4-2012

The Effect of Urban Runoff on the Dew Drop Pond and the Potential for Restoration

Kaitlyn J. Nielson
St. Catherine University, kjnielson@stkate.edu

Recommended Citation

Nielson, Kaitlyn J., "The Effect of Urban Runoff on the Dew Drop Pond and the Potential for Restoration" (2012). *Antonian Scholars Honors Program.* Paper 19.
http://sophia.stkate.edu/shas_honors/19

The Effect of Urban Runoff on the Dew Drop Pond and the Potential for Restoration

Kaitlyn Nielson

St. Catherine University

A Senior Project in Partial Fulfillment of the Requirements of the Honors Program

April 2, 2012

Abstract

The Dew Drop pond, located on the St. Catherine University campus in St. Paul, Minnesota has two functions: (a) to be a beautiful area to the campus for students' enjoyment and (b) the filtration and removal of nutrients and sediment from runoff before it is transferred to the Mississippi River. This study was conducted to evaluate the effect of runoff from the St. Catherine University campus has on the pond, which is filling in and experiencing summer algal blooms. Sediment cores were collected from varying areas of the pond (west buffer (n=4), east buffer (n=3), no buffer (n=3), east island (n=2), west island (n=2)) and segmented into two or three layers based on visual distinction of texture and color in the sediment. Samples were extracted using 2M KCl and 0.5M $NaHCO_3$, shaken 1 hour, and filtered using Whatman 42 filter paper. Extracts were analyzed for ammonium (NH_4-N), nitrate (NO_3-N), soluble reactive phosphorus (SRP), dissolved organic carbon (DOC), and total dissolved nitrogen (TDN). Following statistical analyses of one-way ANOVA, results show significantly higher concentrations of NH_4-N and TDN in the east island region (29404.71mg/kg ($p<0.01$), 32248.15mg/kg ($p<0.05$) respectively), of SRP in the east buffer region and the middle layer of sediments (2236.50mg/kg ($p<0.01$), 1678.32mg/kg ($p<0.05$) respectively), and of DOC in the top layer of the sediments (36.91mg/g ($p<0.05$)). Though no statistical difference was present for NO_3-N data, values were much lower than NH_4-N. Comparison to similar studies performed in river floodplains, wetlands, and retention basins demonstrate that the Dew Drop pond has high nutrient concentrations, indicating the potential for continuation of eutrophication, and this is likely decreasing its ability to efficiently store water and remove nutrients from the runoff before entrance to the river. Recommendations include the dredging of the pond to remove nutrient stored sediments, as well as planting a sustainable and effective buffer zone of native, perennial grasses and shrubs to limit the runoff of excess nutrients into the pond and erosion of the banks.

Introduction

St. Catherine University – Dew Drop Pond

For many who visit, the St. Catherine University campus has the beauty and quiet of a small town in the middle of the big city of Saint Paul, MN. One of the highlights of the campus is in front of the magnificent chapel, where a beautiful wooded area and the Dew Drop pond are located. The pond serves as a place of comfort and serenity to the students and staff of St Catherine University. Unfortunately, the pond is experiencing a decline in water quality and an increase in sedimentation detracting from the beauty it once held.

Figure 1. Aerial photo of Dew Drop Pond, St. Paul, MN. Photo was taken in March 2006. IN = drain inlet; IN – spring = freshwater spring inlet; EXIT – exit stormwater drain.

Built and expanded during various building projects throughout the early additions to the campus, including Our Lady of Victory Chapel (1923), Fontbonne Hall (1931), and the Library (1960), the small pond is located at the bottom of a hill on the west side of the campus (CSC President 1931, 1933, 1961). The pond originally received its water supply from drains flowing from areas across the campus (Fig 1). A campus pool was also drained in 1994 and the drainage was directed to the pond (J. Manship, personal communication). During the addition of the library, which lies uphill to the southeast from the pond, a freshwater spring was discovered. The pond currently receives water from the spring as well as accepts drainage from the campus and runoff down the hill (Fig 1). There is currently one exit route for the water on the west bank of the pond, where it drains into the city storm sewer system for approximately 0.5 miles before emptying into the Mississippi River (P. Nipp, personal communication).

The Dew Drop pond functions as water filter system prior runoff to draining into the river. Like other retention basins and human constructed wetlands (Coveney et al. 2002), the pond acts to filter and retain nutrients and algae out of the runoff water before it enters the Mississippi River (White et al. 2000). The pond also acts as a retention basin for water, keeping our campus runoff out of the streets (P. Nipp, personal communication). The observed increase in sedimentation over the past few years and possible buildup of nutrients indicates that Dew Drop Pond is functioning properly in retaining nutrients and sediment from entering the Minnesota river system. Unfortunately, as time goes on, the ability of retention basins to properly filter nutrients and still be deep enough to provide a space for rainwater decreases (White et al. 2000, Lewitus et al. 2007). The severe decline in the appearance and increase in nutrient content of Dew Drop may be the signal to restore the pond to its proper functionality.

In both 2000 and 2004, St. Catherine biology students studied the pond during the summer months (Rosnow and Major 2000, oral presentation unreferenced, Kuhn 2004, oral presentation, unreferenced). While both studies found that the water column had relatively low levels of nitrogen and phosphorus, the pond had an active fountain to move and aerate water through the year 2010. In the summer of 2011, the large fountain in the southwestern area of the pond was not turned on due to the sediment having risen too high for the fountain to function properly. This loss had a substantial impact on the ability of the pond to be aerated, possibly allowing for more internal loading of nutrients. The sudden abundance of cattails in the pond and their subsequent death and decomposition may also be contributing high quantities of organic carbon into the pond, which is then available to microbial activity and may reduce oxygen further. The previous studies by St. Catherine students also both found a low invertebrate species diversity, indicating a low water quality.

