The site directly behind the dam and the site at Heck's Bridge were known to be in the reservoir. The site at county road 35 was unknown as to whether it was still part of the reservoir, and the site at county road 16 was assumed to be out of the reservoir and thus used as a reference site. The downstream sites were located at 0.05 km, 0.50 km, 3.7 km (upstream of Scott’s Bridge, Figure 3.6), and 4.3 km (downstream of Scott’s Bridge, Figure 3.6). Grab samples were collected at the surface of the water using a 125 ml plastic bottle by either wading into the stream or using a canoe to paddle across the river. Three water grab samples were taken at each site; one at the left bank, one at the right bank, and one in the middle. The samples were put on ice and stored in the
laboratory at 4°C. Samples were shipped in small coolers packed with a single ice pack to STAR Lab in Wooster, Ohio, for chemical analysis. July samples taken at Heck’s Bridge were duplicated and analyzed for nitrate at Heidelberg College in Tiffin, Ohio, as well as Star Lab. Table 3.3 displays the results from Heidelberg compared to those of Star Lab. There was approximately a 26% increase in nitrate concentration in Star Lab’s samples as compared to Heidelberg’s. Phosphate samples taken in July at Heck’s Bridge were also analyzed via ICP AES in the Environmental Engineering Laboratory at The Ohio State University to determine whether Star Lab values were accurate. Star Lab values were consistently below 0.1 mg L$^{-1}$, which were confirmed by the ICP AES analysis measured concentrations of 6.0x10$^{-5}$ mg L$^{-1}$ on the left bank, 4.1x10$^{-5}$ mg L$^{-1}$ in the middle, and 4.0x10$^{-5}$ mg L$^{-1}$ on the right bank.

Five longitudinal surveys were performed from approximately 18 km above the dam down to the dam. DO, specific conductivity, temperature, pH, ORP, and turbidity were measured using a YSI 6920 or a YSI 6600 water quality sonde (Yellow Springs Instruments, Inc.). The longitudinal distributions of these variables were measured in the reservoir by deploying the sonde off the back of a standard five meter canoe and paddling from county road 35 down to the dam. Canoe speed was calculated based on travel time, which was approximated at one meter per second. The sonde was programmed to sample every five seconds and data were averaged over every 70-100 points or approximately 500 m. The surveys were performed once in October and once in November before dam removal and once in the months of December, June, and August after dam removal. A vertical profile was also performed behind the dam using the YSI 6920 to measure DO and temperature. Two methods were used to profile. First the YSI was slowly lowered
to the bottom and slowly retrieved back to the surface for three profiles. The second method was similar, but on the retrieval, the YSI was lifted completely out of the water (also performed three times). Raw data for the vertical profiles were logged every second. Data were binned to 0.2 m averages over depth (Table 3.4). Calibration of the sondes was performed using standard YSI calibration methods (YSI Incorporated, 1999). Accuracies for each of these variables are as follows; (pH +/- 0.2), (DO +/- 2% or 0.2 mg L\(^{-1}\)), (temp +/- 0.15°C), (turbidity +/- 5% or 2 NTU), (specific conductivity +/- 0.5% or 0.001 mS/cm), (ORP +/- 20 mV).

3.3.3 Discharge Data

Flow was determined from the gaging stations at Fremont, Upper Sandusky, and Tymochtee. Table 3.5 shows the three gaging stations and their relative drainage areas used to interpret river discharge near the dam. Table 3.6 shows discharge values for dates of the surveys.

3.3.4 Software

EcoWatch software was used to start, stop, and extract data from the YSI 6920 and YSI 6600 for both the longitudinal and vertical profiles performed. Data were statistically analyzed using MINITAB 14 and other various calculations were performed using Microsoft Excel.

3.4 Results

3.4.1 Vertical Dissolved Oxygen and Temperature Profile (Above the Dam)

Vertical DO and temperature profiles in the reservoir, directly behind the dam, showed decreasing trends with increasing depths (Figures 3.7 & 3.8). Surface to bottom
DO concentrations decreased from 14.31 mg L\(^{-1}\) to 12.95 mg L\(^{-1}\) and temperatures decreased from 6.19°C to 6.01°C at 2 m depth.

3.4.2 Longitudinal Nutrient Patterns

Phosphate and ammonia concentrations above and below the dam were undetectable for all periods sampled. Nitrate concentrations above and below the dam, for pre- and post-removal conditions, were high during the fall and low during the spring (Figure 3.9). After dam removal (i.e. after November 17\(^{th}\), 2003), concentrations were 2 to 2.5 times higher than before removal (September 10, October 10, and November 11, 2003), though this was probably a result of seasonal loading since values decreased in the spring. Generally, nitrate levels increased from 24.1 km upstream of the dam down to the dam, while below the dam, nitrate concentrations leveled off to similar concentrations just upstream of the dam. The largest variance in nitrate was in October, though no trend was evident from upstream to downstream. Immediately following dam removal, nitrate distributions mimicked trends from September and October 2003 with levels increasing from upstream down to the dam then remaining constant downstream of the dam. Three weeks after dam removal, in December 2003, concentrations appeared to level off with similar concentrations both above and below the dam. Spring and summer samples (April, June, July, and August 2004) were all below 4 mg L\(^{-1}\) and showed very little variability from upstream to downstream of the dam. Nitrate loadings calculated for all nine sample periods showed the highest loading during the winter months (Table 3.7). Also, there were no significant differences between the left, middle, and right samples (i.e. cross-sections) upstream and downstream of the dam during the study period (df = 2 and p > 0.569). For the pre-removal months of September, October, and November,
2003, significant differences were found in nitrate concentrations upstream of the dam (Table 3.8). In September, nitrate concentrations upstream were significantly different from concentrations in the reservoir and downstream, however, the reservoir was not significantly different from downstream (Table 3.8). Upstream, reservoir, and downstream nitrate levels were not significantly different from each other in October (Table 3.8). In November, however, upstream nitrate values were significantly different from the reservoir and downstream, and the reservoir was also significantly different from downstream. For the post-removal months, November, December, April, and August, upstream values were significantly different from the former reservoir and downstream, however, the former reservoir was not significantly different from downstream (Table 3.8). In June and July, upstream nitrate levels were significantly different from the former reservoir and downstream, and the former reservoir was also significantly different from downstream.

