May 2018

# The Unintended Benefits of Dams Should Be Considered Prior to Removal

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# THE UNINTENDED BENEFITS OF DAMS SHOULD BE CONSIDERED PRIOR TO REMOVAL

by

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A Thesis Submitted in

Partial Fulfillment of the

Requirements for the Degree of

Master of Science

in Freshwater Sciences and Technology

at

The University of Wisconsin-Milwaukee

May 2018

ABSTRACT

THE UNINTENDED BENEFITS OF DAMS SHOULD BE CONSIDERED PRIOR TO REMOVAL

by

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The University of Wisconsin-Milwaukee, 2018
Under the Supervision of Professor Michael Carvan

Dams provide multiple benefits; however, they also degrade rivers. Many dams no longer serve their intended purpose and are nearing the end of their operational lives. The aging of dams coupled with the cost of restoration and maintenance, regulation, and the ecologic impacts of dams has resulted in removal becoming a viable management alternative. Despite increased utilization, limited research and a lack of quantitative predictive capacity results in large amounts of uncertainty associated with impacts of dam removal. Additionally, dams which no longer serve their intended purpose may still have unintended positive benefits such as the prevention of the spread of invasive species, providing recreational opportunities, and the treatment of water pollution from upstream reaches. Failure to identify unintended benefits of dams can prevent accurate determinations of the potential consequences of removal. Therefore, identifying unintentional benefits of a dam is an essential step in determining the potential impacts of dam removal. The Mill Pond Dam in the Oak Creek Watershed is being considered for removal. The dam may be a barrier to invasive species, provide recreational opportunities, and improve downstream water quality.

A fish passage survey was conducted at all stream crossings within the watershed to determine if the dam is a barrier to invasive species. A bathymetric survey was conducted and *E.*

*coli* data was collected to determine if the dam provides recreational opportunities. A suite of water quality parameters was assessed weekly for 61 weeks at six sites upstream, within, and downstream of the impoundment to determine if the dam improved physical, chemical, or biological aspects of downstream water quality. The results of this study indicate that the Mill Pond Dam may have acted as a barrier to invasive species, the impoundment did not provide recreational opportunities, and the dam did not improve downstream water quality during the study period. The methods utilized in this study are transferable and could be used at other locations to improve the understanding of the potential consequences of dam removal.

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# ACKNOWLEDGEMENTS

I would like to acknowledge the support of the City of Racine Health Department Laboratory. In particular, I would like to thank Julie Kinzelman, Stephan Kurdas, and Adrian Koski, Jennifer Creekmur, and the numerous student interns who assisted in data collection and analysis.

I would like to thank the members of my committee for their insights and the time, the Southeastern Wisconsin Regional Planning Commission for providing stream crossing survey data, and The Fund for Lake Michigan who funded this study.

I would also like to thank my friends and family who support all my ambitions.

# 1. Literature Review

## Dams

### *Effects of dams*

Dams provide a variety of beneficial uses including power generation, flood control, irrigation, water supply, and recreation (Bushaw-Newton et al. 2002, Baxter 1977). For example, the High Falls Dam is a 47 foot high hydroelectric dam located on the Peshtigo River in Wisconsin, the Maplebrook Estates Dam is a private dam located in Bolingbrook Illinois on the Lily Cache Creek that is utilized to provide 97 acre feet of flood storage, the Peckerwood Lake Dam is a private dam on the La Grue Bayou in Arkansas with 28,000 acre feet of storage that is utilized for irrigation, and the Richard B. Russel Dam is a 136 foot high, federally owned dam on the Savannah River that is utilized for water supply and recreation (US ACE, 2017). Despite these benefits, dams degrade rivers by fragmenting channels and altering sediment, thermal, and flow regimes (Baxter 1977, Syvitski et al. 2005, Ward and Stanford 1995). Initially the newly created barrier alters flow regimes by increasing water retention behind the dam, which slows flow and creates a lake-like environment within the impoundment (Bushaw-Newton et al. 2002, Lejon et al. 2009). The slowed flow rates impact the stream's thermal regime within and downstream of the impoundment and result in increased sedimentation within the impoundment (Bednarek 2001, Lejon et al. 2009). As particles settle in the impoundment nutrients, bacteria, and contaminates (ex. heavy metals, PAH's etc.) that are sorbed to the sediment are removed from the water column and deposited into the bottom sediment (Baxter 1977, Bushaw-Newton et al. 2002). The sediment depleted waters from the impoundment cascade over the dam spillway which increases erosion and scour of stream beds and banks downstream of the dam (Pejchar and Warner 2001, Fencl et al. 2015). The erosion and scour downstream of the dam preferentially removes fine sediments which creates armored stream banks and beds (Fencl et al., 2015). Collectively these impacts affect biota by disrupting dispersal and migration, isolating populations, reducing genetic diversity, shifting species composition from lotic to lentic within the impoundment, and reducing habitat quality (Lejon et al. 2009, Evans et al. 2000, Dorobek et al. 2015, Poff and Hart 2002). Over time, sedimentation can result in the filling of the impoundment, and the impoundment may begin to act as a source of sediment to downstream reaches (Stanley and Doyle 2002, Ahearn and Dahlgren 2005). This accumulation of sediment reduces the functional lifespan of the dam and increases the likelihood of dam failure (Stanley and Doyle, 2003).

*Dams in the United States*

The National Inventory of Dams (NID) identifies 90,580 dams within the United States (US ACE, 2017). The Army Corps of Engineers acknowledges that the NID underestimates the total number of dams because the database only includes large or high hazard dams (US ACE, 2017). Other estimates have identified over two million dams <7.6 meters (m) in height in the United States (Graph, 1993). Many of these dams no longer serve their intended purpose and are nearing the end of their operational lives (Grant 2001, Evans et al. 2007).

*Dam Removal*

The aging of dam infrastructure coupled with the costs associated with restoration and maintenance, the ecologic impacts of dams, and increased regulation has resulted in dam removal becoming a viable dam management alternative (Doyle et al. 2003, Bellmore et al. 2016, Pejchar L. and Warner 2001, Stanley 2002). This has led to over 1,100 dam removals in the United States, most of which have occurred within the past 20 years (Bellmore et al., 2016).

Of the over 1,110 dam removals conducted in the United States, only 139 have had any post removal monitoring and most of these monitoring programs were short term (1-2 years) (Magilligan 2016, Bellmore et al. 2016). The large variety of site specific dam characteristics, watershed sizes, and river types in combination with limited post removal research makes quantitative predictions of the impacts of dam removal extremely difficult (Bushaw-Newton et al., 2002). Thus, most of the information on the impact of dam removal is largely conceptual (Gregory et al., 2002).

Dam removal typically results in substantial positive long term impacts to a watershed; however, most removals also result in some negative impacts (Hart et al., 2002). Anticipating the potential negative effects prior to dam removal and incorporating methods to minimize these risks into the removal strategy will increase the likelihood of a successful project. The first step to anticipating the negative effects should be identifying unintended benefits of the dam, because removal of a dam will likely also result in a removal of these benefits.

**Stream Crossing Surveys**

Built up areas typically have a large quantity of stream crossings (Diebel et al. 2014). Stream crossings such as bridges and culverts can act as continuous or periodic barriers to fish passage that collectively can have a more substantial influence on longitudinal stream connectivity than dams (Diebel et al. 2014). The impact of a stream crossing on fish passage is largely determined by the structure's design (Perkin and Gido 2012). Factors that affect fish passage at stream crossings include high water velocities, turbulent flows, shallow water depths, and inlet and outlet drop height (Perkin and Gido 2012, Diebel et al. 2014). Stream crossings surveys assess the physical characteristics of crossings to determine the likelihood that they are a barrier to fish passage. Assessed characteristics can include crossing type, structure shape, structure material, condition of structure, flow conditions, substrate in the structure, and perched height (USFS et al. 2011). Collectively this information is interpreted using published research and professional judgment to determine the likelihood that a stream crossing is a barrier to fish passage.

**Invasive Species**

While dams negatively impact native fish populations, they may prevent or slow the spread of invasive species (McLaughlin et al. 2013). Invasive species are organisms that are not native to the ecosystem they occupy. The introduction of invasive species occurs when organisms are moved to new ranges where they proliferate, spread, and persist (Mack et al. 2000). The occurrence of invasive species is not an exclusively human induced phenomenon; however, the rate of introduction has substantially increased due to human activity (Mack et al. 2000). The spread of invasive species by humans has resulted in most habitats on earth being impacted by invasive species, and the aggregate effect of these introductions has substantially impacted global biodiversity (Mack et al. 2000). Invasive species can substantially alter the fundamental properties of the ecosystems they occupy such as the dominant species in a community, nutrient cycling, and plant productivity (Mack et al. 2000). Invasive species may negatively impact native species by competing for resources and habitat, or by direct predation (Mack et al. 2000). The adverse consequences of the establishment of invasive species substantially varies between species and environments and is difficult to predict because species can act unpredictably outside of their native range (Mack et al. 2000, Kornis et al. 2013).

One invasive species that may be of concern in the Oak Creek watershed is the round goby. The round goby is a euryhaline, aggressive fish, with an extended reproductive period. The round goby is native to the Black and Caspian seas and was introduced to the Laurentian Great

Lakes via the discharge of ballast water from ships (Jude 2001, Kornis et al. 2013). Within the Great Lakes, the round goby is associated with declines in fish and invertebrate abundance (Balshine et al. 2005, Lederer et al. 2008). Since its introduction, the round goby has extended its range to multiple Great Lakes tributaries (Kornis et al. 2013). The establishment of the round goby in tributary streams is dependent on a variety of habitat features. Round goby success in Great Lake tributaries is positively associated with large watershed areas and stream sizes, and negatively associated with low concentrations of dissolved oxygen and ions, cold stream temperatures, high flow velocities, and high stream slopes (Kornis and Vander Zanden 2010, Krakowiak and Pennuto 2008, and Baldwin et al. 2012).

Another invasive species that may be of concern in the Oak Creek watershed is the sea lamprey. The sea lamprey is a parasitic fish that targets multiple large fish species including lake trout and lake whitefish (Cuhel and Aguilar 2012). Sea lamprey start their life cycle in a larval stage that lasts four to eight years (Coble et al. 1990). During this stage sea lampreys reside in soft, fine, sandy sediments with high amounts of organic substrate, in shaded areas of streams with low flow velocities (Cuhel and Aguilar 2012, Potter et al. 1986, Neeson 2010). The larvae are mostly sedentary and feed on unicellular algae, detritus, and microorganisms from overlying water and substrate surface (Potter et al. 1986). At the end of the larval stage sea lampreys metamorphize into a parasitic juvenile stage which lasts twelve to twenty months in freshwater environments (Potter et al. 1986, Neeson 2010). During the juvenile stage sea lampreys migrate into large bodies of water and attach to large fish, to suck blood and bodily fluids, which often results in the death of the host fish (Cuhel and Aguilar 2012). At the end of the juvenile stage, adult sea lampreys migrate up tributaries where they spawn in gravel nests in areas of steady unidirectional flow volumes, then die (Cuhel and Aguilar 2012, Neeson 2010). The sea lamprey became established to the Great Lakes by slowly migrating from the Atlantic Ocean through the St. Lawrence River after the Welland Canal was created to bypass the Niagara Falls (Cuhel and Aguilar 2012). The establishment of sea lamprey in the Great Lakes led to a rapid and severe decline in lake trout populations in the upper Great Lakes during the 1940s-1950's (Lawrie 1970, Smith and Tibbles 1980, Coble et al. 1990). The success of the sea lamprey opened an ecological niche that resulted in the proliferation of the alewife, another invasive species (Cuhel and Aguilar 2012). Since 1965, salmon and trout have been stocked in Lake Michigan to control alewife populations and provide recreational fishing opportunities (Tody and Tanner 1966). To control sea lamprey populations, the Great Lakes Fishery Commission has been applying lampricides to streams since 1958 (Jude et al. 2005). Additionally, as of 2009, 61 physical barriers were installed in tributary streams to the Great Lakes to prevent sea lampreys access to suitable spawning areas (Pratt et al. 2009).

**Assessment of Water Quality**

While dams negatively affect water quality by altering flow, sediment, and thermal regimes, the settling of particles and associated contaminates within the impoundment may improve some aspects of downstream water quality (Baxter 1977, Bushaw-Newton et al. 2002, Syvitski et al. 2005, Ward and Stanford 1995). Monitoring of multiple water quality parameters is necessary to understand how an individual dam impacts water quality. Access to equipment and reagents, staff availability and expertise, and available funds should be considered when selecting parameters (Deborah 1992). The parameters assessed during this study were precipitation, discharge rate, water temperature, total suspended solids, turbidity, specific conductivity, pH, total phosphorus, nitrogen, dissolved oxygen, biological oxygen demand, *chlorophyll-a*, and *E. coli*. Descriptions of the assessed parameters and justification for selection are as follows:

*Precipitation*

Precipitation events cause stormwater runoff which may mobilize contaminates including sediments, nutrients, and fecal pollution (Hauer and Lamberti 2006, DeBarry 2004). The pollutants in stormwater runoff can contribute to the degradation of instream water quality. Precipitation events also increase river discharge and flooding potential. The effects of precipitation are most pronounced in rivers which experience rapid peak discharges. Precipitation data was collected for this study because it is easily accessible and because precipitation and the associated stormwater runoff has a substantial influence on water quality, especially in areas with urban development (Dwight et al. 2011).