Water Quality

The water quality in shallow bodies of water, such as the Dew Drop Pond, is often a direct result of the nutrients present in the sediment and the water column (Carpenter 2005). While nitrogen is widely considered the limiting nutrient in many ecosystems (Downing and McCauley 1992), when nitrogen is in plentiful supply, phosphorus is often the growth limiting nutrient (Daniels and Gilliam 1996). In instances where limiting nutrients are in excess, the eutrophication process starts to take effect. Eutrophication is the overenrichment of a water body by mineral nutrients and is characterized by blooms of primary producers, cyanobacteria (blue-green algae) and algae, which grow quickly and efficiently in nutrient-rich water (Kalff and Knoechel 1978). Algae die and are decomposed by bacteria, reducing water quality by increasing turbidity and decreasing oxygen levels (Chorus and Bartram 1999). Observations of freshwater lakes under natural conditions and those under experimental nutrient enrichment to test the effects of excess nutrients have shown a decrease in water quality, often characterized by increased turbidity, increased algae populations, and a water infused sediment bottom, with the cause identified being nutrient (often phosphorus) excess (Bronmark and Hansson 1998; Sondergaard et al. 2003).

The long term effect of nutrient loading is eutrophication (Bronmark and Hansson 1998, Nayar et al. 2007). Overenrichment starts a chain reaction, beginning with an increase in the number of primary producers (algae) which then dominate the community. The algae die and are decomposed by bacteria, converting the organic material to inorganic nutrient forms (including carbon dioxide, and inorganic nitrogen and phosphorus) and using oxygen in the process, resulting in oxygen depletion (Chorus and Bartram 1999, Susana et al. 2008, Downing and McCauley 1992, Bronmark and Hansson 1998). Long-term aquatic ecosystem effects due to this domination include a decrease in biodiversity, a loss of aquatic plants and other wildlife, and a loss of aesthetic and recreational appeal (Sondergaard et al. 2003).

Nitrogen and phosphorus are present in lakes and ponds in both the water column or in the sediment and frequently move between the two mediums (Sondergaard et al. 2003, Nayar et al. 2007). Unlike nitrogen, which can exist as N_2 gas, produced under low oxygen conditions in the conversion of nitrate to nitrogen gas (Campbell et al. 2008), phosphorus does not have an atmospheric form. Although called a cycle, the phosphorus cycle ends with drainage to a body of water. During transport from source to sink, it can settle into sediment or be incorporated into organic forms of life (Schindler and Vallentyne 2008). The drainage of nutrients into water bodies from watershed runoff is known as external loading. The transfer of settled nutrients, such as phosphorus, into the water column is known as internal loading, from as deep as approximately 20 cm below the lake bed (Schindler and Vallentyne 2008, Sondergaard et al. 2003, Carpenter 2005). Even when external loading of phosphorus is reduced, internal loading can play a large role in decreasing water quality, especially if the body of water is rich in phosphorus (Sondergaard et al. 2003, Carpenter 2005). Internal loading involves the recycling of nutrients within a system. The amount of phosphorus released into the water column can often be a result of the oxygen concentration in the sediment. In anaerobic sediments, a reducing environment is created in the sediment, leading to the reduction of Iron (III) to Iron (II). As this reduction occurs, the phosphate previously bound to Iron (III) is released as soluble inorganic phosphorus and enters the water column (Patrick and Khalid 1974). In aerobic conditions, the reverse occurs, and sediment compounds are oxidized and Iron (II) returns to the Iron (III) state, allowing for phosphate to bind and the compound becomes insoluble and is resorbed into the sediment (Chorus and Bartram 1999, Patrick and Khalid 1974).

Nutrient overenrichment is most often a result of runoff from the surrounding watershed. Runoff is the carrier of both sediments and nutrients attached to sediment particles (Susana et al. 2008, Craft. 1997). Phosphorus, along with nitrogen and potassium, is a key component of fertilizers. Runoff from agricultural areas, as well as from construction sites and developed urban communities, is the highest contributor of nonpoint pollution to water bodies (Rivas et al. 2000, Schindler and Vallentyne 2008, Daniel et al. 1998). Nutrients can also come from animal excretions if the area is heavily inhabited.

Previous studies have suggested that waterfowl populations (geese, ducks, swans, etc.) contribute up to 40% of the external nitrogen input and 75% of the external phosphorus input into wetland areas each year (Manny et al. 1975 and Marion1994, as cited by Unckless and Makarewicz 2007). Nutrient loading is also often correlated with sedimentation of a lake bottom.