3.4.3 Longitudinal Water Quality Patterns (Above the Dam)

Spatial concentrations of DO in the river before and after dam removal showed a significant change from upstream of the reservoir down to the dam in October of 2003 (before removal; Figure 3.10). DO concentrations decreased by as much as 1.5 mg L$^{-1}$, however, in June, DO concentrations increased by 1 mg L$^{-1}$ from the beginning of the reservoir to the dam. In November and August there were no significant changes in DO concentrations. The DO trend can be seen more clearly from upstream to downstream as the deviation from the mean in DO concentrations for the four sampled periods (Figure 3.11). The change in DO concentration from upstream of the reservoir to within the
reservoir was related to changing discharge as you move downstream. This occurred in October, November, and August, but not in June (Figure 3.12).

Spatial temperature distributions showed an increase in temperature toward the dam in October of 2003, however, in November of 2003, there was no significant trend in temperature (Figure 3.13). In December, after dam removal, temperature was consistent throughout the former impoundment until approximately 1 km from the former dam where temperatures increased. The temperature trend can be seen more clearly from upstream to downstream as the deviation from the mean in temperature changes for the four sampled periods (Figure 3.14). In June, there was a significant increase in temperature close to the former dam, however, in August, there was no significant change in temperature throughout the former reservoir and upstream. Like DO, temperature was more variable closer to the dam. Heat transport (temperature values multiplied by discharge) accounted for the changes in temperature from upstream to within the reservoir in October, November, and December, however, in June and August change in heat transport could not be accounted for solely by discharge (Figure 3.15).

There were no changes in spatial turbidity distributions before and after dam removal from upstream down to the dam (Figure 3.16) even though November and December produced fluctuating trends throughout the study area. October and June of 2003 and August of 2004 had more consistent trends (Figure 3.16). Suspended solids loading showed that concentrations increased with increasing discharge (Figure 3.17) and were not associated with dam removal.

Figure 3.18 reveals trends in ORP above the dam. The pre-removal dates in October and November showed a slight increase in ORP closer to the dam. A similar
trend was seen after removal in December of 2003, however, data collected in June of 2004 shows ORP leveled off in the former impoundment with a decrease in the overall ORP compared to the previous months except for a slight decrease near the 12 km mark. This more reducing environment was consistent with the lower DO values in June (Figure 3.10). August showed a slight decrease from 17.9 km above the former dam down to 3 km above the dam where values increased.

In October and November of 2003, pH values were consistent from upstream down to the dam, ranging from $8.035 \pm 0.035$ to $8.17 \pm 0.03$, respectively (Figure 3.19). After dam removal, December values, $8.01 \pm 0.13$, were similar to the pre-removal conditions, but with a higher variation (Figure 3.19). The larger variation was a result of lower values from 17.9 - 15 km upstream of the dam and the higher values from 2 - 0 km upstream of the dam. June values were lower, $7.125 \pm 0.115$, with no significant increase from upstream of the reservoir to the end of the reservoir except approximately 1 km upstream from the former dam where values began to drop. In August, values were the lowest $6.385 \pm 0.145$ with a change in pH approximately 1.5 km upstream from the beginning of the former reservoir and no change downstream to the former dam.

No significant differences in the upstream and downstream values of specific conductivity occurred in any month (Figure 3.20). November and December of 2003, however, showed large fluctuations in specific conductivity from upstream to the former dam. The monthly changes in specific conductivity appeared to be the result of changes in discharge (Figure 3.21).
3.5 Discussion

Temperature and DO vertical profiles taken behind the dam in November of 2003 showed lake-like conditions, since the temperature decreased by 0.15°C and DO degreased by 1.5 mg L⁻¹. However, the vertical variation in temperature was near the accuracy of the probe (± 0.15°C) and may explain the change. The vertical difference in DO with depth was greater than the accuracy of the probe (0.2 mg L⁻¹) but opposite of what would be expected since DO saturation increases with decreasing temperature (and depth). The small change in temperature indicated that the water column was not well mixed. This may have been the result of increasing depth and decreasing water velocity created by the dam. Such conditions may have favored higher respirations rates near the bottom or lower oxygen diffusion from the surface to the bottom accounting for the lower DO in or near the bed.

The fact that there were undetectable levels of phosphate (<0.1 mg L⁻¹) in the water column at any time during the surveys may be a result of phosphate adsorbing to larger sediment particles and settling out in the substrate, thus becoming transported downstream via sediment transport. This is supported by laboratory data that showed total phosphorus concentrations in the reservoir sediments were on average 700 mg L⁻¹ (Qadir, Personal Communication). A combination of agricultural land use and a sediment rich river system can also lead to a high concentration of phosphorus in the substrate. Unlike nitrate, phosphate (PO₄³⁻) adsorbs to sediment particles and either settles out (Kennedy and Walker, 1990) or becomes mobile via sediment transport (Ng et al., 1993). Depending on the size of the particles, resuspension can lead to downstream transport.
Ammonia concentrations were also undetectable (<0.5 mg L\(^{-1}\)) over the survey. Historical ammonia concentrations below the dam (1974-1976) ranged from 0.02-0.52 mg L\(^{-1}\)-N with the lowest concentration in September and the highest in July (USGS Database, 2004). Most ammonia comes from agricultural runoff. The reduced ammonia in this study may have been the result of improved agricultural practices. Alternatively, ammonia may have been converted to nitrate via nitrification in drainage tiles and in floodplain sediment before it had a chance to reach the river.

Before dam removal, the reservoir was a source of nitrate with higher concentrations closer to the dam (Figure 3.9), however, nitrate levels downstream of the dam were the same as those in the reservoir. This is contrary to the data of Stanley and Doyle (2002) who showed elevated concentrations behind dams and reduced levels below dams. Eight days after dam removal, nitrate concentrations increased from upstream down to the dam even though the reservoir had drained. The source of nitrate could have been from drainage tiles, drainage ditches, surface runoff, groundwater discharge, or loading from Sycamore Creek (a significant inflow into the former reservoir). Changes in longitudinal concentrations eight days later may have been caused by those aforementioned factors. Five to nine months later, in the spring and summer, there appeared to be a consistent trend in nitrate from upstream to the dam, however, there was still a significant difference from upstream nitrate concentrations and former reservoir concentrations. The trend appeared more consistent in the summer (July) due to the lower range in concentrations (1.36-1.50 mg L\(^{-1}\)). The statistical differences in nitrate concentrations upstream of the reservoir, in the reservoir, and downstream of the reservoir for post-removal remained similar to pre-removal. Fall (November)
concentrations were much higher (16.30-18.33 mg L$^{-1}$). These higher values allow for higher ranges of values and concentrations that make the data significantly different in the three locations. Because there was no change in nitrate distributions before and after removal, dam removal did not affect the distributions in the river. Historical data (1970-1973) showed values ranging from 5.7-14.4 mg L$^{-1}$ in the summer, 8.6-18.0 mg L$^{-1}$ in the fall, 20.8-23.4 mg L$^{-1}$ in the winter, and 16.1-21-9 mg L$^{-1}$ in the spring (USGS Database, 2004). Fall concentrations fell within historical range, however, summer values were much lower than historical values. This may be the result of better farming practices and erosion control. Nitrate contamination has been seen frequently in the Sandusky River as a result of agricultural runoff. Samples taken at the Fremont gaging station during storm events have shown increased levels of nitrate exceeding the MCL of 10 mg L$^{-1}$. As nitrate concentrations exceed the MCL, exposure to infants becomes dangerous and can lead to “blue baby” syndrome (SRWC, 2002).