*Discharge Rate*

Discharge rate is a measure of the volume of water that passes a cross sectional area over a specified time interval and is commonly measured in cubic feet per second. The discharge rate of a stream can impact sediment erosion and transport (MST 2013). High flow rates can result in erosion of stream bank and bottom sediment, and low flow rates can result in the deposition of suspended sediments and debris (MST 2013). The flow regime of a stream impacts habitat types, channel shape, and bottom sediment composition within the stream (MST 2013). Development of vegetated areas into bare soil or impervious surfaces increases stormwater runoff volume and velocities (Dunne and Leopold 1978). This creates flashy flow conditions with increased risk of flooding after precipitations events (Hollis, 1975). Discharge rate data was collected because it is easily accessible and it can substantially effect the assimilation and transport of pollutants within a waterbody (Deborah 1992).

*Water Temperature*

Water temperature is a measure of heat energy within a body of water and is strongly affected by human activities within a watershed (APHA et al. 2005, Webb 1996). Water temperature is influenced by many factors including net solar radiation, evaporation and condensation, sensible heat transfer between air and water, streambed conduction, friction, advective transfer in precipitation, ground and surface water input temperatures, elevation, stream shading, presence of impoundments, and channel morphology (Webb 1996, Hawkins et al. 1997, Dick et al. 2015). Multiple physical characteristics of water change with water temperatures including: vapor pressure, surface tension, density, viscosity, gas solubility, and chemical reaction rates (Webb, 1996). Water temperature also strongly influences biological oxygen demands (U.S. EPA 2006). Additionally, multiple ecological processes are strongly influenced by water temperature including: geographic distribution, growth, metabolism, reproduction, and disease resistance of organisms (Webb 1996, Dick et al. 2015). Water temperature is an important parameter to assess because it influences multiple chemical, physical, and biological processes within waterbodies (Deborah 1992). The Wisconsin Department of Natural Resources (WI DNR) state standards for surface water temperature based on the designated use of Fish and Aquatic Life (FAL) are established in NR 102.04 4(e). The WI DNR's general water quality criteria for temperature states that there can be no temperature changes that adversely affect aquatic life (NR 102.24 (1)) and that natural daily and seasonal temperature fluctuations must be maintained (NR 102.24 (2)). To achieve this, maximum water temperatures are set for acute and sub-lethal effects for each calendar month. Acute criteria are applied as daily maximum temperatures (NR 102.25 (1)(c)) and sub-lethal criteria are applied as weekly average maximum temperatures (NR 102.25 (1)(b)).

*Total Suspended Solids*

Total Suspended Solids (TSS) is a measurement of the dry weight of all solid materials larger than two microns in a known volume of water. TSS is commonly measured in milligrams per liter (mg/l). In stream sedimentation and siltation are the most common stressors of streams within the United States (U.S. EPA, 1998). Streambank erosion is a major source of TSS to rivers (Robertson et al., 2006). The primary transport mechanism for sediments into streams is stormwater runoff (Robertson et al., 2006). Suspended sediments reduce water clarity and sediment deposition can bury and suffocate fish eggs and aquatic insects (Robertson et al., 2006). Additionally, multiple compounds including nutrients, heavy metals, and bacteria are often attached to suspended sediments, resulting in additional impairment (Robertson et al. 2006, Herngren et al. 2005, Dong et al. 1984, Pandey and Soupir 2013). Collection of TSS data is important because it effects physical aspects of water quality and sediment transport is a major mechanism for pollution transport within waterbodies (Deborah 1992). There are no WI DNR state standards or recommended guidelines for TSS in surface water; however, the United States Geological Survey (USGS) determined a recommended guideline of 19 mg/l for streams and rivers in Ecoregion V (Robertson et al., 2006).

*Turbidity*

Turbidity is a measure of the scatter of light through a water column due to suspended particles and is commonly expressed as Nephelometric Turbidity Units (NTU) (APHA et al., 2005). A variety of factors influence turbidity including stormwater runoff, stream bank erosion, and streambed sediment resuspension and/or loading. High sediment loads reduce light reaching the stream bed, raises water temperatures due to the particles absorbing and retaining heat, and contributes to pollutant loading via particle attachment. Turbidity data was collected in this study because it is easy to assess, it can be used as a proxy for suspended sediment, and it affects plant and algal growth within a waterbody (Deborah 1992). There are no WI DNR state standards or recommended guidelines for turbidity in surface water; however, the United States Environmental Protection Agency (U.S. EPA) has established a recommended guideline value of less than 14 NTU for rivers in Ecoregion VI (U.S. EPA, 2000).

*Specific Conductivity*

Specific conductivity is the measure of a substance's ability to conduct an electric current standardized at 25°C. It is commonly expressed in micro-Siemens per centimeter (µS/cm) (Hem, 1985). Specific conductivity is directly proportional to the amount of dissolved ions in the solution being measured (Hem, 1985). The specific conductivity of instream water is influenced by the specific conductivity of the streams' source water, local geology, evaporation, and anthropogenic activity within the watershed (U.S. EPA, 2012a). Anthropogenic sources that impact specific conductivity concentrations include sanitary sewer leaks, industrial discharge, and agricultural runoff (Brown et al. 2004, U.S. EPA 2012a). One major cause of higher specific conductivity in the northern United States is the use of road salt as a deicing agent (Corsi et al., 2010). Elevated chloride concentrations (as shown by elevated specific conductivity) result in immediate and long term impacts to surface water quality and aquatic life (Corsi et al.,2010). Specific conductivity was assessed in this study because it is a rough indicator of mineral content, it can identify pollution zones (ex. areas effected by effluent discharge), and it can be used as an estimate of the extent of stormwater runoff impacts (Deborah 1992). There are no WI

DNR state standards or recommended guidelines for specific conductivity; however, the U.S. EPA recommends a range between 50 - 1500 μS/cm to support fish and other aquatic life (U.S. EPA, 2012a).

*pH*

pH is the negative $\log_{10}$ of hydrogen ion activity in moles per liter and is measured as standard units (s.u.) (Hem, 1985). The pH of streams is influenced by natural conditions (substrate or geology), source water pH (e.g. surface runoff or groundwater), anthropogenic activity (e.g. industrial discharge or agricultural runoff), and plants and animals (e.g. photosynthesis or respiration) (U.S. EPA, 2012a, U.S. EPA 2006). Most chemical and biological processes within rivers and streams are affected by changes in pH (U.S. EPA, 2012a). Most aquatic organisms prefer water with pH ranges within 6.5 - 8.0 s.u. because pH values outside of this range often result in physiological stress (U.S. EPA, 2012a). This study assessed pH because it influences multiple chemical and biological processes within waterbodies (Deborah 1992). WI DNR surface water standards require pH values to remain between 6 - 9 s.u., with no changes of more than 0.5 s.u. outside the estimated natural seasonal maximum and minimum values (NR 102.04 (4)(c)). The WI DNR further requires a waterbody with at least 10 percent or more of at least 10 samples from a continuous sampling period outside of the required criteria be listed as impaired (Clayton et al., 2012).

*Total Phosphorus*

Total phosphorus is a measurement of the dissolved and suspended phosphorus in a water column. It is commonly reported in mg/l (U.S. EPA, 2012a). Phosphorus is an important nutrient for life; however, excess amounts can result in detrimental impacts to water quality and aquatic biota including eutrophication, the proliferation of harmful algal blooms, and unsafe swimming and recreational conditions (Litke 1999, WI DNR 2012). Minor amounts of phosphorus naturally occur in waterways due to the weathering of soil and rocks, and the decomposition of organic material; however, in some streams, most phosphorus originates from anthropogenic sources such as wastewater treatment effluent, fertilizers, industrial cleaners, runoff from manure storage, and vehicle exhaust (WI DNR 2011, Litke 1999, U.S. EPA 2012a). Total phosphorus was assessed in this study because it is typically the limiting nutrient for algal growth, high concentrations can indicate the presence of a pollutions source, and it contributes to the eutrophication of waterbodies (Deborah 1992). Wisconsin was one of the first states in the United States to adopt state standards requiring numeric criteria for total phosphorus in rivers and streams (WI DNR, 2012). To protect surface water designated as FAL, a total phosphorus criterion of 75 μg/l was established (NR 102.06 (3) (b)). In determining impaired waters, the WI DNR requires six monthly samples from May through October, in which the lower 95% confidence interval of the sample population should not exceed the established threshold (Clayton et al., 2012).

*Nitrogen*

Nitrogen is naturally occurring in waterways; however, anthropogenic sources often contribute to increased concentrations. Excess concentrations of nitrogen can contribute to eutrophication (when sufficient phosphorus is present), and influence water temperatures and dissolved oxygen (DO) concentrations. Nitrogenous compounds are an important metric for water quality as they more readily dissolve in water compared to phosphates. Common sources of nitrogen include wastewater treatment effluent, runoff from fertilized lawns/agricultural fields, and industrial discharge with corrosion inhibitors (U.S. EPA, 2012a). Elevated concentrations indicate potential contributions of human sewage or manure. Nitrogen can be present in multiple forms in water including nitrate, nitrite, ammonium, ammonia, and organic nitrogen (U.S. EPA 2006). Nitrogen was assessed in this study because it influences algal growth, it gives a general indication of nutrient status and organic pollution levels, and high concentrations of select forms of nitrogen can be harmful to humans and aquatic life (Deborah 1992, U.S. EPA 2006). High concentrations of nitrate in drinking water can be harmful to human health and high concentrations of ammonia can be toxic to aquatic life (Walton 1951, U.S. EPA 2006). The WI DNR is currently developing nitrogen criteria for surface waters, but doesn't have sufficient data to create scientifically defensible standards and will continue to analyze data as it becomes available (WI DNR, 2015). The U.S. EPA recommends that inorganic nitrogen (nitrates and nitrites) concentrations be less than 1.798 mg/l and total Kjeldahl nitrogen (TKN) concentrations be less than 0.663 mg/L (U.S. EPA, 2000).

*Dissolved Oxygen*

DO is a measurement of oxygen gas incorporated into water. It is commonly expressed as both concentration (mg/l) and percent saturation (% sat) (Hem, 1985). Oxygen is incorporated into water through atmospheric diffusion or through aquatic vegetation as a byproduct of photosynthesis (U.S. EPA, 2012a). The release of oxygen due to photosynthesis and the influence of temperature on oxygen solubility result in fluctuations of dissolved oxygen concentrations on daily and seasonal scales (U.S. EPA, 2012a). Oxygen is also consumed by the oxidation of organic waste, biological respiration, and biological decomposition. Low DO levels result in inhospitable conditions for many types of aquatic life (Zimmerman, 1993). Dissolved oxygen levels are influenced by water temperature (oxygen solubility decreases as water temperature increases), atmospheric pressure (oxygen solubility increases as pressure increases), and salinity (oxygen solubility decreases when salinity decreases) (Hem, 1985). Dissolved oxygen is a fundamental parameter for water quality assessments because it is involved in or influences nearly all chemical and biological processes within waterbodies (Deborah 1992). WI DNR surface water state standards for warm water FAL use designation states that DO levels must remain above five mg/l (NR 102.04 (4)(a)), except waters specifically designated by the WI DNR, which need to remain above two mg/l (NR 104.06(2) a.3).

*Biological Oxygen Demand*

Biological oxygen demand is a measurement of the oxygen consumed in water by both chemical and biological processes (U.S. EPA, 2006). The standard test period for biological oxygen demand is five days and it is typically reported as BOD-5 day (BOD) in mg/l. The BOD of a waterway is influenced by contributions from multiple sources including: leaves, woody debris, dead plants and animals, animal waste, pulp and paper mill effluents, wastewater treatment plant effluents, failing septic systems, airport runoff, and urban stormwater runoff, among other things (Corsi et al. 2001, U.S. EPA 2006). The rate of oxygen consumption due to these sources is affected by multiple variables including water temperature and microbial community composition and concentration (U.S. EPA, 2006). BOD was assessed in this study because it directly affects DO concentrations, with higher BOD resulting in lower DO levels (U.S. EPA, 2006). Some waterways are naturally organically rich and have high BOD; however, elevated BOD levels are generally indicative of polluted or eutrophic waters (U.S. EPA, 2006).

*Chlorophyll-a*

*Chlorophyll-a* is the most common photosynthetic pigment found in plants, algae, and cyanobacteria (KDHE 2011). *Chlorophyll-a* is frequently used to estimate the primary productivity and trophic status of a water body (KDHE 2011). *Chlorophyll-a* was used in this study instead of direct measures and counts of algal communities because they are prohibitively expensive and time consuming (KDHE 2011). There are no WI DNR state standards or recommended guidelines for *chlorophyll-a*; however, the State of Kansas recommends considering waterbodies with concentrations greater than 8 μg/L as productive (KDHE 2011).