Sediment and Nutrients

Runoff can transport sediment, nutrient, and organic matter particles to holding places, but after summer algal blooms due to increases in nutrients, the organic matter input and algae are broken down by bacteria, which gives affected water bodies a distinct murky appearance due to the increased organic material suspended in the water (Bronmark and Hansson 1998, Schindler and Vallentyne 2008). Sediment also enters water from bank erosion caused by runoff or by animals entering and exiting the lake. Without a strong riparian plant buffer zone surrounding the lake to protect it from runoff supplied by the surrounding landscape, sedimentation rates can increase (Schindler and Vallentyne 2008, Matson 1997). The holding capacity of the lake is then decreased until it is filled in with sediment and organic matter (Nayar et al. 2007). This external loading of sediment and nutrients can be mitigated in many different ways. Buffer zones (vegetation in the riparian area of a water body) are effective ways of reducing sediment and nutrient inputs (Mankin et al. 2007). The vegetation allows for uptake of incoming nitrogen and phosphorus, and providing a dense network that slows the velocity of runoff flow, decreasing sedimentation by allowing more sediment particles to settle prior to entrance into the water body. Nutrient and sediment filtration occurs within the first 10 m of the buffer zone vegetation and has been shown to reduce sediment and nutrient loading. A combination of decreasing the pollutant load that enters the watershed and implementing effective riparian buffer zones by both increasing the width of vegetation to at least 8 m but also using nutrient retaining tall grasses and shrubs (native to Minnesota to alleviate the possibility of exotic invasion) along the bank that an act to limit the entrance of sediment and nutrients into water bodies (Daniels and Gilliam 1996, Matson, 1997, Nayar et al. 2007). The most effective vegetation in this process are switchgrass and fescue species of grasses (Mankin et al. 2007, Blanco-Canqui et al. 2004).

Internal loading can be a problem for small lakes and ponds, even when all tactics to prevent external loading from runoff have been executed (Daniels and Gilliam 1996, Sondergaard et al. 2003, Rasumssen and Ceballos 2009). The implementation of an aeration system can limit the solubility of some nutrients, including phosphorus, from sediments by introducing oxygen into the environment which will keep them bound to oxidized metals (Daniels and Gilliam 1996, Matson 1997, Nayar et al. 2007). However, one of the only effective ways to mitigate the detrimental effects of internal loading is to take the saturated sediment out of the ecosystem by dredging, which acts not only to remove unwanted nutrients, but also to deepen the water body (Peterson 1982, Omega Lake Services), which allows it to recover.

Study Aims

In the time since previous expansions of the Dew Drop Pond attempting to widen and deepen it, other campus building projects have taken place, including the construction of the Coeur de Catherine (2003) and two residence halls to the southeast (2007) (Fig 2). lthough damaging phosphorus based fertilizers have been used in the past, they have not been used on campus since 2002. A potentially effective (6m wide) vegetated buffer zone was planted in 2002 to reduce loading and reduce

Figure 2. Aerial photo of St. Catherine University campus, St. Paul, MN. Photo taken in 2006. 1=Chapel. 2=Coeur de Catherine. 3=Library. 4=2007 Residence Halls. 5=Dew Drop Pond. Compass rose in the top left corner.

waterfowl erosion, but the pond is still filling in (P. Nipp, personal communication). Only strongly protected from the west, nothing but a steep drop exists on the southern bank, while the eastern bank includes some tall grasses and the northern bank has a grassy slope. The pond is heavily occupied by geese and ducks year round and the waste runs into the pond. Currently, an aeration system is in effect for the east side of the pond, between the island and the east bank. Unfortunately, due to decreasing depth, the fountain intended to aerate the western side was not used in the summer of 2011 (J. Manship, personal communication). In recent years, the Dew Drop Pond had increased in summer algal blooms in parallel with a decline in visual appeal.

As the greatest nutrient source for the pond is campus runoff and possibly from waterfowl excretion, the purpose of this study was to determine the potential effect of nutrient inputs by external and internal on the Dew Drop Pond by quantifying the accumulation of nutrients in the sediment, and to determine if these accumulations may correlate to the eutrophication of and visual decline in the pond. By determining the amounts of nitrogen and phosphorus stored in Dew Drop Pond sediments and comparing to previous studies of similar ecosystems, I will determine if the pond is in need of an intervention (e.g. dredging) to remove excess sediment, in combination with the planting of a more effective (8 m) perennial grass and shrub buffer zone to filter nutrient excesses by sequestering nutrients and slowing flow of sediment. This will aid in returning Dew Drop Pond to both a beauty of St. Kate's, but also to an effective filter and storage mechanism for sediment and nutrients for the Mississippi River. Following this study, should a threat to the future aesthetic and biodiversity elements of Dew Drop Pond be threatened, the possibilities for restoration will be considered.

Methods

Study Site

The area of study for this research is the Dew Drop Pond, in St. Paul, MN. The pond is human made and can first be seen in a photos during the time of the building of Our Lady of Victory Chapel in 1923. As St. Catherine University, then called College of St. Catherine, began to expand, the pond was enlarged during the building of Fontbonne Hall in 1931 and the Library in 1960. It has undergone multiple landscaping projects (in 1934, 1955, and2002) in order to expand the pond or build a new bridge. The pond sits at the bottom of the hill on the west end of the campus (Fig 2). Sources of water include drainage from all areas of campus, as well as a fresh water spring located beneath the library, located up the hill to the southeast. Currently, water leaves the pond by one exit and travels through the storm sewer system leading to the Mississippi River, located approximately 0.5miles west. Near the east bank of the pond is an island with a wooden bridge connecting it to the bank. Currently, no vegetated buffer zone exists along the south side, but the north and west banks are dominated by cattails. The east bank offers some buffer vegetation. The pond area is occupied by geese and ducks year round. An aeration system has been used in the past on the east side, between island and east bank