Soluble reactive phosphorus (SRP) and nitrate increase with increasing discharge. Peak concentrations of nitrate lag behind peak flows because most of the nitrate is transported via tile drainage. Concentrations of SRP and nitrate may become as high as five times the baseline concentration (Richards et al., 2001). Transformation of nitrate in large river systems is relatively small as a result of a smaller wetted perimeter to water volume ratio (Alexander et al., 2000). Reservoirs with higher hydraulic retention times promote phosphorus settling and retention of nitrate (Kelly et al., 1987; Jansson et al., 1994). Other studies suggest that nitrate removal in former impoundments will decrease as a result of a smaller sediment-water contact area (Stanley and Doyle, 2002). The channel will eventually widen and the depth of the water will decrease creating a larger
wetted perimeter to water volume ratio. Studies performed on four rivers in northwest Ohio found increases of nitrate relative to historical values (Richards et al., 2001). Causes were unknown, but the authors speculated that nitrate may have leached from soils as a result of mineralization of soil organic matter when pasture land was converted to cropland. Uncharacteristically high nitrate concentrations in the winter and low concentrations in the autumn were seen in the rivers from 1975-1995. These trends may have been a result of a combination of three things: more soy bean fields (which may have a slower rate of nitrate mineralization in the root nodules than corn crops), increased conservation tillage leading to an increase in coarse residue and a reduction of nitrate mineralization, and the decrease of winter flows and increase of autumn flows. Normally nitrate concentrations follow discharge patterns, but in this case they were opposite.

Studies performed by the Ohio Environmental Protection Agency have shown siltation, habitat alteration, and organic enrichment lead to low DO. These appear to be the major factors contributing to aquatic life impairment in the Sandusky River Watershed. Other factors such as non-irrigated crop production, urban runoff, channelization, and municipal point sources are also major contributors to decreased water quality.

The large decrease in DO from November to August (approximately 5.5 mg L⁻¹) was a result of seasonal change, not dam removal. However, the removal had a significant impact on the variability of the longitudinal DO surveys. The decreasing DO from upstream to downstream occurring during the pre-removal dates of October and November reciprocate the fact that the reservoir had a lower retention time (i.e. low velocity and increasing cross-sectional area closer to the dam). The variability in DO
levels before dam removal was larger than after removal. Because water was moving slower, shaded areas of the river may have had more of an impact on cooling the water, therefore, creating patchiness in DO. Dam removal created a faster moving river, shallower waters, and a smaller cross-sectional area throughout the entire study reach, which promoted less variability in DO concentrations.

Temperature trends cannot be determined based on the amount of pre-removal data obtained. One pre-removal data set showed an upward trend while the other showed a downward trend in the reservoir. There was, however, more variability in temperature in the reservoir than there was upstream of the reservoir. This may have been a result of decreased velocity and increased depth creating horizontal stratification of temperature throughout the reservoir. Post-removal temperature trends also cannot be determined. One result showed an increasing trend in the former reservoir while two other data sets showed a consistent temperature distribution. The post-removal data showed less variability than the pre-removal data. This may be a result of shallower depths and increased velocities creating a shorter residence time and higher turbulent mixing rates.

Historical data taken below the dam (1974-1976) showed temperature values ranging from 0-24°C (temperature values averaged per month) with the lowest temperature in February (USGS Database, 2004). The highest temperature was in August, coincidently with low flow (3.55 m³/s) conditions (USGS Database, 2004). For the dam removal, the lowest temperature was 0°C in February of 2004 and nearly 21°C in August of 2004. The fact that the highest temperature (August) was 3°C below normal, may have been a result of a colder summer, the higher discharge (37.21 m³/s), or the dam removal, which created a faster moving and shallower reach with lower temperatures.
Suspended sediments have posed a moderate problem throughout the Sandusky River. Following runoff events, small clay sized particles become suspended in the water column leading to an increase in turbidity. The turbidity (suspended solids) data did not show any conclusive evidence of a spatial trend when comparing upstream of the reservoir and in the reservoir nor is any temporal trend evident following dam removal. For all the dates, discharge was the major factor determining the turbidity. Storm runoff and bed erosion were the likely causes of seasonal turbidity changes.

ORP values were higher before dam removal as a result of seasonal variations. There was no consistent trend in ORP before or after dam removal. Further, there was no difference in variability of ORP before or after removal or upstream of the reservoir compared to the reservoir. Any variability was accounted for by the ± 20 mV accuracy of the probe.

No spatial trend in pH could be established from the data, however, there was a larger variability in pH after the dam was removed in and out of the reservoir. The variability cannot be used to interpret a trend because the accuracy of the pH probe was ± 0.2. Historical data taken below the dam (1970-1973) showed pH ranging from 7.79-8.11 (pH values averaged per month) with the lowest pH in June and the highest pH in November (USGS Database, 2004). This was consistent with the current spatial data. The highest value of 8.2 was obtained in November while June was lower at 7.1. The lowest value was 6.4 obtained in August. Historical data were taken below the dam and current data were taken above the dam, but no drastic change in pH should take place in a matter of 50 m. Thus, pH did not appear to be affected by dam removal.
Specific conductivity showed seasonal differences, however, there was no change from upstream of the reservoir to within the reservoir. There was also no change in trends after the dam was removed. The variability was large for one pre-removal sampling date and one post-removal sampling date. The other three periods had smaller variability within the reservoir. The large variability in values may have been a result of ions being utilized, released, and flowing into the system. It may have also been a result of the way in which the probe was corrected for temperature. Different ions react differently at different temperatures (Hem, 1989). Historical data taken below the dam (1970-1973) showed increasing specific conductivity with decreasing flow (USGS Database, 2004), which confirms the observed loading (Figure 3.21 and Table 3.6). High discharges from sub-watersheds, ditches, and drainage tiles may also explain any decreasing trends closer to the dam (Table 3.6).