*E. coli*

There are currently over 150 known waterborne pathogens, including bacteria, viruses and protozoa (McLellan et al., 2013). These microorganisms originate from a variety of sources and are delivered to receiving waters via mechanisms of transport such as wastewater treatment plant effluent, agricultural runoff, landscape run-off (frequently containing animal/wildlife feces), failing sanitary sewer infrastructure, septic system overflows, and urban stormwater runoff (Mallin et al. 2000, Burzynski and Helker 2002, DeBarry 2004). Testing of all potential waterborne pathogens would be prohibitively difficult, time consuming, and expensive. Therefore, fecal indicator bacteria (FIB) are utilized as a proxy to measure risk of illness due to waterborne pathogens present in the aquatic environment. *Escherichia coli* (*E. coli*), a member of the fecal coliform group, is a commonly utilized FIB because it is present in the gut of warm blooded animals, elevated *E. coli* concentrations are correlated with increased risk of human illness, it is easily cultured and enumerated, and testing is relatively inexpensive (McLellan et al. 2013, US EPA 1986, Hammer and Hammer, Jr. 2008). Multiple studies have demonstrated that *E. coli* can persist, and at times proliferate, within the environment due to seasonal variability and environmental influences, suggesting there are limitations that must be considered when utilizing *E. coli* including the inability to directly identify the origin or conveyance mechanism responsible for elevated concentrations (McLellan et al. 2013, Whitman et al. 2006, Burzynski and Helker 2002, Izbicki, et al. 2009, Lawrence 2012, Field 2008). Thus, the implications of elevated *E. coli* concentrations should be taken in context with other assessed environmental parameters. *E. coli* is an essential parameter to assess in any waterbody where direct or indirect human consumption of the water may occur (Deborah 1992). Wisconsin surface water standards have historically utilized total and fecal coliforms rather than *E. coli*. Fecal coliform standards still pertain to all surface waters of Wisconsin; however, Great Lakes tributaries can also apply an *E. coli* standard, i.e. "The Great Lakes system includes all the surface waters within the drainage basin of the Great Lakes" (WI DNR 2010). Additionally, the U.S. EPA introduced new recreational water quality criteria for enterococci and E. coli in the 2012 Recreational Water Quality Criteria document and implemented these water quality criteria in the 2014 National Beach Guidance and Required Performance Criteria for Grants document (U.S. EPA 2012 b, US EPA 2014). The WI DNR has not yet adopted these standards; however, the department is currently revising the state's water quality standards and is proposing to change bacteria water quality criteria from fecal coliform to *E. coli* as recommended in the US EPA 2012 revised water quality criteria (WI DNR 2016 a). Subsequently, this study will utilize the US EPA 2012 revised water quality criteria for an estimated illness rate of 36 for 1000 primary contact recreators. These standards state that the geometric mean of *E. coli* in the waterbody should not be greater than 126 colony forming units per 100 milliliters (cfu/100 ml) in any 30 day interval and that *E. coli* in the waterbody should not exceed a statistical threshold value (STV) of 410 cfu/100 ml in greater than ten percent of samples in the same 30 day interval (U.S. EPA 2012b). The standards also recommend utilizing the same water quality standards based on the same illness rate in upstream waters that are used in downstream waters and utilizing a beach action value of 235 cfu/100 ml as a conservative precautionary tool for making beach notification decisions.

**Dams Impact on Water Quality**

Dams can have multiple impacts on water quality (Kurunc et al. 2005). Dam impoundments slow flow and increase the residence time (Kurunc et al. 2005). The longer residence time allows particles and particle associated contaminates from the inflow to settle to the bottom of the impoundment, causing a decrease in organic and inorganic nitrogen and phosphorus, dissolved solids, suspended sediments, turbidity, and *E. coli* (Hirsch et al. 1991, Morris and Fan 1998, Stow et al. 2001, Stanley and Doyle 2002, Baxter 1977). The longer residence time also increases waters exposure to solar radiation within the impoundment, resulting in increased water temperatures (wei et al. 2008). The increased water temperatures can reduce dissolved oxygen within the impoundment (wei et al. 2008). Overtime phosphorus deposited in lake bed sediments can result in eutrophication of the water body (Bayram et al. 2012). The eutrophication of water results in increased pH and decreased dissolved oxygen levels within the impoundment (Zakova et al. 1993, Bayram et al. 2012).

## 2. Introduction

Dams are built for multiple purposes including flood control, irrigation, water supply, and recreation (Bushaw-Newton et al. 2002, Baxter 1977). Despite these benefits, dams degrade rivers by fragmenting channels and altering sediment, thermal, and flow regimes (Baxter 1977, Syvitski et al. 2005, Ward and Stanford 1995). Estimates for the number of dams in the United States vary from 90,580 to over 2 million (US ACE 2017, Graph, 1993). Many of these dams no longer serve their intended purpose and are nearing the end of their operational lives (Grant 2001, Evans et al. 2000). The aging of dam infrastructure coupled with the cost of restoration and maintenance, regulation, and the ecologic impact of dams has resulted in removal becoming a viable management alternative (Bellmore et al. 2016, Pejchar and Warner, 2001). This has led to over 1,100 dam removals in the United States, most of which have occurred in the past 20 years (Bellmore et al., 2016). Despite increased utilization, limited research and a lack of quantitative predictive capacity results in large amounts of uncertainty associated with the site specific impacts of dam removal (Bushaw-Newton et al. 2002). Additionally, dams which no longer serve their intended purpose may still have unintended positive benefits such as the prevention of the spread of invasive species, providing recreational opportunities, and the treatment of water pollution from upstream reaches. Failure to identify unintended benefits of dams can prevent accurate determinations of the potential consequences of removal. Therefore, identifying unintentional benefits of a dam is an essential step in determining the potential impacts of dam removal.

The Mill Pond Dam in the Oak Creek Watershed is being considered for removal; however, it may serve as a barrier to invasive species from Lake Michigan, such as the round goby and the sea lamprey (McLaughlin et al. 2013). Additionally, the impoundment has historically been utilized as a community gathering place for recreational activities and local citizen groups have formed to restore recreational opportunities such as boating and swimming within the impoundment. The dam is also located near the confluence of Oak Creek and Lake Michigan. Previous research indicates that the Oak Creek negatively impacts water quality at Grant Park beach, which is impaired due to high *E. coli* concentrations (Koski and Kinzelman, 2013). The dam could provide the unintended benefit of improving water quality in Oak Creek before discharging into Lake Michigan. If the dam acts as a major barrier to invasive species, decision makers would have to determine if the positive impacts of dam removal on desirable fish populations would outweigh the potential negative impacts of the establishment of invasive species populations in upstream reaches. Determining the suitability of the impoundment for recreational activities will inform discussions about dam management decisions between decision makers and the community. If the dam improves downstream water quality, decision makers would have to determine if the positive impacts of dam removal would outweigh the negative impacts to nearshore water quality in Lake Michigan.

The objective of this study was to determine if the Mill Pond Dam provides the unintended benefits of acting as a barrier to invasive species, creating recreational opportunities, and improving downstream water quality. Stream crossing surveys were utilized to determine if the Mill Pond Dam provides protection against the potential habitation of invasive species to upstream areas. Creating a bathymetric map and comparing *E. coli* values to recreational contact standards evaluated the impoundments suitability for recreation. The suite of parameters utilized to assess water quality in this study characterized the dam's impact on physical, chemical, and biological aspects of water quality. We hypothesized that the Mill Pond Dam provides the unintended benefit of acting as a barrier to invasive species, and that the Dam does not provide the unintended benefits of providing recreational uses or improving water quality due to the degraded conditions within the impoundment. Specifically, we hypothesize that the dam is a barrier to invasive species and that removal of the dam would increase the risk of invasive species habitation to a large portion of upstream areas, the dam's impoundment isn't suitable for recreational contact, and the dam doesn't improve downstream water quality. The methods utilized in this study are transferable and could be used at other locations to improve the understanding of the potential consequences of dam removal.

## 3. Methods

Surveys were conducted at all stream crossings within the watershed to identify substantial barriers to fish passage. A bathymetric map was created to assess the impoundment's suitability for recreation. Various environmental, physical, chemical, and biological indicators were assessed upstream, within, and downstream of the Mill Pond Dam impoundment at the study sites. Assessed parameters included antecedent precipitation, discharge rate, water temperature (WT), TSS, turbidity, specific conductivity, pH, total phosphorus (TP), nitrate + nitrite-nitrogen ($NO_2^-$ + $NO_3^-$), TKN, total nitrogen (TN), DO (concentration and percent saturation), BOD, *chlorophyll-a,* and *E. coli*.

**Study Area and Sampling Sites**

This study was conducted on the Oak Creek Watershed in Southeastern Wisconsin. Oak Creek is comprised of two major tributaries (Mitchell Field Drainage Ditch and The North Branch) and a 22.2 kilometer (km) mainstem, which flows in a northeastern direction and discharges into Lake Michigan adjacent to the southern end of Grant Park Beach (SEWRPC 2007, SEWRPC 2015). As of 2014, the Oak Creek main-stem appears on the Wisconsin WI DNR list of impaired surface waters (Clean Water Act, Section 303(d)) for chloride, total phosphorus, and unknown pollutants (WI DNR 2016 b). Additionally, Grant Park beach is currently 303(d) listed as impaired due to excessive beach closures and advisories. Previous research, conducted by the Racine Health Department (RHD), indicates that Oak Creek negatively influences nearshore surface water quality at Grant Park Beach (WI DNR 2016 c, Koski and Kinzelman 2013).

The Mill Pond Dam and associated impoundment are located approximately 1.6 km upstream of the confluence of Oak Creek and Lake Michigan. A wooden dam was originally built in the 1840's to serve as an energy source for a mill (Friends of the Mill Pond and Oak

Creek Watercourse Inc. 2015). This was replaced by a 5.4 m high and 10.4 m wide dolomite dam in the mid 1930's (Friends of the Mill Pond and Oak Creek Watercourse Inc. 2015). A WI DNR safety report rated the dam conditionally fair, indicating that the dam and impoundment need maintenance (Lourigan 2012). Several community organizations are advocating for restoration of the impoundment to reestablish recreational benefits of the dam while creating a community gathering space. Additionally, the Southeastern Wisconsin Regional Planning Commission (SEWRPC) is conducting a watershed wide study is evaluating restoration options for the dam including repair or removal of the dam.

The study location and surface water collection site locations are summarized in Figure 1. Four locations were sampled once weekly for 61 weeks (06/29/2015 – 08/29/2016). Oak Creek Parkway (OCP), located directly upstream of Mill Pond, was selected to be representative of stream water quality before the influence of the impoundment. Mill Pond (MP), located within the impoundment near the outflow, was selected to be representative of water quality within the impoundment. The Falls, located 10 m downstream of the dam, was selected to assess the impoundments impact on water quality immediately downstream of the dam. Oak Creek Mouth (OCM), located at the confluence of Oak Creek and Lake Michigan, was selected to assess the impact of the impoundment on water quality further downstream. After a period of sampling, two additional surface water collection sites were added to facilitate a more accurate characterization of the impoundments' impact on water quality. Specifically, MP-FLOW was added to capture differences in water quality within the impoundment (07/06/2015 – 08/29/2016), and Hawthorne (HAW) was added because Lake Michigan mixing at OCM made assessment of downstream impacts difficult (11/23/2015 – 08/29/2016). Collectively, these sites (n=6) are referred to as "study sites".

Monthly samples were collected at the study sites (n=6), and analyzed for TP, $NO_2^-$ + $NO_3^-$, TKN, and total nitrogen (TN). Stratified weekly samples were collected near the inflow of the impoundment and analyzed for *chlorophyll-a* at 8 centimeters (cm) (MP1-S) and 1.2 m (MP1-D) below the water surface from 07/29/15 – 08/18/15 (Figure 2). BOD data was collected monthly at Nicholson Ave. (Nich Ave.), approximately six river km upstream of OCP (Figure 2). Samples analyzed for BOD were collected on the same day and at a similar time as samples collected for the study sites. BOD data was utilized to estimate the nightly decline in DO within the Mill Pond impoundment. Collectively, Nich Ave., MP1-S, and MP1-D are referred to as "supplemental sampling sites" as shown in Figure 2.