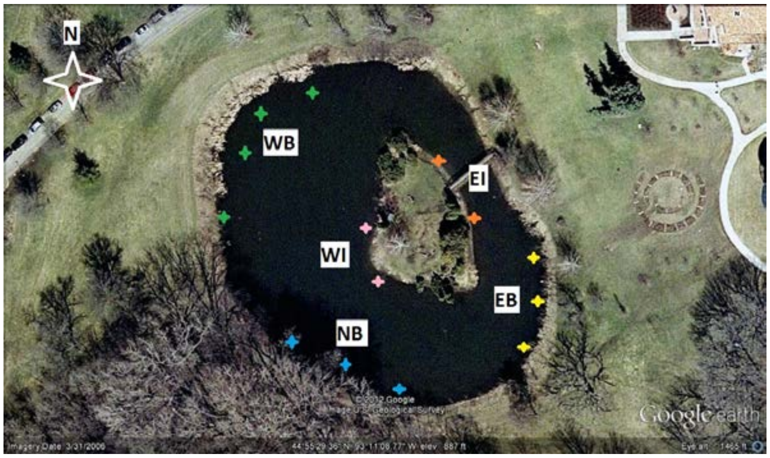

Figure 3. Aerial Photo of Dew Drop Pond, St. Paul, MN. Photo was taken March 2006. Dots indicate sites of coring. Abbreviations represent regions. WB = west buffer; NB = no buffer; WI = west island; EB = east buffer; EI = east island

Data Collection

A total of 14 sites in the pond were randomly selected and sediment cores were collected and placed into bags for transport. The number of cores collected from each region of the pond was determined based on the areas a particular region covered (Fig 3). Areas included: west buffer (4), east buffer (3), east island (2), west island (2), and no buffer (3). Sediment cores ranged from 18-28 cm in depth and were sectioned into three segments based on visual difference existing in the sediment through the depth. The top layer was primarily sand, whereas the middle and bottom layers were much finer, and the middle layer showed a slightly lighter color. Segmented lengths were similar between cores. Two cores (west buffer, 1; east island, 1) did not have complete middle layer distinction from the top and bottom layers and were therefore sectioned into two sections. Sediment bags were transported to the lab, weighed, and stored at 4°C until extraction.

Sediment subsamples were extracted using both 0.5M $NaHCO_3$ (10 g wet sediment in 100 mL) and 2M KCl (10 g wet sediment in 50 mL) for use in subsequent analyses. Sediments were shaken in solution for 1hr and filtered using Whatman 42 paper. Filtered extracts were kept frozen until nutrient analyses could be completed. $NaHCO_3$ extracts were used in soluble reactive phosphorus (SRP) analysis, while KCl extracts were analyzed for ammonium (NH_4-N), nitrate (NO_3-N), total dissolved nitrogen (TDN) and dissolved organic carbon (DOC). All bottles and glassware were acid washed in 10% HCl to avoid contamination. Extractions were then frozen until later analysis. Eight samples prepared for ammonium analysis from the west buffer region were not analyzed due to lack of a reagent necessary for analysis. Analyses were performed at St. Catherine University and St. Olaf College.

Ammonium nitrogen was analyzed fluorometrically (Turner Designs Trilogy) using analytical method of Holmes et al (1999). Soluble reactive phosphorus was measured colorimetrically using a spectrophotometer (Thermo AquaMate) at 880 nm. Nitrate nitrogen was measured colorimetrically (Lachat QuickChem 8600 flow injection analyzer). Dissolved organic carbon and total dissolved nitrogen were measured by combustion (Shimadzu TOC-VCSN Analyzer).

Statistical Analysis

Nutrient concentrations were determined by the use of standard curves and were organized by both layer of sediment (top, middle, bottom) and by region from which each core was taken (west island, east island, no buffer, east buffer, west buffer). One-way analysis of variance (ANOVA) analyses were completed using SPSS software to determine if significant differences between nutrient concentration means among regions and layers of sediment existed. Previous studies were used for comparison, although raw data were not available, so these were not subjected to ANOVA testing.

Results

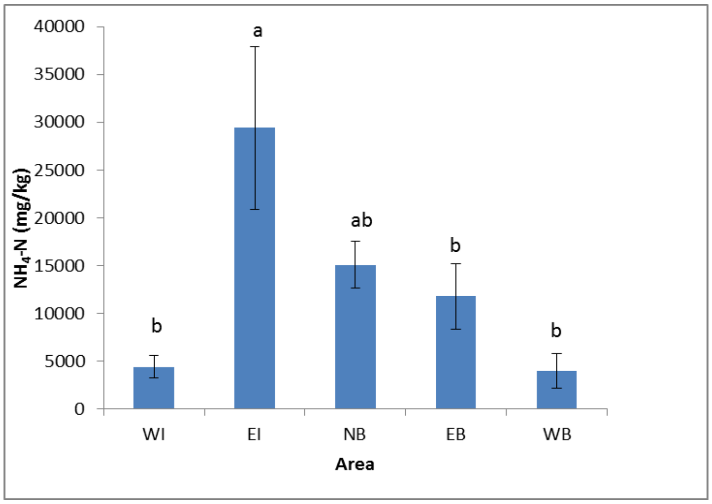

Figure 4: Core placement analysis of NH_4-N storage concentration in sediments (mg/kg). Sediment cores were collected from varying areas of Dew Drop pond in mid-October (2011) in St. Paul, MN and separated into two or three segments based on differences in visual appearance of sediment. Regions include: WI (west island, n=6), EI (east island, n=5), NB (no buffer, n=9), EB (east buffer, n=9), and WB (west buffer, n=3). Samples were grouped based on region, extracted with 50 mL of 2 M KCl, shaken 1hr, and filtered with Whatman 42 paper. Extractions were analyzed using a fluorometer and are reported as mg N/kg wet sediment. Each column represents mean concentration based on region. Error bars indicate ±1 S.E. Letters denote statistically significant differences $p<0.05$ (one-way ANOVA).