3.6 Conclusions

Dam removal did not longitudinally affect pH, ORP, specific conductivity, turbidity, or nitrate distribution above the former dam. Changes in those variables appeared to be a function of season and/or discharge. However, dam removal eliminated vertical stratification of temperature and DO above the former dam. Also, variability in temperature and DO within the reservoir did decrease after removal as a result of decreased depth and increased velocity.
3.7 References


Chatterjee P. 1997. Dam busting. New Scientist. 3434


<table>
<thead>
<tr>
<th>Dam</th>
<th>Waterway</th>
<th>Pre-removal</th>
<th>Post-removal</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Glen Canyon</td>
<td>Colorado River</td>
<td>Water quality parameters affected habitat to a larger extent than did geomorphological influences</td>
<td></td>
<td>Stevens et al (1997)</td>
</tr>
<tr>
<td>Granby</td>
<td>Colorado River</td>
<td>Mean values of temperature, DO, and pH decreased with increasing distance downstream from dam</td>
<td></td>
<td>Camargo &amp; Voelz (1998)</td>
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<tr>
<td>Burgomillodo</td>
<td>Colorado River</td>
<td>Mean values of temperature, DO, and pH decreased with increasing distance downstream from dam</td>
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<td>Camargo &amp; Voelz (1998)</td>
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<td>Willow Falls</td>
<td>Willow River</td>
<td>Dam increased temperatures, caused sediment accumulation, and decreased DO upstream of the dam</td>
<td>Removal of dam allowed sediment to flow downstream</td>
<td>American Rivers et al (1999)</td>
</tr>
<tr>
<td>Mounds</td>
<td>Willow River</td>
<td>Dam increased temperatures, caused sediment accumulation, and decreased DO upstream of the dam</td>
<td>Removal of dam allowed sediment to flow downstream</td>
<td>American Rivers et al (1999)</td>
</tr>
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<td>Jackson Street</td>
<td>Bear Creek</td>
<td></td>
<td>Water temperatures decreased in the former impoundment</td>
<td>American Rivers et al (1999)</td>
</tr>
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<td>Williamsburg Station</td>
<td>Juniata River</td>
<td></td>
<td>The former impoundment became vegetated, nutrients flowed downstream, and DO increased</td>
<td>American Rivers et al (1999)</td>
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<tr>
<td>Woolen Mills</td>
<td>Milwaukee River</td>
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<td>In the former impoundment DO levels increased and temperature decreased</td>
<td>American Rivers et al (1999)</td>
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<tr>
<td>North Avenue</td>
<td>Milwaukee River</td>
<td></td>
<td>In the former impoundment DO increased, photosynthesis increased, and benthic respiration decreased</td>
<td>Hajda &amp; Novotny (1996)</td>
</tr>
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Table 3.1 – Water quality parameters pre- and post-removal at various study sites
Table 3.2 – Land use and area in the Sandusky River Watershed
(Courtesy of the Sandusky River Watershed Coalition)

<table>
<thead>
<tr>
<th></th>
<th>Total</th>
<th>Urban</th>
<th>Agricultural</th>
<th>Shrubs</th>
<th>Wooded</th>
<th>Water</th>
<th>Wetlands</th>
<th>Barren</th>
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<td>km²</td>
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<td>43</td>
<td>3090</td>
<td>18</td>
<td>465</td>
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<td>10</td>
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<tr>
<td>% cover</td>
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<td>1.2</td>
<td>84</td>
<td>0.5</td>
<td>12.6</td>
<td>0.4</td>
<td>1.1</td>
<td>0.2</td>
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Table 3.3 – Nitrate concentrations in mg/L at Heck’s Bridge in July
as determined by Heidelberg College and Star Lab

<table>
<thead>
<tr>
<th></th>
<th>Left bank</th>
<th>Middle</th>
<th>Right bank</th>
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<tr>
<td>Heidelberg</td>
<td>1.10</td>
<td>1.11</td>
<td>1.10</td>
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<tr>
<td>Star Lab</td>
<td>1.48</td>
<td>1.50</td>
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<td>% change</td>
<td>25.68</td>
<td>26.00</td>
<td>26.67</td>
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Table 3.4 – Number of samples grouped per depth profile
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<th>Gaging Station</th>
<th>Drainage (m$^3$)</th>
<th>Average Flow/yr (cfs)</th>
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<tr>
<td>Sandusky River near Upper Sandusky</td>
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<td>246</td>
</tr>
<tr>
<td>Tymochtee Creek at Crawford</td>
<td>229</td>
<td>188</td>
</tr>
<tr>
<td>Sandusky River near Fremont</td>
<td>1251</td>
<td>1027</td>
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Table 3.5 – Drainage areas and average yearly flow of gaging stations used to calculate flow near the dam

<table>
<thead>
<tr>
<th>Date</th>
<th>Q (m$^3$/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>9/10/2003</td>
<td>1.33</td>
</tr>
<tr>
<td>10/10/2003</td>
<td>2.21</td>
</tr>
<tr>
<td>10/24/2003</td>
<td>3.96</td>
</tr>
<tr>
<td>11/11/2003</td>
<td>2.89</td>
</tr>
<tr>
<td>11/25/2003</td>
<td>8.86</td>
</tr>
<tr>
<td>12/9/2003</td>
<td>15.77</td>
</tr>
<tr>
<td>4/23/2004</td>
<td>9.6</td>
</tr>
<tr>
<td>6/29/2004</td>
<td>4.47</td>
</tr>
<tr>
<td>7/30/2004</td>
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</tr>
<tr>
<td>8/30/2004</td>
<td>37.21</td>
</tr>
<tr>
<td>9/10/2004</td>
<td>10.87</td>
</tr>
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</table>

Table 3.6 – Discharge data for sampling dates

47
<table>
<thead>
<tr>
<th>Date</th>
<th>Discharge</th>
<th>Ave NO3</th>
<th>Load</th>
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<tr>
<td></td>
<td>m^3/s</td>
<td>mg/l</td>
<td>kg/dy</td>
</tr>
<tr>
<td>9/10/2003</td>
<td>1.33</td>
<td>8.54</td>
<td>982</td>
</tr>
<tr>
<td>10/10/2003</td>
<td>2.21</td>
<td>12.09</td>
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<td>11/11/2003</td>
<td>2.89</td>
<td>10.35</td>
<td>2583</td>
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<td>11/25/2003</td>
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<td>17.81</td>
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<td>12/9/2003</td>
<td>15.77</td>
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<td>4/23/2004</td>
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<td>3.32</td>
<td>1283</td>
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<tr>
<td>7/30/2004</td>
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<td>1.40</td>
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<td>8/30/2004</td>
<td>37.21</td>
<td>2.06</td>
<td>6622</td>
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Table 3.7 – Nitrates loadings for the nine sample periods