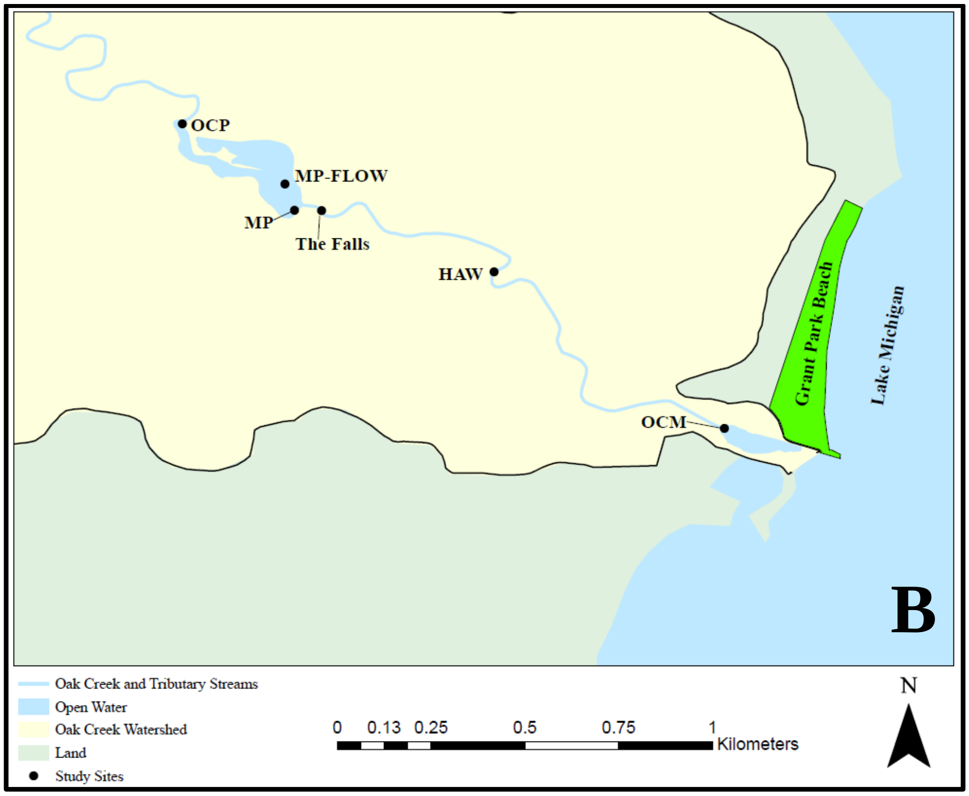


**Figure 1.** A. Map of study location. B. Map of study site locations

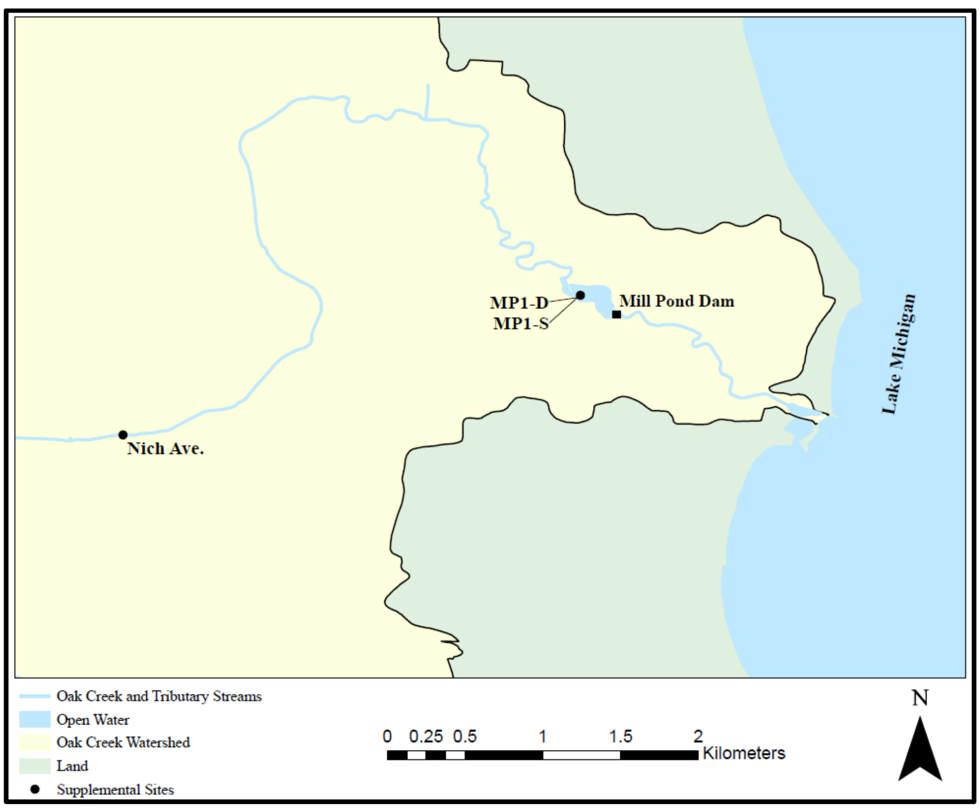


**Figure 2.** Map of supplemental sampling sites.

**Stream Crossing Survey**

Stream crossing surveys were conducted and analyzed SEWRPC at all bridge and culvert crossings on Oak Creek. Data from each crossing was recorded on a stream crossing data sheet. Determinations of the barriers to fish passage were made using professional judgment based on physical measurements, overall observed conditions, and criteria set forth in Diebel et. al 2014.

**Bathymetric Map**

A bathymetric map of the Mill Pond Dam impoundment was created to assess the impoundments suitability for canoeing and kayaking. A rope, weighted by a brick, marked in 0.3 m increments was utilized to measure water depth. Depth measurements were performed in a grid with a 15 m spatial resolution. This distance was reduced when substantial depth changes occurred between measurement locations. Measurement locations were recorded using a Garmin Model 72 GPS (Olathe, KS, USA). The bathymetric map was created with Surfer 12 (Golden Software, Golden, CO, USA), using the Kriging gridding method. The impoundment surface area and volume were calculated utilizing Surfers' gridding report. Results were compared to a previous bathymetric survey conducted at Mill Pond in 1970 to determine changes in water depth, volume, and surface area.

**Sample Collection**

Replicate grab samples were collected at each study site using 532 and 1242 milliliter sterile Whirl-Pak® bags and sampling bottles provided by University of Wisconsin Oshkosh's Environmental Research and Innovation Center (ERIC) (Nasco, Fort Atkinson, WI, USA). The 532 ml replicate was used to analyze *E. coli*, specific conductivity pH, and turbidity. The 1242 ml field replicate was utilized to conduct TSS analysis. The bottles provided by ERIC were utilized to conduct TP, TKN, $NO_2^-$ + $NO_3^-$, TN, and BOD analysis. Surface water samples were collected from 10 cm below the water surface whenever possible. If water levels were prohibitively low and would not allow for the sample to be taken from 10 cm below the water surface, the sample was collected from the deepest area available. Care was taken to ensure bottom sediments were not suspended during the sample collection process. After sample collection, the Whirl-Pak® bags and sampling bottles were sealed and placed in a cooler on ice packs to maintain a temperature of 4° C during transport. At the laboratory, samples in the Whirl-Pak® bags were stored in a refrigerator at a temperature of 0 - 4.5 ° C until they were analyzed and the sampling bottles were shipped on ice overnight to ERIC. All laboratory analysis was conducted within the recommended/required holding time for each of the parameters evaluated.

**Parameter Measurements**

Water temperature and dissolved oxygen were measured in situ using a Yellow Springs Instrument (YSI, Yellow Springs, OH, USA) 550a (06/69/2015 – 07/20/2015) or YSI Professional-Plus (07/27/2015 – 08/29/2016). Standard Methods 2540 was utilized to measure TSS (Eaton et al., 1998). Turbidity was determined utilizing a Micro 100 Turbidimeter (HF Scientific, Inc. Fort Myers, FL, USA). Specific conductivity and pH were analyzed in the lab from 06/69/2015 to 07/20/2015 using an Oakton Con 510 conductivity meter (Oakton Instruments, Vernon Hills, IL, USA) and Corning pH meter 430 (Corning Incorporated, Corning, NY, USA) respectively. Specific conductivity and pH were measured in situ from 7/27/2015 - 8/29/2016 with a YSI Professional-Plus. Sample analysis for TP (EPA Method 365.1), $NO_2^-$ + $NO_3^-$ (APHA Method SM 4500-F), TKN (EPA Method 351.2), BOD-5 day (standard method 5210b), and *chlorophyll-a* (standard method 10200H) was performed by the University of

Wisconsin-Oshkosh Environmental Research and Innovation Center (WDNR Laboratory ID No. 471183460, EPA ID No. WI01087). TN was calculated by adding $NO_2^-$ + $NO_3^-$ and TKN concentrations. *E. coli* was quantified utilizing IDEXX Colilert - 18® (IDEXX Inc., Westbrook, ME, USA).

**Meteorological Data**

Precipitation data was accessed online from the National Oceanic and Atmospheric Administration's KMKE weather station, approximately 5.6 km northwest of the Mill Pond Dam at the General Mitchell International Airport in Milwaukee, Wisconsin (42°57′ N, 87°54′ W) (http://w1.weather.gov/data/obhistory/KMKE.html).

**Discharge Rate**

Stream discharge data was accessed online from the United States Geological Survey (USGS) Oak Creek at South Milwaukee, WI gaging station (#04087204) located approximately 2.9 river km upstream of the Mill Pond Dam at 42°55′ 30″ N, 87°52′12″ W (https://waterdata.usgs.gov/nw is/uv?site_no=04087204).

**Impoundment Residence Time**

The impoundment residence time was calculated using Equation 1.

$$\text{Equation 1: } R = \frac{V}{I} \qquad (1)$$

Where R is the residence time in hours, V is volume in cubic meters ($m^3$), and I is inflow in cubic meters per hour $\left(\frac{m^3}{h}\right)$. The inflow used in residence time determinations was the median discharge rate at the USGS gaging station for the study period. A previous study conducted by the Racine Health Department Laboratory indicates that the northern and southern portions of the impoundment are mostly stagnant, and that water entering the impoundment flows through the center of the impoundment in a limited channel area when the impoundment is not influenced by precipitation (Turner et al. 2017). To account for the observed circulation within the impoundment, two residence times were calculated. For the first residence time determination, the volume of the entire basin was used. For the second residence time determination, the volume of the limited channel area was used. The residence time determinations assumed that the Oak Creek was the only inflow to the impoundment and that the dam spill way was the only impoundment outflow. The residence time determination also assumed that the impoundment inflow was equal to the impoundment outflow.

**Nighttime BOD Estimates**

Substantial aquatic plant and algal growth occurred in the Mill Pond Dam impoundment during this study. Samples for this study were collected during the afternoon, when DO concentrations tend to be the highest. This sampling strategy did not capture the nighttime DO concentrations, which may have dropped below regulatory standards. We wanted to ensure that DO concentrations were accurately characterized at the study sites, but we did not have the equipment for continuous monitoring. Monthly BOD samples were being collected from the Nicholson Ave. sampling site (Nich Ave.) approximately 6 km upstream of the impoundment for another study. We used this BOD data to estimate the nighttime potential dissolved oxygen concentrations within the impoundment. Since Nich Ave. is approximately 6 km upstream of the impoundment, BOD samples were collected at both Nich Ave. and the study sites on 05/08/2017. The Nich Ave. results were compared to the study sites to determine if the BOD concentrations were similar. This was done to test our assumption that BOD concentrations did not substantially change between Nich Ave. and the study sites.

The estimated night time decrease in dissolved oxygen was calculated using Equation 2.

$$\text{Equation 2: } DO_d = \frac{BOD}{120} \times h \qquad (2)$$

Where $DO_d$ is the estimated nighttime decrease in DO (mg/l) and h is the hours of dark. The length of dark for a given sampling day was determined using tables provided by the U.S. Naval Observatory Astronomical Applications Department (USNOAAD 2015). Estimations of nighttime DO concentrations were then calculated using Equation 3.

$$\text{Equation 3: } DO_e = DO_o - DO_d \qquad (3)$$

Where $DO_e$ is the estimated nighttime dissolved oxygen concentration (mg/l) and $DO_o$ is the observed daytime dissolved oxygen concentration (mg/l).

**Exceedance Rate Determinations**

Water quality monitoring results were interpreted in the context of WI DNR standards, US EPA recommended guidelines, USGS criteria, and published research to determine exceedance rates for each parameter. Exceedance ranges and levels for state standards and recommended guidelines, by parameter are summarized in Table 1. Wisconsin monthly acute and sub-lethal water temperature standards, are summarized in Table 2. Results for MP-FLOW were excluded in the analysis of exceedance rates for TP, TN, Nitrate + Nitrite, and TKN because limited samples were collected (n = 4). There are multiple applicable standards for *E. coli*, including a geometric mean (GM), statistical threshold value (STV), and beach action value (BAV). If the results were higher than one or more of the above stated standards, it was considered an *E. coli* exceedance. *E. coli* results between Memorial Day to Labor Day were used to determine percent exceedance rates during the recreation season.