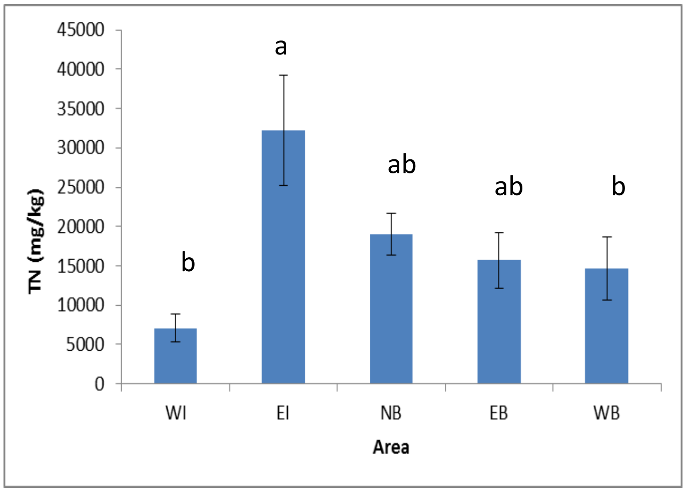

Figure 5: Core placement analysis of TDN storage concentration in sediments (mg/kg). Sediment cores were collected from varying areas of Dew Drop pond in mid-October (2011) in St. Paul, MN and separated into two or three segments based on differences in visual appearance of sediment. Regions include: WI (west island, n=6), EI (east island, n=5), NB (no buffer, n=9), EB (east buffer, n=9), and WB (west buffer, n=11). Samples were grouped based on region, extracted with 50 mL of 2 M KCl, shaken 1hr, and filtered with Whatman 42 paper. Extractions were analyzed by combustion and are reported as mg N/kg wet sediment. Each column represents mean concentration based on region. Error bars indicate ±1 S.E. Letters denote statistically significant differences $p<0.05$ (one-way ANOVA).

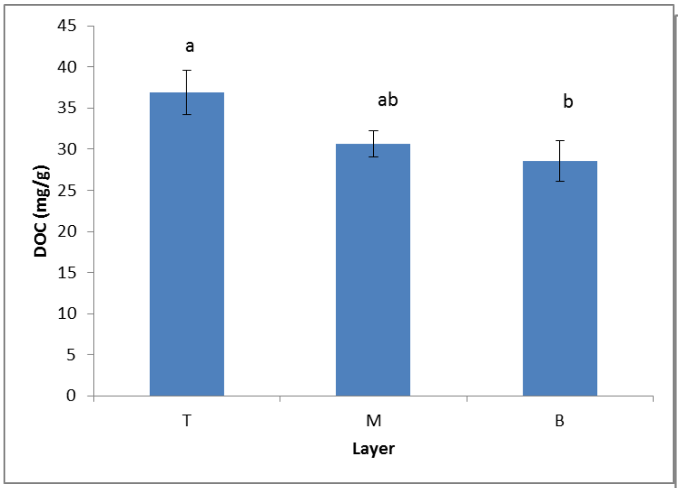

Figure 6: The effect of sediment layer (top, middle, bottom) on DOC storage concentration (mg/g) in Dew Drop pond sediments. Sediment cores were collected from Dew Drop pond in mid-October (2011) in St. Paul, MN and separated into two or three segments based on differences in visual appearance of sediment. Layers include: T (top, n=14), M (middle, n=12), and B (bottom, n=14). Samples were grouped based on layer, extracted with 50 mL of 2 M KCl, shaken 1hr, and filtered with Whatman 42 paper. Extractions were analyzed by combustion and are reported as mg DOC/g wet sediment. Each column represents mean concentration based on region. Error bars indicate ±1 S.E. Letters denote statistically significant differences $p<0.05$ (one-way ANOVA).

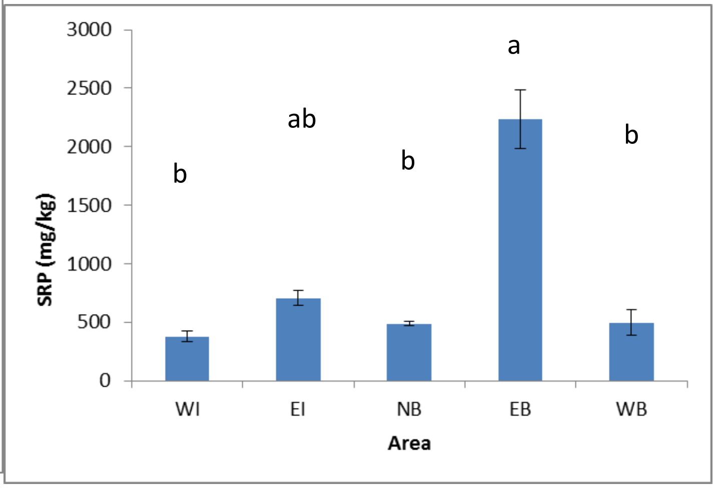

Figure 7: Core placement analysis of SRP storage concentration in sediments (mg/kg). Sediment cores were collected from varying areas of Dew Drop pond in mid-October (2011) in St. Paul, MN and separated into two or three segments based on differences in visual appearance of sediment. Regions include: WI (west island, n=6), EI (east island, n=5), NB (no buffer, n=9), EB (east buffer, n=9), and WB (west buffer, n=14). Samples were grouped based on region, extracted with 100 mL of 0.5 M $NaHCO_3$, shaken 1hr, and filtered with Whatman 42 paper. Extractions were analyzed using a spectrophotometer at 880nm and are reported as mg SRP/kg wet sediment. Each column represents mean concentration based on region. Error bars indicate ±1 S.E. Letters denote statistically significant differences $p<0.05$ (one-way ANOVA).