<table>
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<th>Nitrate Significance</th>
<th>Tukey α = 0.05</th>
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<td><strong>Removal/Date</strong></td>
<td><strong>DF</strong></td>
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<tr>
<td>Pre 9/10/2003</td>
<td>7</td>
</tr>
<tr>
<td>Pre 10/10/2003</td>
<td>7</td>
</tr>
<tr>
<td>Pre 11/11/2003</td>
<td>7</td>
</tr>
<tr>
<td>Post 11/21/2003</td>
<td>7</td>
</tr>
<tr>
<td>Post 12/9/2003</td>
<td>7</td>
</tr>
<tr>
<td>Post 4/23/2004</td>
<td>7</td>
</tr>
<tr>
<td>Post 6/29/2004</td>
<td>7</td>
</tr>
<tr>
<td>Post 7/30/2004</td>
<td>7</td>
</tr>
<tr>
<td>Post 8/30/2004</td>
<td>7</td>
</tr>
</tbody>
</table>

U = upstream  R = reservoir  D = downstream
U, R, D = If letters are present, there is a statistical significant difference

Table 3.8 – Statistical Tukey test to determine if nutrient values are significantly different from upstream of the reservoir to downstream of the reservoir
3.9 Figures

Figure 3.1 – Sandusky River Watershed (Hydrologic Unit Code, HUC, 04100011070)
Figure 3.2 – Sample site at County Road 16 (upstream of dam)

Figure 3.3 – Sample site at County Road 35 (upstream of dam)
Figure 3.4 – Sample site at Heck’s Bridge (upstream of dam)

Figure 3.5 – Sample sites at St. John dam (upstream and downstream of dam)
Figure 3.6 – Sample sites at Scott’s Bridge (downstream of dam)
Dissolved Oxygen vs Depth profile

Figure 3.7 – Mean (plus-minus std deviation) of vertical DO profile above the dam before removal

Temperature vs Depth profile

Figure 3.8 – Mean (plus-minus std deviation) of vertical temperature profile above the dam before removal
Figure 3.9 – Spatial nitrate distribution at eight different locations before and after dam removal upstream and downstream of St. John Dam
Figure 3.10 – St. John spatial DO profile (plus-minus two standard deviations) (points averaged over 500 m)

Figure 3.11 – St John spatial DO profile above the dam (mean subtracted from each point)
Figure 3.12 – DO values normalized to discharge upstream of the reservoir (17.9 km-13.7 km above the dam) and within the reservoir (9 km-5 km above the dam) (plus-minus 2 standard deviations)
Figure 3.13 – St. John spatial temperature profile (plus-minus two standard deviations) (points averaged over 500)

Figure 3.14 – St. John spatial temperature profile above the dam (mean subtracted from each point)
Figure 3.15 – Temperature values normalized to discharge upstream of the reservoir (17.91km-13.7km above the dam) and within the reservoir (9km-5km above the dam) (plus-minus 2 standard deviations)
Figure 3.16 - St. John spatial turbidity profile (plus-minus largest 2 standard deviations) (points averaged over 500 m)
Note: pre-removal is on the left axis; post-removal is on the right axis

Figure 3.17 – Suspended solids loading upstream of the reservoir (17.9 km-13.7 km above the dam) and within the reservoir (9 km-5 km above the dam) (plus-minus 2 standard deviations)
Figure 3.18 - St. John spatial ORP profile (plus-minus 2 standard deviations) (points averaged over 500 m)

Figure 3.19 - St. John spatial pH profile (plus-minus 2 standard deviations) (points averaged over 500 m)
Figure 3.20 - St. John spatial specific conductivity profile (plus-minus largest 2 standard deviations) (points averaged over 500 m) Note: pre-removal is on the left axis; post-removal is on the right axis.

Figure 3.21 – Specific conductivity values normalized to discharge upstream of the reservoir (17.9 km-13.7 km above the dam) and within the reservoir (9 km-5 km above the dam) (plus-minus 2 standard deviations).
CHAPTER 4

DENITRIFICATION POTENTIAL AFTER DAM REMOVAL

4.1 Abstract

Denitrification potential was measured longitudinally and laterally in the former reservoir of the St. John Dam after dam removal. There were no significant differences in denitrification potential in sediments longitudinally down the channel, from the upper to the lower limits of the former reservoir. There were also no lateral differences in potential across the channel, from the river bed to the newly exposed banks to the historic floodplain. However, there was a significant difference between samples treated with deionized water (control treatment) and samples treated with nitrate, indicating nitrate limitation. There was also a significant difference between the control and samples treated with nitrate + carbon. However, there was no significant difference between the nitrate and nitrate + carbon treatments, suggesting no carbon limitation. Further, as a result of dam removal, there was a loss of sediment-water contact from the decrease in reservoir depth, however, this loss did not significantly change the amount of nitrate being lost in the system.
4.2 Introduction

4.2.1 Background

Denitrification is a microbial process where nitrate (NO$_3^-$) is first reduced to nitrous oxide (N$_2$O), then nitrogen gas (N$_2$) under anoxic conditions. Denitrifying bacteria regulate this process while requiring nitrogen oxides (terminal electron acceptors) and organic carbon (electron donars) (Livingstone et al., 2000). Denitrification can be an important mechanism in nitrogen-rich waters that can lead to decreases in nitrate concentrations and increases in atmospheric nitrous oxide. Denitrification mainly occurs in the substrate of aquatic environments due to the lack of oxygen in the sediment. This process is controlled by concentrations of nitrate, dissolved oxygen, and organic carbon (Esteves et al., 2001).

Of the three factors affecting denitrification, nitrate is usually the most limiting factor in freshwater environments (Seitzinger, 1988; Grofman, 1991). Studies performed on Lake Batata, an Amazonian Lake, found nitrate limitation as a result of increased production of N$_2$O in samples amended with nitrate (Esteves et al., 2001). Carbon was determined to be an ancillary factor in controlling the denitrification in the lake sediments. The assays determined that temperature and oxygen at the sediment-water interface did not affect denitrification in the sediments.

Studies performed on nitrate-rich riverbed sediments in the South Platte River, Colorado, determined that N$_2$O production did not increase with increasing nitrate concentrations (Pfenning and McMahon, 1996). This study concluded that denitrification was not limited by nitrate availability. However, there was a significant increase in N$_2$O production when acetate concentrations were increased. This led to the conclusion that
organic carbon was limiting in the system. Further data indicated that shallow riverbed sediments may support higher denitrification rates than deeper areas of the river. The difference in organic carbon may contribute to the change in rates.