**Table 1**. Regulatory Standards and Recommended Guidelines Used to Determine Exceedance Rates. *30 day rolling geometric mean, **cannot be exceeded in >10% of samples used to calculate geometric mean, ***cannot be exceeded in any individual sample.

| Parameter | Exceedance Values | Source |
|---|---|---|
| Water Temperature | Standards vary by month (see Table 2) | WI NR 102 (WI DNR 2010) |
| TSS | > 19 mg/l | Robertson et al. 2006 |
| Turbidity | > 14 NTU | U.S. EPA 2000 |
| Specific Conductivity | > 1500 µS/cm | U.S. EPA 2012 a |
| pH | < 6 or > 9 s.u. | WI NR 102 (WI DNR 2010) |
| TP | ≥ 0.075 mg/l | WI NR 102 (WI DNR 2010) |
| $NO_2^- + NO_3^-$ | > 1.798 mg/l | U.S. EPA 2000 |
| TKN | > 0.663 mg/l | U.S. EPA 2000 |
| TN | >2.461 mg/l | Addition of NO3 + NO2 and TKN Guidelines |
| DO Concentration | < 5 mg/l | WI NR 102 (WI DNR 2010) |
| DO Saturation | > 140 % saturation | Kutty 1987 |
| *Chlorophyll-a* | 8 µg/l | KDHE 2011 |
| *E. coli* GM* | 126 MPN/100 ml | EPA 2012 b |
| *E. coli* STV** | 410 MPN/100 ml | EPA 2012 b |
| *E. coli* BAV*** | 235 MPN/100 ml | EPA 2012 b |

**Table 2**. Wisconsin Sub-Lethal and Acute Water Temperature Standards for FAL by Month. Source: WI DNR 2010.

| Month | Sub-Lethal (°C) | Acute (°C) |
|---|---|---|
| January | 9.4 | 24.4 |
| February | 10.0 | 24.4 |
| March | 11.1 | 25.0 |
| April | 12.8 | 26.1 |
| May | 18.3 | 27.8 |
| June | 24.4 | 28.9 |
| July | 27.2 | 29.4 |
| August | 27.2 | 28.9 |
| September | 22.8 | 27.8 |
| October | 16.1 | 26.7 |
| November | 9.4 | 25.0 |
| December | 9.4 | 24.4 |

### Statistical Analysis

Before statistical analysis, ten percent of collected data was randomly selected, then compared to field not to ensure data was entered correctly. Three sampling events (01/18/16, 01/25/16, and 02/15/16) were not included because most of the study sites were frozen on those dates.

Descriptive statistics were conducted utilizing R version 3.2.3 (R Core Team 2016). Trend analysis was conducted using WINKS SDA (TexaSoft, Cedar Hill, TX, USA) (TexaSoft 2011). Normality tests were conducted utilizing SigmaPlot® 12 (Systat Software, San Jose, CA, USA). Data summary tables were produced in Microsoft® Office Excel (2010). P values of < 0.05 were considered significant for correlations and tests of significant difference. Basic statistical analyses were conducted to determine the Mill Pond Dam impoundment's impact on water quality. The Kolmogorov Smirnov normality test was first applied to determine if parametric or non-parametric tests should be utilized based on data distribution. Tests of significant difference between sites were conducted on data from the entire study period grouped by sampling location utilizing the Kruskal-Wallis test and Tukey Kramer post hoc test. Spearman's rank correlations were calculated for select variables. Statistical comparisons were not conducted for *chlorophyll-a* results because of the small sample size (n=8) and the proximity of the sampling sites. Correlations for TSS and *E. coli* were conducted by site. Due to the small sample size, correlations for nutrients were conducted on the whole stream reach by combining nutrient data from all study sites. The change in water quality from upstream to downstream of the impoundment for each sampling event was determined by subtracting results at OCP from results at The Falls for each parameter.

## 4. Results

**Fish Passage**

Sixteen significant fish passage barriers were identified at stream crossings within the watershed (Figure 3). A total of 14.58 river km exists between the Mill Pond Dam and the next significant barriers to fish passage. Of this, 13.26 river km are on the mainstem and 1.29 river km are on the Mitchell Field Drainage Ditch tributary.

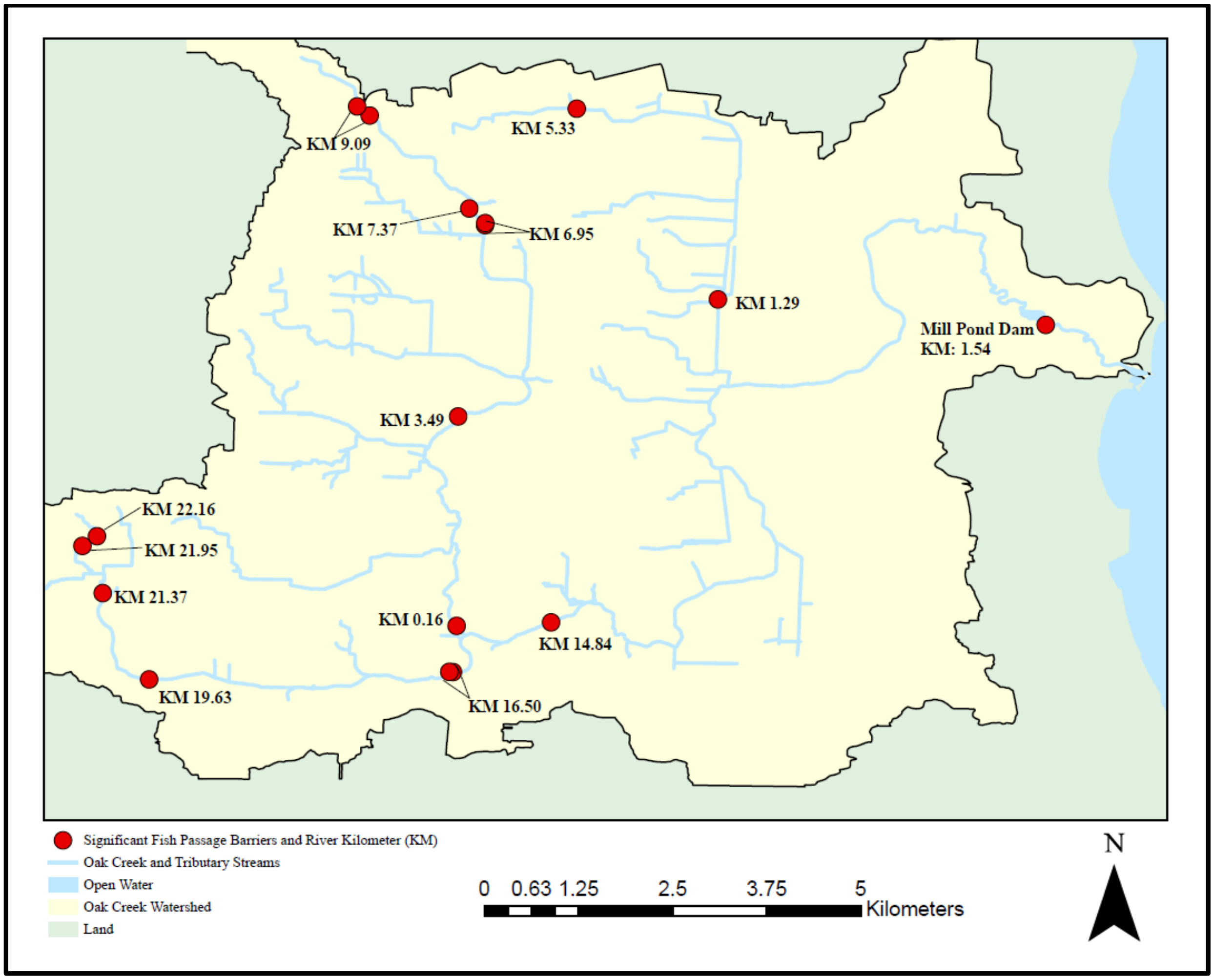


**Figure 3.** 2015 Map of stream crossings that are significant fish passage barriers Source: SEWRPC, 2016.

## Bathymetry

The Mill Pond impoundment had an area of 17,000 $m^2$ and a water volume of 3,800 $m^3$ (Figure 4). The mean depth of the impoundment was 0.22 m and the maximum depth was 1.3 m.

Water depth was < 0.9 m in 99.9% of the impoundment, and most of the impoundment had a depth of < 0.5 m.

Since 1970 (Figure 5), new landforms have formed within the impoundment including a large island and two sandbars. There has been a 5,000 $m^2$ (23%) decrease in the overall surface area of the impoundment and a 25,200 $m^3$ (87%) reduction in water volume since 1970. Maximum depth in the impoundment has decreased 1.1 m (46%).