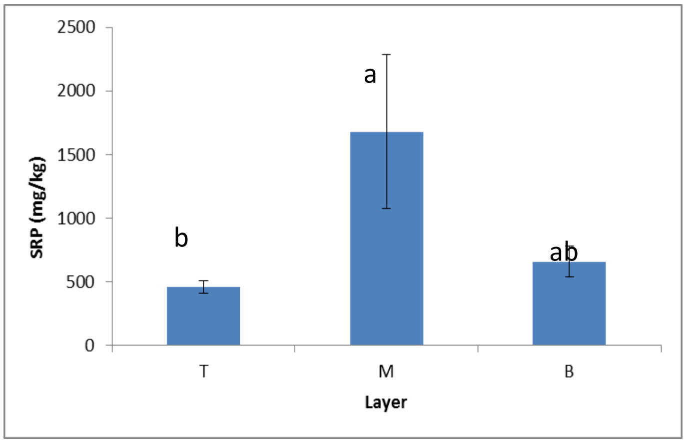

Figure 8: The effect of sediment layer (top, middle, bottom) on SRP storage concentration (mg/kg) in Dew Drop pond sediments. Sediment cores were collected from Dew Drop pond in mid-October (2011) in St. Paul, MN and separated into two or three segments based on differences in visual appearance of sediment. Layers include: T (top, n=14), M (middle, n=12), and B (bottom, n=14). Samples were grouped based on layer, extracted with 100 mL of 0.5 M $NaHCO_3$, shaken 1hr, and filtered with Whatman 42 paper. Extractions were analyzed using a spectrophotometer at 880nm and are reported as mg SRP/kg wet sediment. Each column represents mean concentration based on region. Error bars indicate ±1 S.E. Letters denote statistically significant differences p<0.05 (one-way ANOVA).

Table 1. A comparison of ammonium nitrogen and nitrate nitrogen storage concentrations (mg/kg) by region of Dew Drop Pond. A representation of nitrate nitrogen present by region of Dew Drop Pond. Asterisks denote statistical differences between means, and only occur in ammonium nitrogen data (p<0.05, one-way

Location	NH_4 (mg/kg)	NO_3 (mg/kg)
WI	4391.91364*	278.856142
EI	29404.7055*	241.534891
NB	15097.136	259.456516
EB	11791.7255*	212.197801
WB	3980.02*	226.837233

Nitrogen

There was no statistically significant difference in ammonium nitrogen (one-way ANOVA, p>0.05, data not shown), nitrate nitrogen (one-way ANOVA, p>0.05, data not shown), or total dissolved nitrogen (one-way ANOVA, p>0.05, data not shown) existed between layers of sediment however, the nutrient storage was significantly higher in the east island region than the west island, east buffer, and west buffer regions for ammonium nitrogen (Fig. 4, one-way ANOVA, p<0.01), and higher in the east island region than the west island and the west buffer regions for total dissolved nitrogen (Fig. 5, one-way ANOVA, p<0.05). Nitrate nitrogen values were not statistically significant different among regions, although were much lower than ammonium nitrogen in magnitude (Table 1, one-way ANOVA, p<0.05).

Dissolved Organic Carbon

There was also no statistically significant difference in dissolved organic carbon (DOC) sediment storage between pond areas (one-way ANOVA, p>0.05, data not shown); yet DOC was significantly higher in the top layer than in the bottom layer of the sediment (Fig. 6, one-way ANOVA, p<0.05).

Phosphorus

Soluble reactive phosphorus (SRP) concentration was significantly higher in the middle layer of the sediment than in the top layer (Fig. 8, one-way ANOVA, p<0.05) and was also significantly higher in the east buffer region than in the west island, no buffer, and west buffer regions of the pond (Fig. 7, one-way ANOVA, p<0.01).

Discussion

Location	TN(mg/kg)	SRP(mg/kg)	DOC(mg/kg)
Dew Drop	16952.7	882.1	32.1
Lake Yue	5178	1100	1.1
Lake Wuli	1884	600	0.7
Mai Po	2060		
Ontario Guidelines	4800		

Table 2. A comparison of TN, SRP, and DOC concentrations (mg/kg) between study location Dew Drop pond and similar studies, organized by location. Lake Yue and Lake Wuli are hypertrophic lakes in China. Mai Po is a wetland marsh located in China and characterized by high pollution inputs. The Ontario Guidelines provide a basis for sediment concentrations of SRP before it starts to affect the health of the ecosystem (Wang et al. 2011, Varol and Sen 2012).