Three groups of bacteria perform denitrification in flooded waters; free-living bacteria in the water, bacteria in the flooded sediment, and bacteria on periphyton attached to plants (Weisner et al., 1994; Eriksson and Weisner, 1999). Studies found that denitrification takes place at oxygen concentrations less than 2 mg L\(^{-1}\) in water or sediment (Seitzinger, 1988). Other studies performed on biofilm determined that denitrification begins at oxygen concentrations of 0.3-0.6 mg L\(^{-1}\) (Nielsen et al., 1990; Dalsgaard and Revsbech, 1992). However, in a similar study it was determined that denitrification will take place in nitrate-rich rivers at oxygen levels of 0.8 mg L\(^{-1}\) or less (Venterink et al., 2003). It was also determined that denitrification rates were higher in flooded sediments than at surface waters. Studies performed on a Danish stream found that diffusion of nitrate from the water column into the substrate was the major nitrate source (Christensen et al., 1990), however, another stream study determined that nitrate produced in the sediment was the major nitrate source in substrate denitrification (Seitzinger, 1988).

Denitrification potential was found to be highest in the top 5 cm of the substrate and lower with increasing depth (Livingstone et al., 2000). Organic carbon was found to be highest in the top 1 cm of the substrate. Seasonally, denitrification potential was highest in the mid to late summer and was found to significantly correlate with C/N ratios and temperatures.
Floodplains play a large role in nitrate removal from river water through denitrification and uptake (Gilliam et al., 1997). The uppermost part of the floodplain soil is rich in soluble carbon and has a high potential for denitrification, however, the soil is rarely saturated with water. At greater depths the soil becomes saturated with water more often, however, there is little soluble carbon and denitrification potential is low (Burt and Haycock, 1996). Denitrification rates in submerged floodplains were similar to normal rates of denitrification in wetland soils (Venterink et al., 2003). For dammed rivers, reservoirs with higher hydraulic retention times increase denitrification (Kelly et al., 1987; Jansson et al., 1994). Other studies suggest that nitrate concentrations in previously impounded areas will decrease as a result of a smaller sediment-water contact area (Stanley and Doyle, 2002).

4.2.2 Objective of the Study

The main objective was to determine how denitrification potential differed longitudinally and laterally in the reservoir for the first nine months following dam removal.

4.2.3 Hypotheses

1) Denitrification would increase in newly exposed reservoir sediments

2) The floodplain would have a higher potential than the river bed and the newly exposed mudflat as a result of more available carbon

3) The upper and lower reach of the reservoir would have a higher denitrification potential than the middle reach of the reservoir
4.3 Methods

The study site was the former reservoir of the St. John Dam, located 50 km upstream of county road 6, on the Sandusky River, Ohio. The dam was removed on November 17, 2003. Sediment cores were collected in the spring, summer, and fall of 2004 at locations 0.05 km upstream of the pre-existing dam, 6.5 km upstream of the pre-existing dam (Heck's Bridge), and 11 km upstream of the pre-existing dam (Mexico Bridge). Mexico was the upper limit of the reservoir. At each site, sediment samples were taken from the submerged river bed, from the newly exposed bank (mudflat), and from the historic floodplain. Five cores were taken at each location (15 at each site) using a T-shaped soil sampler. Each core was approximately 3 cm in diameter and 30 cm long. The five cores were homogenized in a Ziploc bag and the samples were stored at 4°C until they were ready to be analyzed.

All samples were analyzed for denitrification potential using a soil slurry acetylene block method (Esteves et al., 2001). For each sample, 25 grams of wet soil were added to three 120 ml bottles (i.e., 25 g per bottle). Sub-samples of soil were dried for dry weight determinations. The first set of bottles was filled with 25 ml of deionized (DI) water and served as a control. The second set of bottles was amended with 25 ml of 100 mg L⁻¹ NO₃⁻ (as KNO₃). The third set of bottles was filled with 12.5 ml of 200 mg L⁻¹ NO₃⁻ (as KNO₃) and 12.5 ml of 200 mg L⁻¹ C₁₂H₂₀O₁₂ (dextrose as a carbon source). The bottles were capped with rubber septums and aluminum seals. Oxygen was evacuated from each bottle by vacuum for approximately five minutes and then flushed with helium gas for 3 - 5 seconds to create anaerobic conditions. Acetylene (12 ml) was added through the septums and the bottles were then incubated at 26°C. Gas samples
(6 ml) were taken with syringes after 3, 6, 12, and 24 hours. Prior to sampling, syringes were used to flush the gas in and out of the bottle for complete mixing. The gas samples were injected into pre-evacuated 5 ml bottles and then stored at 4°C. They were then analyzed on a Shimadzu 14C Gas Chromatograph with an ECD detector.

Statistical analysis was done using MINITAB 14. All tests were performed using an ANOVA at a 95% confidence interval. A Tukey test was used to compare means. There were no replicates taken for any of the site samples, therefore, all data were pooled together from spring, summer, and fall where no difference in rates were observed. Longitudinal sites (Mexico, Heck’s, St. John), cross-sectional sample sites (river, mudflat, floodplain), and the three treatment types (control, nitrate, nitrate + carbon) were also grouped with one another to increase the number of replicates.

4.4. Results

At Mexico Bridge (11 km upstream), denitrification potential was lowest in the river bed when no nitrate or carbon (control) was added, higher when amended with nitrate only, and highest when amended with nitrate + carbon (Figure 4.1). The results were similar for samples taken from mudflat and floodplain soils, however, both the mudflat and floodplain had higher potentials than the river for the control, the nitrate only treatment, and the nitrate + carbon treatment. The denitrification potential in the floodplain was higher than in the mudflat for the control and nitrate + carbon treatments. However, the potential in the mudflat was higher than the potential in the floodplain for the nitrate treatment.
At Heck’s Bridge (6.5 km upstream), denitrification potential was low in the control and higher when amended with nitrate only and nitrate + carbon treatments for all three cross-sections (Figure 4.2). Denitrification potential in the river was higher with nitrate than it was with nitrate + carbon, however, the mudflat and floodplain denitrification potentials were higher when amended with nitrate + carbon than they were when amended with nitrate only.

At St. John Dam (0.05 km upstream), denitrification potential was low in the control samples, higher when amended with nitrate only, and highest when amended with nitrate + carbon for all three cross-sections (Figure 4.3). The river substrate had similar denitrification potentials when amended with nitrate only and when amended with nitrate + carbon. The floodplain had the highest control potential and the highest potential in the sample amended with nitrate only. The mudflat had the highest potential when amended with nitrate + carbon.