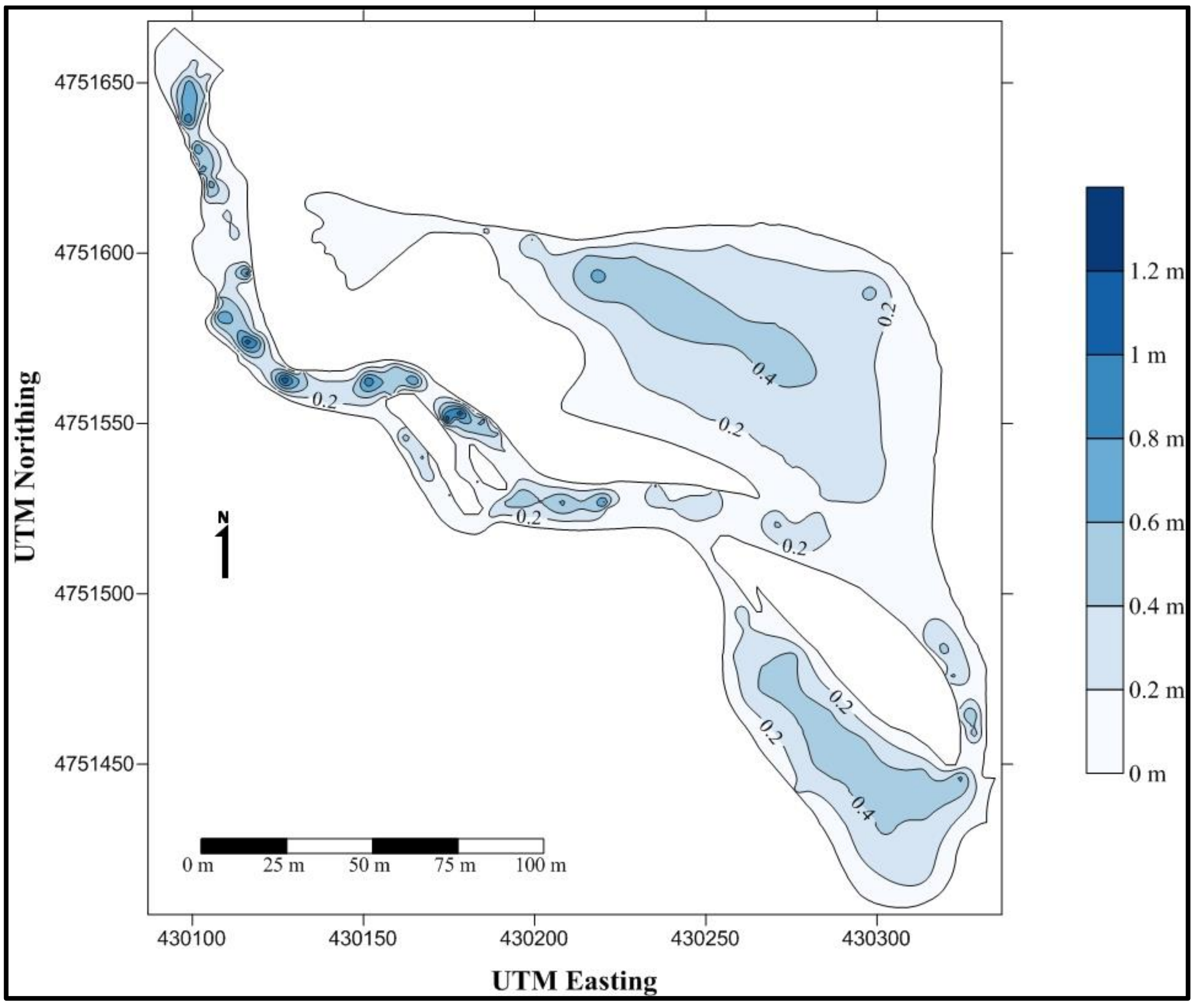


**Figure 4.** 2015 Bathymetric map of the Mill Pond Dam Impoundment.

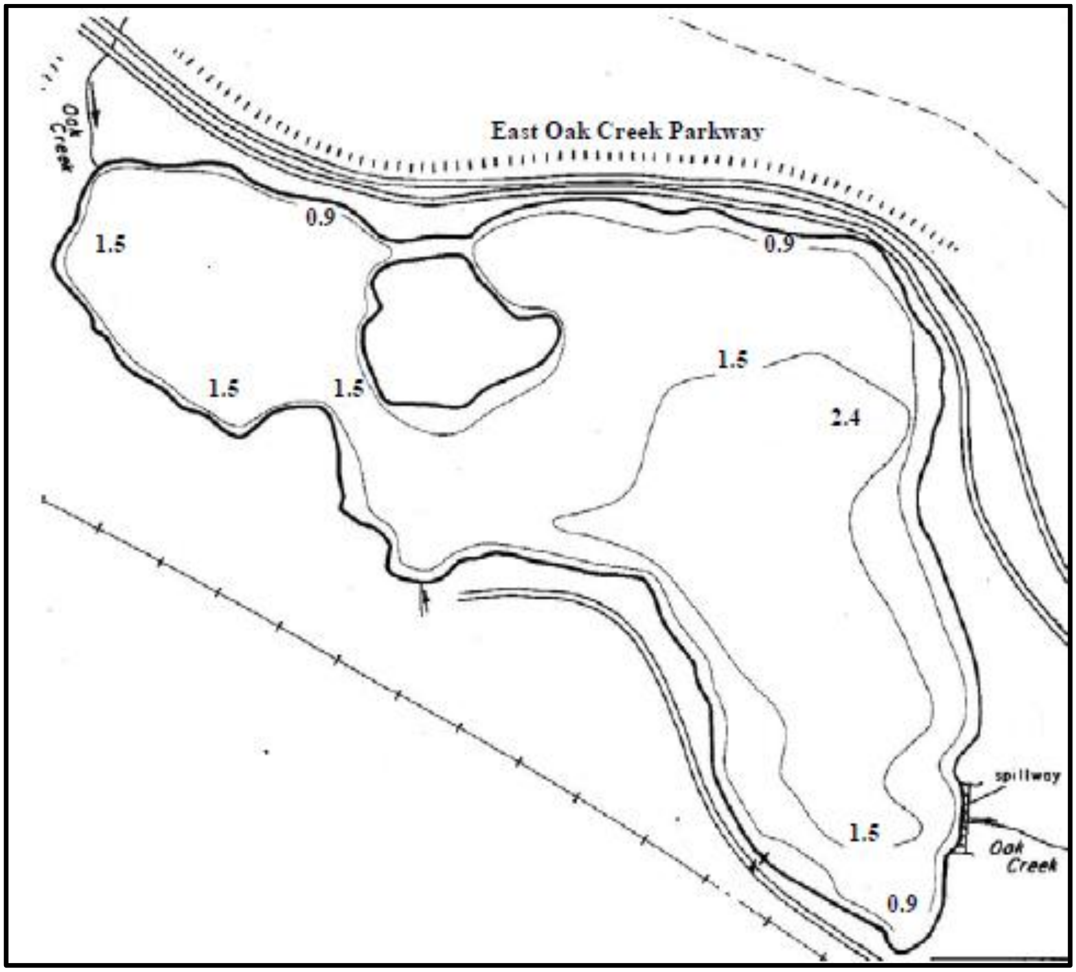


**Figure 5.** Modified 1970 Bathymetric map of the Mill Pond Dam Impoundment. Source: Roth et al. 1970. Note: The water depth units on the source map were feet. Water depth units were converted to meters for comparison purposes.

### Impoundment Residence Time

The median discharge of the Oak Creek during the study was 1,620 $\frac{m^3}{h}$. The residence time calculated using the volume of the whole impoundment was 2.35 hours and the residence time calculated using the limited channel area was 0.71 hours.

### Water Quality

*Water Temperature*

There was no statistical difference in water temperature between study sites ($p > 0.05$). Water temperature increased between OCP and The Falls on 82.5% of sampling events (Table 3). Water temperature decreased between OCP and The Falls on 7.0% of sampling events (Table 3). There was no difference in water temperature between OCP and The Falls for 10.5% of sampling events (Table 3). The median single sampling event change in water temperature between OCP and The Falls was +1.1 °C (Table 3). Wisconsin state standards for sub-lethal water temperature were exceeded in 7% of samples upstream of the impoundment at OCP (Table 4). Within the impoundment, regulatory standards for sub-lethal water temperature were exceeded in 15.2% and 21.2% of samples at MP-FLOW and MP respectively (Table 4). Downstream of the impoundment, sub-lethal water temperature standards were exceeded in 13.8%, 13.5%, and 8.9% of samples at The Falls, HAW, and OCM respectively.

**Table 3.** Percentage of sampling events where parameter results, increased, decreased, or did not change between OCP and The Falls and the median change in parameter results between OCP and The Falls. n = number of sampling events, Increase= percent of sampling events where parameter results increased from OCP to The Falls, Decrease = percent of sampling events where parameter results decreased from OCP to The Falls, No Difference = percent of sampling events where parameter results were the same at OCP and The Falls, Median Change = median change in single sampling event parameter results between OCP and The Falls.

| Parameter | n | Increase | Decrease | No Difference | Median Change |
|---|---|---|---|---|---|
| WT | 57 | 82.5 | 7.0 | 10.5 | + 1.1 °C |
| TSS | 58 | 87.9 | 8.6 | 3.4 | + 3.85 mg/l |
| Turbidity | 58 | 91.4 | 8.6 | 0.0 | + 2.32 NTU |
| Specific Conductivity | 57 | 19.3 | 80.7 | 0.0 | - 26 µS/cm |
| pH | 57 | 84.2 | 7.0 | 8.8 | + 0.13 s.u. |
| TP | 11 | 54.5 | 36.4 | 9.1 | + 0.004 mg/l |
| $NO_2^- + NO_3^-$ | 12 | 25.0 | 50.0 | 25.0 | - 0.005 mg/l |
| TKN | 12 | 50.0 | 25.0 | 25.0 | + 0.03 mg/l |
| TN | 12 | 66.7 | 25.0 | 8.3 | + 0.053 mg/l |
| DO Concentration | 57 | 77.2 | 22.8 | 0.0 | + 0.56 mg/l |
| DO Saturation | 57 | 75.4 | 24.6 | 0.0 | + 8.1 % saturation |
| EC | 58 | 60.3 | 34.5 | 5.2 | + 40 MPN/100 ml |

**Table 4.** Percent exceedance of regulatory standards/recommended guidelines by study site for assessed parameters. - = Not Assessed

| | Study Sites | | | | | |
|---|---|---|---|---|---|---|
| **Parameter** | **OCP** | **MP-FLOW** | **Mill Pond** | **The Falls** | **HAW** | **OCM** |
| Sub-Lethal WT | 7.0 | 15.2 | 21.2 | 13.8 | 13.5 | 8.9 |
| TSS | 10.3 | 15.2 | 21.2 | 19.0 | 10.8 | 10.7 |
| Turbidity | 15.5 | 18.2 | 34.6 | 22.4 | 13.5 | 14.3 |
| Specific Conductivity | 61.4 | 72.7 | 53.8 | 60.3 | 70.3 | 44.6 |
| pH | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| TP | 16.7 | - | 33.3 | 25.0 | 25.0 | 8.3 |
| $NO_2^- + NO_3^-$ | 0.0 | - | 0.0 | 0.0 | 0.0 | 0.0 |
| TKN | 41.7 | - | 58.3 | 66.7 | 75.0 | 58.3 |
| TN | 8.3 | - | 8.3 | 8.3 | 12.5 | 0.0 |
| DO Concentration | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| DO Saturation | 5.3 | 33.3 | 36.5 | 0.0 | 2.7 | 10.7 |
| EC (GM+STV+BAV) | 75.9 | 60.6 | 74.1 | 75.9 | 67.6 | 75.9 |

*TSS and Turbidity*

TSS upstream of the impoundment (OCP) was significantly lower than within the impoundment (MP-FLOW and MP) and immediately downstream of the impoundment (The Falls) ($p < 0.05$). There were no statistical differences in TSS between OCP, HAW, or OCM ($p > 0.05$). TSS increased between OCP and The Falls on 87.9% of sampling events (Table 3). TSS decreased between OCP and The Falls on 8.6% of sampling events. Twenty four hour antecedent precipitation exceeded 6.35 millimeters in 80% of events where TSS decreased between OCP and The Falls (Table 3). There was no difference in TSS at OCP and The Falls on 3.4% of sampling events (Table 3). The median single sampling event change in TSS between OCP and The Falls was +3.85 mg/l (Table 3). Percent exceedance of recommended guidelines for TSS and Turbidity are summarized in Table 4. Upstream of the impoundment (OCP), recommended guidelines for TSS were exceeded in 10.3% of samples. Within the impoundment, recommended guidelines for TSS were exceeded in 15.2% and 21.2% of samples at MP-FLOW and MP respectively. Downstream of the impoundment, recommended guidelines for TSS were exceeded in 19%, 10.8%, and 10.7% at The Falls, HAW, and OCM respectively. TSS was positively correlated with 24, 48, 72 hour antecedent precipitation at OCP, MP, The Falls, HAW, and OCM ($p < 0.05$) (Table 5). There was no correlation between TSS and discharge rate at OCP, MP, The Falls, HAW, and OCM ($p > 0.05$) (Table 5). TSS was positively correlated with turbidity at OCP, MP-FLOW, MP, The Falls, HAW, and OCM ($p > 0.05$) (Table 5).

Turbidity at OCP was statistically lower than turbidity at MP-FLOW, MP, and The Falls ($p < 0.05$). There was no statistical difference in turbidity between OCP, HAW, or OCM ($p > 0.05$). Turbidity increased between OCP and The Falls on 91.4% of sampling events (Table 3). Turbidity decreased between OCP and The Falls on 8.6% of sampling events (Table 3). The median single sampling event change in turbidity between OCP and The Falls was +2.32 NTU (Table 3). Upstream of the impoundment (OCP), recommended guidelines for turbidity were exceeded in 15.5% of samples. Within the impoundment, guidelines for turbidity were exceeded in 18.2% and 34.6% of samples at MP-FLOW and MP respectively. Downstream of the impoundment, recommended guidelines for turbidity were exceeded in 22.4%, 13.5%, and 14.3% of samples at The Falls, HAW, and OCM respectively.

**Table 5.** Correlation of TSS with 24, 48, and 72 hour antecedent precipitation, discharge rate, and turbidity by study site. $r_s$ = Spearman's rho, * = p values < 0.5, Precip = antecedent precipitation.

| Site (Upstream to Downstream) | Correlations | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 24 hr Precip. | | 48 hr Precip. | | 72 hr Precip. | | Discharge Rate | | Turbidity | |
| | $r_s$ | p | $r_s$ | p | $r_s$ | p | $r_s$ | p | $r_s$ | p |
| OCP | 0.495 | < 0.001* | 0.593 | < 0.001* | 0.540 | < 0.001* | 0.486 | < 0.001* | 0.820 | < 0.001* |
| MP-FLOW | 0.171 | 0.342 | 0.311 | 0.078 | 0.224 | 0.210 | 0.120 | 0.507 | 0.593 | < 0.001* |
| MP | 0.386 | 0.005* | 0.366 | 0.008* | 0.353 | 0.010* | 0.034 | 0.812 | 0.787 | < 0.001* |
| The Falls | 0.348 | 0.008* | 0.367 | 0.005* | 0.334 | 0.010* | 0.062 | 0.644 | 0.706 | < 0.001* |
| HAW | 0.415 | 0.011* | 0.720 | < 0.001* | 0.588 | < 0.001* | -0.082 | 0.629 | 0.755 | < 0.001* |
| OCM | 0.357 | 0.007* | 0.463 | < 0.001* | 0.471 | < 0.001* | 0.252 | 0.061 | 0.841 | < 0.001* |

*Specific Conductivity and pH*

There were statistical differences in Specific conductivity between study sites ($p < 0.05$); however, post hoc tests were unable to identify where the differences occurred. Specific conductivity increased between OCP and The Falls on 19.3% of sampling events (Table 3). Specific conductivity decreased between OCP and The Falls on 80.7% of sampling events (Table 3). The median single sampling event change in specific conductivity between OCP and The Falls was -26 µS/cm (Table 3). Percent exceedance of recommended guidelines for specific conductivity are summarized in Table 4. Upstream of the impoundment (OCP), recommended guidelines for specific conductivity were exceeded in 61.4% of samples. Within the impoundment, guidelines for specific conductivity were exceeded in 72.7% and 53.8% at MP-FLOW and MP respectively. Downstream of the impoundment, recommended guidelines for specific conductivity were exceeded in 60.3%, 70.3%, and 44.6% at The Falls, HAW, and OCM respectively.

OCP had significantly lower pH than MP-FLOW, MP, The Falls, and HAW ($p < 0.05$). There was an increase in pH between OCP and The Falls on 84.2% of sampling events (Table 3).

There was a decrease in pH between OCP and The Falls on 7.0% of sampling events (Table 3). There was no difference in pH between OCP and The Falls on 8.8% of sampling events (Table 3). The median single sampling event change in pH between OCP and The Falls was +0.13 s.u. (Table 3). There were no observations of pH values outside of the range stated in Wisconsin regulatory standards

*TP*, $NO_2^- + NO_3^-$, *TKN*, *TN*

There were no statistical differences in TP between study sites ($p > 0.05$). TP increased between OCP and The Falls on 54.5% of sampling events (Table 3). TP decreased between OCP and The Falls on 36.4% of sampling events (Table 3). There was no difference in TP between OCP and The Falls on 9.1% of sampling events (Table 3). The median single sampling event change in TP between OCP and The Falls was +0.004 mg/l (Table 3). Percent exceedance of recommended guidelines and regulatory standards for TP, TKN, and TN are summarized in Table 4. Upstream of the impoundment (OCP), state standards for TP were exceeded in 16.7% of samples. Within the impoundment (MP), TP standards were exceeded in 33.3% of samples. Downstream of the impoundment, TP standards were exceeded in 8.3% of samples at OCM and 25.0% of samples at The Falls and HAW. Reach wide, TP was negatively correlated with antecedent precipitation and discharge rate, and positively correlated with TSS ($p < 0.05$) (Table 6).