Nutrient inputs may be to blame for the observed visual decline and possible decrease in ability of the Dew Drop pond to function as a filter prior to water entering the Mississippi River. Nitrogen in nitrate was lower in comparison to both ammonium and total nitrogen concentrations (Table 1). This is likely due to the process of denitrification, which converts nitrate into nitrogen gas under low oxygen conditions, allowing it to escape the system atmospherically (Campbell et al. 2008). The highest sediment storage (mg/kg) of ammonium was found in the east island region of Dew Drop Pond, and the lowest amount was found in west island and west buffer regions (Fig 4). This was also true of the total dissolved nitrogen content in pond sediments (Fig 5). The drain lines into the pond are located on the southeast, east, and north east banks (Fig 1). The east and northeast pipe drains take runoff from the majority of the campus and empty it into the pond. The layout of the pond and the island allows for catchment of compounds and debris on the east side of the island without proceeding around the island to the outlet drain. Studies of other wetlands have confirmed the accumulation of nutrients in sediments to be near inflows of the area (White et al. 2000, Lau and Chu 2000). This prevents nutrients from getting around to the west banks of the pond or around to the west side of the island. It is also worth noting that the east side of the island is where recent cutting of bank vegetation occurred and therefore vegetation available for nutrient uptake is limited, leaving nutrients to settle into the sediment below the water column. Studies conducted in wetland reserves in China have documented increases in pollution pressures due to the heightened urbanization occuring around them. Lau and Chu (1999) noted that an area of the wetland had a sediment total nitrogen load of 1,023 mg/kg, which was reported to be double the concentration of total nitrogen in what was considered a “background wetland” characterized as being relatively rural and therefore not affected by pollution to the same extent as urban areas. Studies on the Tigris river have documented similar results, with total nitrogen ranging from 700-2,664mg/kg (Varol and Sen 2012). The Ontario Guidelines for healthy waters state that total nitrogen above 4,800mg/kg is considered highly polluted and could be damaging to ecosystem. The average total dissolved nitrogen content for Dew Drop sediments was approximately 17,000mg/kg, indicating that a signifucant amount of nitrogen is stored in the sediment (Table 2). It should also be noted that all outside study data are reported in terms of mass per dry weight of sediment, whereas the concentrations presented in this study are mass per wet weight of sediment. This is important to note as mass per wet mass of sediment is a lower estimation of nutrient concentration due to the division over a higher value.

Dissolved organic carbon was not significantly different among pond areas (one-way ANOVA, $p>0.05$, data not shown), but there was a high degree of variability found in west buffer region. This region is curretly overrun with cattails, as is the east buffer region of the pond and may be contributing organic carbon to the pond sediments across regions (Fig 3). This was the area of the pond that was herbicided in the attempt to remove the cattail population, which may be influencing the rate of decomposition. Algal blooms from the summertime are also decomposing within the pond, adding to the deposition of organic carbon. The top sediment layer across the pond showed a significantly higher load of dissolved organic carbon than the bottom layer (Fig 6, $p<0.05$). This result could indicate the difference between time periods when heavy vegetation was not present, which has only intensified in the past ten years (P. Nipp, personal communication). The cattails have been arising in most areas of the pond due to their ability to expand once introduced into a new system, and their degradation may be contributing high amounts of carbon to the pond sediment. The average whole pond sediment dissolved organic carbon for Dew Drop was approximately 32.1g/kg. Lakes Yue and Wuli in China have been characterized as hypertrophic (a state of high eutrophication) lakes and have had recent studies that show sediment dissolved organic carbon concentrations of 1.06g/kg and 0.65g/kg, respectively (Table 2, Wang et al. 2011).

The highest sediment storage of phosphate was found in east buffer region, while the lowest concentrations were found in the no buffer, west buffer, and west island regions of the pond (Fig 8). These results are similar to those of the nitrogen sediment storage values (Fig 4, 5). The layout and positions of the input drains allow for catchment of debris on the east bank of the island without movement toward the other side of the island (Fig 1). This does not allow nutrients to settle in the west buffer or west island regions. Other studies have supported this phenomenon, showing that inflow regions of retention basins often have the highest concentrations of nutrients (White et al. 2000, Lau and Chu 2000). These results may also be indicative of aeration occuring in the east buffer region, causing the phosphate to be sequestered into the sediment in that area, as opposed to being released into the water column. The highest load of phosphate was found in the middle sediment layer, while the lowest occurred in top layer (Fig 7). The use of phosphorus based fertilizers ended in 2002 (P. Nipp, personal communication), which may explain the differences in phosphate content observed between layers. Studies conducted on highly polluted and hypertrophic lakes in China show inorganic phosphorus levels ranging fron 600-1100mg/kg (Wang et al. 2009), whereas the sediment in the Dew Drop pond contains approximately 880mg/kg of wet sediment (Table 2). This comparison shows the intensity of phosphorus settled in the sediment. Inorganic phosphorus was also measured in a drainage basin in New Zealand to determine the effects of dredging on sediment nutrient concentrations (Nguyen and Sukias 2002). Following a five year time period with no dredging while the sediment received storm and wastewater inputs, sediment accumulations of inorganic phosphate was found to be approximately 0.027mg/L. Three to six months following a dredging project, the sediment inorganic phosphate concentration was found to be approximately 0.0047mg/L. The Dew Drop sediments currently contain approximately 0.09mg/L phosphate, indicating a high concentration of phosphate located in the sediments (Table 3).

Location	**SRP** (mg/L)
Dew Drop	0.09
Australia 5 yrs Undredged	0.027
Australia 3-6 mo After Dredge	0.0047

Table 3. A comparison of inorganic phosphorus (IP) concentration (mg/kg) between study location Dew Drop pond and a drainage catchment in New Zealand. Concentrations of SRP (mg/L dry) from a previous study conducted in New Zealand are compared to Dew Drop pond (mg/L wet, Nguyen and Sukias 2002). The drainage basin receives storm and wastewater runoff from surrounding urban areas. The study aimed to learn the effects of dredging on sediment nutrient concentrations. The basin was left untouched for 5 years and then dredged. SRP was measured pre and post dredging.

The purpose of a human made retention basin is to filter nutrients and sediment from runoff water from a variety of land sources (agriculture, industrial, urban, construction, animal farming, etc.) and store runoff water. Studies indicated that wetlands filter nutrients from larger waters (Coveney et al. 2002, Datry et al. 2003, Wang et al. 2004, White et al. 2000), and the Dew Drop pond was expanded twice in order to serve this purpose for the Mississippi River The ability of a wetland retention basin to reduce the influx of nutrients into another water sink depends on the health and sediment load within the wetland. Slower water flow allows for more sediment to settle, whereas faster water movement associated with shallower waters tends to move accompanying nutrient particles to the larger body.