The seasons were grouped together with the treatments, for added replicates, to test denitrification potential in the river, mudflat, and floodplain. There was no significant longitudinal difference in denitrification potential for the river substrate (p = 0.321), for the mudflat soil (p = 0.685), or for the floodplain soil (p = 0.548). When the seasons were grouped along with the cross-sections, there was no significant difference in the denitrification potential longitudinally for the controls (p = 0.894), the samples amended with nitrate only (p = 0.503), and the samples amended with nitrate + carbon (p = 0.382). When the seasons were grouped with the cross-sections and the treatments there was also no longitudinal statistical difference (p = 0.442).
The longitudinal sites were grouped with the seasonal data to compare the control with the nitrate treatment for the three cross-sections. There was not a significant difference between the controls and the treatment with nitrate for the river soil (p = 0.018), the mudflat soil (p = 0.013), or the floodplain soil (p = 0.026). There was, however, a significant difference between the control and nitrate treatments when all cross-sections where grouped along with the seasons and longitudinal sites (p < 0.001) (Figure 4.4).

The longitudinal sites were again grouped with the seasonal data, this time to compare the control and nitrate + carbon treatment in the three cross-sections. There was a statistical significant difference between the control and nitrate + carbon for the river soil (p = 0.001), the mudflat soil (p < 0.001), and the floodplain soil (p < 0.001). When the cross-sections were also grouped with the seasonal data and longitudinal sites, there was also a significant difference in treatments (p < 0.001).

The longitudinal sites were once again grouped with the seasonal data to compare the nitrate treatment with the nitrate + carbon treatment in the three cross-sections. There was no significant difference between nitrate and nitrate + carbon for the river soil (p = 0.687), the mudflat soil (p = 0.068), or the floodplain soil (p = 0.069). When the cross-sections were also grouped with the longitudinal sites and seasonal data there was also no significant difference in treatments (p = 0.015).

4.5 Discussion

There was no significant difference in denitrification potential longitudinally or laterally in the former reservoir. The nitrate treatment was significantly different than the
control, indicating nitrate limitation. The nitrate treatment was not different than the nitrate + carbon treatment, suggesting no carbon limitation. The nitrate + carbon treatment was significantly different than the control, and since the nitrate + carbon treatment was not significantly different than the nitrate treatment, nitrate limitation can once again be concluded.

Denitrification potential for the former St. John Reservoir averaged 5 mg N₂O g⁻¹ soil h⁻¹ (based on controls). This is a factor of 10³ higher than in a river system in Oxford, England (Burt et al., 1999), which had an N₂O production rate of approximately 100 kg N₂O ha⁻¹ day⁻¹ which is equivalent to 5x10⁻³ mg N₂O g⁻¹ soil h⁻¹. The discrepancies in denitrification values may have resulted from a slightly different denitrifying enzyme assay technique used by Burt et al. (1999). Studies performed on denitrification potential in the South Platte River using a similar acetylene block technique found river samples amended with nitrate had denitrification potentials of 2x10⁻⁴ mg N₂O g⁻¹ soil h⁻¹ (Phenning and McMahon, 1996). Studies performed on denitrification potential in the Belfast and Strangford Loughs in Northern Ireland using the acetylene block technique found an average denitrification potential of 10⁻⁴ mg N₂O g⁻¹ soil h⁻¹ (Livingstone et al., 2000). Studies performed on denitrification potential in Lake Batata in the northern Brazilian Amazon using the acetylene block technique found denitrification potentials of 10⁻⁴ mg N₂O g⁻¹ soil h⁻¹ (Esteves et al., 2001). These are 10⁷, 10⁶, and 10⁵ fold decreases from the results in the Sandusky River study where values were determined to be 10² mg N₂O g⁻¹ soil h⁻¹. This discrepancy may be the result of several differences including; river characteristics, laboratory techniques, and/or incubation temperatures.
Sediment-water contact is imperative for denitrification to occur in river substrates. Before the dam was removed, the wetted perimeter was 45.04 m and the length of the reservoir was approximately 13.7 km. Studies have determined that denitrification potential is highest in the top 0.05 m of substrate (Livingstone et al., 2000). A depth of 0.05 m was assumed for determining the volume of available substrate for denitrification. The total volume of the reservoir was 30,852 m$^3$ (13,700 m x 0.05 m x 45.04 m). Using a soil porosity of 0.45 (i.e., 45% water and 55% soil), the total volume of sediment in the reservoir was 16,986 m$^3$ (30,852 m$^3$ x 0.55). After the dam was removed the wetted perimeter decreased to 42.00 m. The total volume of the former reservoir was 28,770 m$^3$ (13,700 m x 0.05 m x 42.06 m). Again, taking into account the porosity of the soil, the total volume of sediment left in the former reservoir was 15,823 m$^3$ (28,770 m$^3$ x 0.55). The sediment volume lost was 1,163 m$^3$ (16,986 m$^3$ - 15,823 m$^3$). The depth of the reservoir dropped 1.52 m (a linear distance of 3.04 m taking into account both banks). Mass of sediment lost was 3.0x10$^6$ kg (2,650 kg m$^{-3}$ density of sediment x 1,163 m$^3$ of sediment).

Assuming for every mole of NO$_3^-$ taken up there was one mole of N$_2$O produced, denitrification potential results produced, on average, 100 mg N$_2$O g$^{-1}$ soil hr$^{-1}$ when nitrate was not limiting. The amount of sediment volume lost translates into a loss of denitrification, a result of dam removal. The loss of denitrification was estimated as 3.0x10$^{11}$ mg NO$_3^-$ h$^{-1}$ (3.0x10$^6$ kg of soil x 100 mg N$_2$O g$^{-1}$ soil h$^{-1}$) or 84 kg NO$_3^-$ s$^{-1}$ based on the volume of the reservoir that was lost. The percent of sediment volume lost was 6.8% \([(1 - (15,823 \text{ m}^3 / 16,986 \text{ m}^3)) \times 100], \) which translates into the percent of denitrification lost by draining the reservoir. The total denitrification before dam
removal was 1,240 kg NO$_3^-$ s$^{-1}$ (84 kg NO$_3^-$ s$^{-1}$ / 0.068). The total after dam removal was 1,156 kg NO$_3^-$ s$^{-1}$ (1,240 kg NO$_3^-$ s$^{-1}$ – 84 kg NO$_3^-$ s$^{-1}$).

It was assumed that nitrate in the sediment was in equilibrium with concentrations in the overlying water column. For high nitrate concentrations in the river, (18 mg L$^{-1}$ in winter, 2003), there was 2,500 kg of NO$_3^-$ (0.45 x 30,852 m$^3$ x 18 mg L$^{-1}$) in the pore water before dam removal and 2,330 kg of NO$_3^-$ (0.45 x 28,770 m$^3$ x 18 mg L$^{-1}$) in the pore water after dam removal. The former reservoir still had the ability to take up 1,156 kg NO$_3^-$ s$^{-1}$. This rate was still high enough to denitrify the 2,330 kg of NO$_3^-$ in approximately two seconds. Dam removal did not significantly affect the ability of the system to remove nitrate.