There were no statistical differences in $NO_2^- + NO_3^-$ between study sites ($p > 0.05$). $NO_2^- + NO_3^-$ increased between OCP and The Falls on 25.0% of sampling events (Table 3). $NO_2^- + NO_3^-$ decreased between OCP and The Falls on 50.0% of sampling events (Table 3). There was no difference in $NO_2^- + NO_3^-$ between OCP and The Falls on 25.0% of sampling events (Table 3). The median single sampling event change in $NO_2^- + NO_3^-$ between OCP and The Falls was

–0.005 mg/l (Table 3). There were no observed exceedances of recommended guidelines for $NO_2^- + NO_3^-$ during this study. Reach wide, $NO_2^- + NO_3^-$ was correlated negatively with TSS and positively with 24-hour antecedent precipitation, 48-hour antecedent precipitation, and discharge rate ($p < 0.05$) (Table 6).

There were no statistical differences in TKN between study sites ($p > 0.05$). TKN increased between OCP and The Falls on 50% of sampling events (Table 3). TKN decreased between OCP and The Falls on 25% of sampling events (Table 3). There was no difference in TKN between OCP and The Falls on 25% of sampling events (Table 3). The median single sampling event change in TKN between OCP and The Falls was +0.03 mg/l (Table 3). Upstream of the impoundment (OCP), recommended guidelines for TKN were exceeded in 41.7% of samples. Within the impoundment (MP), TKN guidelines were exceeded in 58.3% of samples. Downstream of the impoundment, guidelines for TKN were exceeded in 66.7%, 75.0%, and 58.3% of samples at The Falls, HAW, and OCM respectively. Reach wide, TKN was negatively correlated with 72-hour antecedent precipitation and positively correlated with TSS ($p < 0.05$) (Table 6).

There were no statistical differences in TN between study sites ($p > 0.05$). TN increased between OCP and The Falls on 66.7% of sampling events (Table 3). TN decreased between OCP and The Falls on 25.0% of sampling events (Table 3). There was no difference in TN between OCP and The Falls on 8.3% of sampling events (Table 3). The median single sampling event change in TN between OCP and The Falls was +0.053 mg/l (Table 3). Recommended guidelines for TN were exceeded in 8% of samples at OCP, MP, and The Falls. Guidelines for TN were exceeded in 12.5% and 0.0% of samples at HAW and OCM respectively. Reach wide, TN was positively correlated with discharge rate ($p < 0.05$) (Table 6).

**Table 6.** Reach wide correlations of TP, TN, TKN, and $NO_2^- + NO_3^-$ with 24,48, and 72 hr antecedent precipitation, discharge rate, and TSS. $r_s$ = Spearman's rho, * = p values < 0.5, Precip = antecedent precipitation

| Parameter | Correlations | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | **24 hr Precip.** | | **48 hr Precip.** | | **72 hr Precip.** | | **Discharge Rate** | | **TSS** | |
| | $r_s$ | p | $r_s$ | p | $r_s$ | p | $r_s$ | p | $r_s$ | p |
| TP | -0.327 | 0.011* | -0.372 | 0.003* | -0.398 | 0.002* | -0.599 | < 0.001* | 0.452 | < 0.001* |
| NO3+NO2 | 0.343 | 0.007* | 0.355 | 0.005* | 0.175 | 0.181 | 0.626 | < 0.001* | -0.277 | 0.032* |
| TKN | -0.184 | 0.158 | -0.149 | 0.255 | -0.328 | 0.011* | 0.251 | 0.054 | 0.417 | < 0.001* |
| TN | 0.224 | 0.086 | 0.092 | 0.485 | -0.136 | 0.302 | 0.546 | < 0.001* | 0.135 | 0.303 |

*DO, BOD-5 Day,* and *Chlorophyll –a*

Results for $DO_o$, $DO_d$, and $DO_e$ are summarized in Table 7. Comparison samples collected at Nich Ave. and the study sites on 05/08/2017 had a BOD of < 2 mg/l at all assessed sites. BOD at Nich Ave. had a median of 2.4 mg/l ranged from 1.4 – 9.1 mg/l. $DO_d$ ranged from 0.13 – 0.94 mg/l and had a median of 0.22 mg/l. $DO_e$ ranged from 6.79 – 35.11 mg/l and had a median of 13.59 mg/l.

DO concentrations were significantly higher at MP-FLOW and MP than at OCP and OCM ($p < 0.05$). DO concentrations increased between OCP and The Falls on 77.2% of sampling events (Table 3). DO concentrations decreased between OCP and The Falls on 22.8% of sampling events (Table 3). The median single sampling event change in DO concentrations between OCP and The Falls was +0.56 mg/l (Table 3). There were no observations of DO concentrations below state standards. Estimated nighttime dissolved oxygen concentrations ($DO_e$) were within regulatory limits.

**Table 7.** $DO_o$, $DO_d$, and $DO_e$ results for MP-FLOW and MP. $DO_o$ = observed daytime dissolved oxygen (mg/l), $DO_d$ = estimated nighttime decrease in dissolved oxygen (mg/l), $DO_e$ = estimated nighttime dissolved oxygen concentration (mg/l).

| Date | MP-FLOW $DO_o$ | MP-FLOW $DO_d$ | MP-FLOW $DO_e$ | MP $DO_o$ | MP $DO_d$ | MP $DO_e$ |
|---|---|---|---|---|---|---|
| 7/27/2015 | Not Collected | Not Collected | Not Collected | 20.16 | 0.23 | 19.93 |
| 8/31/2015 | Not Collected | Not Collected | Not Collected | 12.58 | 0.13 | 12.45 |
| 9/28/2015 | Not Collected | Not Collected | Not Collected | 13.21 | 0.20 | 13.01 |
| 10/26/2015 | Not Collected | Not Collected | Not Collected | 7.08 | 0.29 | 6.79 |
| 11/30/2015 | Not Collected | Not Collected | Not Collected | 14.36 | 0.36 | 14.00 |
| 12/28/2015 | 15.84 | 0.30 | 15.54 | 13.89 | 0.30 | 13.59 |
| 3/8/2016 | 13.49 | 0.94 | 12.55 | 14.09 | 0.94 | 13.15 |
| 4/4/2016 | 13.36 | 0.22 | 13.14 | 14.59 | 0.22 | 14.37 |
| 5/2/2016 | 15.50 | 0.57 | 14.93 | 13.51 | 0.57 | 12.94 |
| 6/13/2016 | 24.61 | 0.17 | 24.44 | 35.28 | 0.17 | 35.11 |
| 7/11/2016 | 14.03 | 0.18 | 13.85 | 14.08 | 0.18 | 13.90 |
| 8/8/2016 | 12.65 | 0.20 | 12.45 | 11.19 | 0.20 | 10.99 |

MP-FLOW and MP had significantly higher DO % sat than OCP, The Falls, and OCM ($p < 0.05$). DO % sat increased between OCP and The Falls on 75.4% of sampling events (Table 3). DO % sat decreased between OCP and The Falls on 24.6% of sampling events (Table 3). The median single sampling event change in DO % sat between OCP and The Falls was +8.1 % (Table 3). DO% sat increased between MP and The Falls in all sampling events where DO was not supersaturated at MP. DO % saturation decreased between MP and The Falls in 87% of samples where DO was supersaturated at MP. Percent exceedance of recommended guidelines for DO % sat are summarized in Table 4. Upstream of the impoundment (OCP), recommended guidelines for DO % sat were exceeded in 5.3% of samples. Within the impoundment, recommended guidelines for DO % sat were exceeded in 33.3% and 26.5% of samples at MP-FLOW and MP respectively. Downstream of the impoundment, recommended guidelines for DO % sat were exceeded in 0.0%, 2.7%, and 10.7% at The Falls, HAW, and OCM respectively.

*Chlorophyll-a* results are shown in Table 8. *Chlorophyll-a* concentrations ranged from 1.00 to 87.40 µg/l and had a median of 9.12 µg/l. Recommended guidelines for *chlorophyll-a* were exceeded in 75% samples at MP1-S and MP1-D.

**Table 8**. Observed Chlorophyll-a concentrations (µg/l) at MP1-S and MP1-D.

| Date | MP1-S | MP1-D |
|---|---|---|
| 7/29/2015 | 43.80 | 87.40 |
| 8/5/2015 | 8.99 | 9.25 |
| 8/11/2015 | 8.50 | 11.40 |
| 8/18/2015 | 4.35 | 1.00 |

*E. coli*

There were no statistical differences in EC between study sites ($p < 0.05$). EC increased between OCP and The Falls on 60.3% of sampling events (Table 3). EC decreased between OCP and The Falls on 34.5% of sampling events (Table 3). There was no difference in EC between OCP and The Falls on 5.2% of sampling events (Table 3). The median single sampling event change in EC between OCP and The Falls was +40 MPN/100ml (Table 3). OCP, The Falls, and OCM had an *E. coli* exceedance rate of 75.9% (Table 4). MP-FLOW, MP, and HAW had exceedance rates of 60.6%, 74.1% and 67.6% respectively (Table 4). EC exceedance rates during the recreation season were 92.9% and 91.7% at MP-FLOW and MP respectively. EC correlation results are summarized in Table 9. EC was positively correlated with 24, 48, and 72 hour antecedent precipitation at all (6/6) study sites ($p < 0.05$). EC was positively correlated with TSS at OCP, MP-FLOW, The Falls, HAW, and OCM. There were no correlations between EC and discharge rate ($p > 0.05$).

**Table 9.** Correlation of EC with 24, 48, and 72 hour antecedent precipitation, discharge rate, and TSS by study site. $r_s$ = Spearman's rho, * = p values < 0.5, Precip = antecedent precipitation

| Site (Upstream to Downstream) | Correlations | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 24 hr Precip. | | 48 hr Precip. | | 72 hr Precip. | | Discharge Rate | | TSS | |
| | $r_s$ | p | $r_s$ | p | $r_s$ | p | $r_s$ | p | $r_s$ | p |
| OCP | 0.370 | 0.005* | 0.482 | < 0.001* | 0.462 | < 0.001* | 0.104 | 0.442 | 0.457 | < 0.001* |
| MP-FLOW | 0.384 | 0.027* | 0.550 | < 0.001* | 0.564 | < 0.001* | -0.124 | 0.493 | 0.375 | 0.031* |
| MP | 0.308 | 0.026* | 0.446 | < 0.001* | 0.497 | < 0.001* | -0.020 | 0.890 | 0.144 | 0.310 |
| The Falls | 0.327 | 0.012* | 0.453 | < 0.001* | 0.479 | < 0.001* | -0.061 | 0.647 | 0.622 | <0.001* |
| HAW | 0.415 | 0.011* | 0.640 | < 0.001* | 0.618 | < 0.001* | -0.022 | 0.900 | 0.642 | < 0.001* |
| OCM | 0.427 | 0.001* | 0.483 | < 0.001* | 0.508 | < 0.001* | 0.090 | 0.508 | 0.567 | < 0.001* |

## 5. Discussion

The objective of this study was to determine if the Mill Pond Dam provides the unintended benefits of acting as a barrier to invasive species, creating recreational opportunities, and improving downstream water quality. Stream crossing surveys were utilized to determine if the Mill Pond Dam provides protection against the potential habitation of invasive species to upstream areas. Creating a bathymetric map and comparing *E. coli* values to recreational contact standards determined the impoundments suitability for recreation. The suite of parameters utilized to assess water quality in this study characterized the dam's impact on physical, chemical, and biological aspects of water quality. The methods utilized in this study are transferable and could be used at other sites to improve the understanding of the potential consequences of dam removal.

### Suitability as Invasive Species Barrier

Removal of the Mill Pond Dam would result in an increase of total mainstem river km without significant barriers to fish passage from 7.2% to 66.9%. Since the stream crossing survey determined the Mill Pond Dam was a significant fish passage barrier and removal of the dam would result in an increased risk of potential habitation by invasive species in a substantial portion of the stream, the dam was deemed a good barrier to invasive species passage. Despite this, dams are not perfect barriers. Invasive species, including round gobies, have established populations upstream of dams; presumably from transfer via bait buckets (Carman et al. 2006). Conversely, removal of the dam does not guarantee the introduction of invasive species upstream, and the increase in stream connectivity could benefit desirable fish species.

Further study is needed to determine if there are sensitive populations or species upstream of the impoundment that would be severely impacted by the introduction of invasive species, if invasive species are present either upstream or downstream of the dam, if the upstream habitat and water quality is suitable for the proliferation of key invasive species, and how dam removal would impact multiple species within the Oak Creek and Lake Michigan.

**Suitability for Recreation**

The shallow water depths throughout the impoundment make it unsuitable for kayaking and canoeing. Anecdotally, the kayak used for collecting bathymetric data frequently became lodged in bottom sediment while completing surveys. The accumulation of substantial amounts of fine sediment throughout the impoundment results in conditions where entrapment in the sediment could lead to drowning. Additionally, high *E. coli* recreational standard exceedance rates within the impoundment indicate that bacteriological water quality is not suitable for recreational contact. The results indicate that EC within the impoundment originates from upstream sources. Efforts to establish water quality suitable for recreational contact within the impoundment would have to mitigate the effects of *E. coli* loads from upstream sources.

**Effects on Water Quality**

*Water Temperature*

Statistical comparisons between study sites indicate that the Mill Pond Dam and impoundment do not impact water temperature; however, single sample event comparisons and differences in the exceedance of state standards suggest that they do. Specifically, water temperature increased from OCP to The Falls on 82.5% of sampling events, with a median change of + 1.1 °C, and sub-lethal water temperature standard exceedances increased 8.2 – 14.2% within the impoundment, and 6.5 – 6.8% downstream of the dam where mixing with Lake Michigan does not occur. The increase in water temperature is likely attributed to the slowed flow rates and reduced shade within the impoundment, causing increased exposure to solar radiation, and the filling of the impoundment with sediment, increasing the average exposure to solar radiation within the water column. This analysis of sub-lethal water temperature exceedances is limited because it is based on weekly discreet samples; whereas; Wisconsin's regulatory standards are based on the weekly average maximum temperatures (NR 102.25 (1) (b)). If continuous sampling produced similar results, then the stream would be considered impaired for elevated water temperature within and downstream of the impoundment. Thus, despite a lack of statistical significance in water temperature between study sites, the dam's impact is substantial enough to have management implications.

*TSS and Turbidity*

Analysis of statistical differences between sites, single sampling event comparisons, and comparisons to recommended guidelines produced similar results for both TSS and turbidity, and the correlation between TSS and turbidity suggests that turbidity is mostly influenced by TSS. Additionally, TSS and turbidity results both indicate that the dam and impoundment negatively impact suspended sediment concentrations. Suspended sediment concentrations within the impoundment are higher than areas immediately upstream. The suspended sediment concentrations remain elevated downstream of the dam, with a median change of +3.85 mg/l between OCP and The Falls; however, this impact is spatially limited, and is not observed 0.55 km downstream of the dam. Suspended sediment concentrations are typically lower within and immediately downstream of impoundments (Poff and Hart 2002). The results indicate that the Mill Pond Dam and impoundment have the opposite effect on suspended sediment concentrations. The substantial reduction in water volume from 1970 to 2015 and the shallow conditions observed throughout the impoundment indicate it has filled with sediment. The reduced storage capacity of the impoundment prevents it from acting as a sink for upstream sediments. Instead, the impoundment acts as a source of suspended sediment to downstream areas when it is not influenced by precipitation events. This negatively impacts downstream water and habitat quality by reducing water clarity, burying fish eggs, and suffocating aquatic insects (Robertson et al. 2006). Additionally, multiple compounds including nutrients, heavy metals, and bacteria (e.g. *E. coli*) are often attached to suspended sediments, and previous studies have identified arsenic, cadmium, chromium, copper, lead, mercury, nickel, zinc, polycyclic aromatic hydrocarbons, and polychlorinated biphenyl in Oak Creek sediment. (Robertson et al. 2006, Herngren et al. 2005, Dong et al. 1984, Pandey and Soupir 2013, SEWRPC 2007). Additional studies are needed to determine if contaminates bound to sediment leaving the impoundment result in additional downstream impairments.

*Specific Conductivity and pH*

Statistical differences in specific conductivity between study sites were observed, but post hoc tests failed to determine where the statistical differences occurred. The single sampling event comparisons suggest that the impoundment slightly decreased conductivity. It is not clear what caused this decrease in specific conductivity.

Single sampling event comparisons and significant differences in pH suggest that the dam causes an increase in pH within and downstream of the impoundment where mixing with Lake Michigan does not occur. The median change in pH was + 0.13 s.u. which was not substantial enough to cause pH values outside of regulatory limits. The pH of water increases as temperature increases, and aquatic vegetation consumes $CO_2$ during photosynthesis, which increases pH. The observed increase in pH may have been caused by increased water temperature within and downstream of the impoundment, and eutrophication within the impoundment.

*TP*, $NO_2^-$ + $NO_3^-$, *TKN, and TN*

The lack of statistical significance observed between study sites for TP, $NO_2^-$ + $NO_3^-$, TKN, and TN may be due to the small sample size for each site ($n \leq 12$). Additional TP, $NO_2^-$ + $NO_3^-$, TKN, and TN samples for each site are needed to ensure an accurate characterization of statistical differences between study sites.

TN concentrations increased between OCP and The Falls on most sampling events, and $NO_2^-$ + $NO_3^-$ decreased between OCP and The Falls on most sampling events. The changes in TN and $NO_2^-$ + $NO_3^-$ were not substantial enough to influence exceedances of recommended guidelines. TKN concentrations increased between OCP and The Falls on most sampling events. The exceedances of recommended guidelines for TKN progressively increased from upstream to downstream, excluding OCM. This suggests that the increases observed in TKN are not related to the dam or impoundment. Instead, the increases observed in TKN appear to be related to gradual increases in TKN from upstream to downstream.

Exceedance of regulatory standards for TP substantially increase between OCP and MP. Downstream of the impoundment, exceedances occurred less frequently than within the impoundment, but more frequently than upstream of the impoundment. The downstream impact on TP regulatory limit exceedances is observed until mixing with Lake Michigan occurs. TP increased between OCP and The Falls in over half of sampling events, decreased in 36.4% of sampling events, and had a median change of +0.004 mg/l. This further suggests that overall, TP concentrations increased within and downstream of the impoundment. The increase in TP did not consistently occur, and was minimal when it did occur; however, the increases were enough to increase the occurrence of regulatory limit exceedances downstream of the impoundment. Previous studies have linked nutrient concentrations to TSS (Robertson et al. 2006). The substantial accumulation of sediment within the impoundment, and the release of this sediment to downstream areas likely caused the differences in TP exceedances. The strong relation between reach wide TP and TSS indicates that a large proportion of the phosphorus load is attached to sediment. Negative correlations between nutrient concentrations and antecedent precipitation are likely a result of a dilution of in-stream concentrations. It is unlikely that precipitation results in a reduction in the streams overall nutrient loads.

*DO, BOD-5 Day and Chlorophyll-a*

Analysis of statistical differences and guideline exceedances produced similar results and exhibited similar patterns for DO concentration and percent saturation. DO concentrations are elevated within the impoundment; however, this does not significantly affect downstream concentrations. Despite this, DO concentration and DO % sat increased between OCP and The Falls in 77.2% and 75.4% of single sampling events respectively. This suggests that the impoundment increases downstream DO concentrations; however, the magnitude of these increases is dampened due to agitation of supersaturated water flowing over the dam resulting in the release of oxygen to the atmosphere.

Frequent exceedances of DO % sat guidelines indicate that the impoundment is highly eutrophic. Observed *chlorophyll-a* concentrations and visual identification of excessive aquatic vegetation and algal growth further suggest that the impoundment is highly eutrophic. The high DO % sat concentrations and eutrophic conditions observed within the impoundment likely negatively impacts macroinvertebrate and fish assemblages (Kutty 1987). DO % sat increases between MP and The Falls when DO is not supersaturated at MP, and decreases between MP and The Falls when DO is supersaturated at MP. This indicates that the agitation of water that occurs as it flows over the dam releases oxygen into the atmosphere when the water is supersaturated with oxygen, and incorporates oxygen from the atmosphere into the water when it is not supersaturated.

The $DO_e$ values suggest that the diurnal fluctuation within the impoundment is minimal, and that DO does not decrease to concentrations that are dangerous to aquatic life at night. The maximum estimated diurnal fluctuation of DO was 0.94 mg/l. This fluctuation is much smaller than what is expected for a highly eutrophic water body. The low estimated diurnal fluctuations may be a result of inaccurate assumptions made about BOD concentrations in the stream. The BOD samples used to estimate the diurnal fluctuation were collected approximately 6 river km upstream of the impoundment. BOD samples were collected at the study sites and Nich Ave. on a single day in May to assess differences in BOD between sampling sites. This sampling event was conducted early in the growing season for aquatic vegetation. If the samples were collected in warmer conditions later in the growing season, differences in BOD may have been observed. The average difference in DO concentration between impoundment sites and the upstream site

(Nich Ave.) was 3.91 and 14.38 mg/l in May and June samples respectively. This suggests that more substantial differences in water quality occur during the warmer months and that the assumption that BOD does not change between sites was invalid. Additionally, it is not known if the DO concentration reached zero before the five-day BOD analysis was completed. If this occurred, it would have resulted in a more conservative estimate of the nighttime decrease in DO. Continuous monitoring of DO within the impoundment during multiple meteorological seasons would more accurately characterize the diurnal fluctuation within the impoundment.

*E. coli*

Sediment associated EC accounts for a major portion of in-stream EC concentrations (Pandley and Soupir 2013). Upstream sediments contaminated with EC can have a substantial impact on downstream EC concentrations (Pandley and Soupir 2013). This impact is exacerbated when high flow rates result in the re-suspension of stream bed sediment (Pandley and Soupir 2013). In addition to instream sources, stormwater runoff is a major contributor of bacteria and sediment to streams (Clary et al. 2008, McCarthy et al. 2012). The high exceedance rate of EC standards at OCP indicate that the high EC concentrations observed within and downstream of the impoundment were not from localized sources, or degraded conditions in the impoundment. Instead, the high EC concentrations within and downstream of the impoundment originate from upstream sources. The positive correlation between EC and TSS suggests that much of this EC is associated with the transport of upstream sediment into the impoundment. The positive correlation of antecedent precipitation with both TSS and EC suggests that the upstream sediment is mobilized during precipitation events. The lack of correlation of discharge rate with both TSS and EC suggests that the sediment associated EC does not originate from resuspended stream bed sediment. Instead, the sediment associated EC likely originates from stormwater runoff transporting non-point source pollution to upstream reaches.

The amount and size of sediment that can be transported by a stream typically decreases when a stream enters an impoundment (Baxter 1977). This reduction typically results in the deposition of sediment within the impoundment. Furthermore, the deposition of sediment would result in the reduction of sediment associated contaminates in the water column, including EC. This wasn't observed in the Mill Pond. Instead, percent exceedance rates of regulatory standards for EC were the same at the inflow (OCP) and outflow (The Falls) of the impoundment, and there was no significant difference in EC between study sites. This indicates that the shallow conditions and short residence time in the impoundment prevent the unintended benefit of reducing EC concentrations in the water column from being realized.

## 6. Conclusions

The Mill Pond Dam provided a good barrier to fish passage within the timeframe of this study. If the Mill Pond Dam was removed, a substantial portion of the watershed would experience an increased risk of invasive species habitation. Although physical barriers prevent invasive species from moving upstream, they do not prevent introduction via other mechanisms such as bait bucket transfer. Conversely, removal of the dam doesn't guarantee that invasive species will be introduced to upstream areas and the dam acts as an upstream barrier to all species, including fish of high social value. Further studies are needed to determine the suitability of upstream habitat for invasive and desirable species, if key invasive species are present upstream, downstream, or within the impoundment, and to determine if sensitive or desirable upstream species would be negatively affected by the removal of the dam. Completion of these studies would help determine if the potential benefits of removal would outweigh the potential negative impacts.

The Mill Pond did not provide the unintended benefit of recreational access within the timeframe of this study. The substantial accumulation of sediments within the impoundment has created shallow conditions which are unsuitable for canoeing, kayaking, or swimming. Additionally, the bacteriological water quality is not supportive of recreational use. Efforts to establish water quality suitable for recreational contact within the impoundment would have to mitigate the effects of E. coli loads from upstream sources.

The Mill Pond Dam and impoundment did not provide the unintended benefit of improving water quality for any of the measured parameters within the timeframe of this study. This is likely due to the short residence time within the impoundment (0.71 – 2.35 hours). Instead, the dam and impoundment negatively influenced water quality. The negative impacts on water quality can be attributed to substantial accumulation of sediment within the impoundment. Slowed flows combined with shallow conditions result in elevated water temperatures within the impoundment. This increased water temperatures downstream of the dam, which caused the entire downstream reach to be impaired for elevated water temperatures. The excessive accumulation of sediments has resulted in the impoundment being a source of sediment to areas immediately downstream. Additionally, the increase in TP, observed in all areas downstream of the dam where mixing with Lake Michigan does not occur, is likely attributed to the release of sediment bound phosphorus from the impoundment. Within the impoundment, the accumulated sediments and the associated increases in TP have created eutrophic conditions which result in DO saturations that are harmful to fish and aquatic life. Further studies are needed to definitively determine if sediment bound contaminates leaving the impoundment negatively impact downstream water quality.

Further research is needed to determine the potential impacts of dam removal on beneficial and invasive species. Despite this, our methods are a good first step in determining if the dam provides the unintended benefit of acting as a barrier to invasive species. If the stream survey identified significant barriers to fish passage immediately upstream of the impoundment, it could be determined that the dam does not provide substantial protection against the habitation of invasive species in this watershed. This would prevent the need for further studies to identify sensitive upstream species, and to determine habitat suitability for species blocked by the dam. The addition of continuous monitoring of WT and DO, and an increased sampling frequency of TP, $NO_2^-$ + $NO_3^-$, TKN, and TN would improve the assessment of the dam's impact on water quality. The water quality parameters selected for this study were effective at determining this dam's impact on water quality; however, future studies should consider local water quality issues and concerns to determine the best suite of parameters to assess the dam of concern.

This study focused on the potential unintended benefits of improving water quality, providing recreational opportunities, and providing a barrier to invasive species. These are potential unintended benefits that many dams may provide; however, this is not an exhaustive list of potential unintended benefits. Future studies should carefully consider the potential unintended benefits the dam of concern may be providing.

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