Stanley and Doyle (2002) proposed a conceptual model for a dam removal in Wisconsin. It was determined that nitrate removal was highest during pre-removal (Stage A) and post-removal aggradation and widening (Stages E & F). It was lowest during intermediate stages, while water depth decreased and channel degradation and widening occurred (Stages B – D). According to Stanley and Doyle (2002), the decrease in nitrate removal was the result of a smaller sediment-water contact area. Contrary to Stanley and Doyle (2002), removal of St. John Dam did not trigger Stages B – D, so there was very little change in wetted perimeter. The small wetted perimeter (denitrification) that was lost as a result of dam removal was insignificant because of the amount of wetted perimeter left for denitrification to occur. Of the wetted perimeter lost, most of it was lost near the dam with the smallest amounts lost near the beginning of the former reservoir, however, denitrification potential was not significantly different at any of the longitudinal locations.
Burt et al. (1999) found that the acetylene block method overestimated field denitrification rates by two orders of magnitude. This may have been one of the reasons why results were six and seven orders of magnitude larger. Five cores were taken at each location within the cross-section and homogenized. Incomplete mixing may have led to different microbial population sizes in each sample. Even if the samples were completely homogenized, there may have been a different microbial community and population at a sampling location ten feet away. This would not be representative of the river as a whole. After collection, samples were stored in a cooler for as long as two weeks where microbial communities may have changed. This may have led to denitrification potential results not indicative of microbial communities in the river. The method used sediment slurries with precise solutions of nitrate, nitrate + carbon, and gas injections of helium and acetylene. These measurements may have been different than other studies and may have even been slightly different for each sample causing inaccuracies. The slurries were incubated at temperatures larger than in situ temperatures. This would not have given a good representative measurement of what the microbial activity was like. Replicates at each site would have also given a more accurate statistical analysis when looking for statistical differences. This may have been why carbon appeared (Figure 4.1) to be a limiting factor but statistically was not.

4.6 Conclusions

It was determined that denitrification did not significantly increase in the newly exposed reservoir sediments and there were no significant differences in the cross-sections of the river. Also, it was found that denitrification was not significantly different
iongitudinally in the former reservoir. Further, it was concluded that even though wetted perimeter was lost, enough remained to provide similar amounts of nitrate removal. For the Sandusky River, nitrate appeared to limit denitrification. Nitrate loading appeared to be associated with seasonal discharges, thus denitrification would also be seasonal.

Given the above changes in reservoir and wetted perimeter, dam removal had no significant effect on the remediation of nitrate.

4.7 References


4.8 Figures

**Figure 4.1** – Denitrification potential averaged over spring, summer, and fall samples at Mexico Bridge.
Figure 4.2 – Denitrification potential averaged over spring, summer, and fall samples at Heck’s Bridge.
Figure 4.3 – Denitrification potential averaged over spring, summer, and fall samples at St. John Dam.
Figure 4.4 – Denitrification potential with pooled seasonal, longitudinal, and cross-sectional data. This graph indicates nitrate limitation when variables are pooled.
CHAPTER 5

DISCUSSION

Dam removal was once seen as having negative impacts on local riparian environments. Recreational opportunities were lost as well as city water supplies and downstream flood control measures. Today, however, dam removal has become a new and popular tool for river restoration as well as an alternative to expensive dam maintenance, at least for older dams. Effects of dam removal have not been well studied in the past. Recent studies have begun to document removals and are determining whether the removal procedure is causing permanent, temporary, positive, negative, and/or neutral effects to a river system (Shuman, 1995; Calaman and Ferreri, 2004; Stanley et al., 2002; Lehnart, 2000; American Rivers et al., 1999).

Removal of the St. John Dam was a short and relatively non-problematic procedure. The removal brought the river closer to natural conditions by retuming the former reservoir to a more riverine system. Riffles were exposed, velocities increased, and new floodplains began to form in reaches of the former reservoir. There is no longer stratification of temperature and DO directly behind the dam and longitudinal surveys indicated less variability in temperature and DO throughout the study reach. Thus, while local aeration capacity was lost at the dam as the water spilled over the crest, exposure of
riffles upstream may compensate by increasing DO as a result of turbulent mixing. However, the dominant changes in DO and temperature from the upstream reach to within the reservoir were caused by fluctuations in discharge.

Nutrient concentrations were seasonal, while nutrient loading was dependent on discharge. There is significant downstream to upstream variations in nitrate as suggested by Stanley and Doyle (2002), however, longitudinal distributions of nitrate for the St. John Dam showed no significant changes from upstream to downstream before or after removal. Additionally, none of the water quality variables changed longitudinally.

Turbidity and sediment loadings indicated that temporal trends and spatial patterns in turbidity were driven by discharge (i.e. rainfall and runoff) events and were not a result of the dam removal. Immediately following the dam removal, turbidity increased to a magnitude equivalent to a moderate storm event, however, the duration of the increase was shorter. The conclusion is that dam removal did not significantly affect downstream reaches with abnormally high transport of suspended sediment. The breach of the St. John Dam caused an insignificantly slight increase in volumetric flow which could not be seen 53 km downstream. On the other hand, dam removal caused a larger increase in flow that was evident at the same location 53 km downstream. This was determined to be similar to a 2 cm rainfall event that occurred less than one week later. This concluded that discharge did not unnaturally affect downstream reaches.

The studies performed on the St. John Dam were a good start, but these studies need to continue to fully determine how dam removal affects a riverine system.

The denitrification lost as a result of the lost sediment-water contact was insignificant compared to the amount of denitrification remaining in the study reach. The
removal of nitrate via denitrification was more a function of nitrate limitation, and seasonal temperatures than the dewatering of the reservoir. If the geomorphology of the former reservoir causes incision of the bed and widening of the channel, loss of nitrate by denitrification could increase (Stanley and Doyle, 2002). If this happens, dam removal would remain a useful restoration tool without losing the potential for denitrification.

In the future, dam removal will become more and more common as the life expectancy of many dams comes to an end. Studies of dam removals need to continue to document pre-versus post-removal effects upstream and downstream of the dams. New and better tools need to be developed to increase the ability to restore reaches surrounding dams. Further, guidelines for removal and restoration need to be prepared from case studies documenting dam removal procedures. Consequently, data need to be taken for each dam removal, along with dam characteristics, to appropriately classify how each dam affects the surrounding environment. With the tools, resources, and knowledge from such studies, dam removal will become much more predictable and easier to accomplish from an engineering standpoint.
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