Even if external loading is severely reduced, as in the case of a Lake Yue in China characterized by intense algal blooms every summer, internal loading can keep a lake in a eutrophic state (Zhong et al. 2008). Dredging can be used to reduce internal loading by removing nutrient contaminated sediment. While dredging can release sediment and nutrients into suspension in the water column, it may take as few as three months for the sediment to settle back to the bottom (Palmer-Felgate et al. 2011). The flux of phosphorus from sediment to water was significantly reduced (90%) following the dredging process in Lake Yue, demonstrating the rehabilitation effects of drawing out nutrient enriched sediment (Zhong et al. 2008). Dredging would allow the pond to continue its function as a filter for nutrien, sediment, and organic matter for the Mississippi river and enhance its ability to retain stormwater. Dredging can have a positive effect on benthic communities, enhancing their diversity (Zhang et al. 2010).

This study has documented the abundance of nutrients in the sediments of Dew Drop pond. Given the high sediment nutrient loads overalll, the sedimentation, and the high organic matter inpus that have caused shallowing in the pond with sediment approximately 0.5-1.0 meters deep, there is need to remove nutrient contaminated sediments. Dredging is used to deepen the water and to allow for more diversity of aquatic organisms. It is also an effective means of removing excess sediment, nutrients, and organic matter from the body of water (Peterson 1982, Omega Lake Services). However, dredging will not be enough to completely eliminate algal blooms and sedimentation in Dew Drop pond. Proper bank vegetation and the use of aerators is essential for reducing external loading of nutrients from runoff (Daniels and Gilliam 1996). The St. Catherine University facilities staff has already ceased the use of phosphorus based fertilizers (Nipp, personal communication), but sedimentation and nutrient loading can still be caused by other sources, such as animal movement and runoff from waste products. Unfortunately, it may be a difficult task to reduce the numbers of geese and ducks in the area due to the fact that there is not another area such as this part of St. Catherine University campus for them to inhabit. However, a proper buffer zone would decrease runoff to the water (Daniels and Gilliam 1996) and it would also act to deter waterfowl from migrating into and out of the pond, decreasing soil erosion on the banks of the pond. I propose a removal of the cattails in order to expand the open water of the pond and decrease the amount of organic matter being contributed to the pond, coupled to the removal of sediment from the bottom, particularily in areas of intense nutrient concentrations, and the planting of vegetation on banks to reduce runoff and erosion. Vegetation along the banks should be at least 8 m in width and consist of perennial grass species and shrubs native to Minnesota to effectively remove nutrients from runoff and slow sedimentation. The most effective vegetation for nutrient removal include switchgrasses and fescue (Mankin et al. 2007, Blanco-Canqui et al. 2004). Following these improvements, the Dew Drop pond should be restored to the point where it can continue the task of successful and efficient removal of nutrients and sediment prior to water delivery to the Mississippi River. The pond will also be able to retain water in order to reduce the possibility of flooding in the neighborhood. The Dew Drop pond will also be ready for incoming students to enjoy without the intense, harmful blooms of algae that detract from the beauty of the St. Catherine University campus.

Acknowledgements

I would like to thank field volunteers (Hannah Beth Starr, Mary Benbenek, Sarah Nelson), as well as St. Catherine University faculty member Dr. Kay Tweeten for the use of her sediment core instrument. I would like to thank Hannah Kaup for her extensive assistance in the laboratory processes. St. Catherine University faculty and staff members of my Honors committee needing thanks include Dr. Martha Phillips, Dr. Lynne Gildensoph, Dr. Jill Welter, Dr. John Flynn, and Peter Nipp. Final thanks to St. Olaf College faculty Dr. John Schade for the use of his laboratory to complete core analyses.

References

Blanco-Canqui H, Gantzer C J, Anderson S H, Alberts E E, Thompson A L. 2004. Grass barrier and vegetative filter strip effectiveness in reducing runoff, sediment, nitrogen, phosphorus loss. Soil Science Society of America Journal 68:1670-1678.

Bronmark C, Hansson L. 1998. The Biology of Lakes and Ponds. New York (NY): Oxford University Press; p.216.

Campbell N A, Reece J B, Urry L A, Cain M L, Wasserman S A, Minorsky P V, Jackson R B. 2008. Biology. San Francisco (CA): Pearson, Benjamin Cummings; p. 1233.

Carpenter S R. 2005. Eutrophication of aquatic ecosystems: Bistability and soil phosphorus. Proceedings of the National Academy of Sciences 102 (29): 10002-10005.

Chorus I, Bartram J. 1999. Toxic Cyanobacteria in Water: A guide to their public health consequences, monitoring and management. London (Great Britian): WHO; p.253.

Coveney M F, Stites D L, Lowe E F, Battoe L E, Conrow R. 2002. Nutrient removal from eutrophic lake water by wetland filtration. Ecological Engineering 19:141-159.

Craft C B. 1997. Dynamics of nitrogen and phosphorus retention during wetland ecosystem succession. Wetlands Ecology and Management 4:177-187.

Daniel T C, Sharpley A N, Lemunyon J L. 1998. Agricultural phosphorus and eutrophication: A symposium overview. Journal of Environment Quality 27:251-257.

Daniels R B, Gilliam J W. 1996. Sediment and chemical load reduction by grass and riparian filters. Soil Science Society of America Journal 60(1): 246-251.

Datry T, Malard F, Vitry L, Hervant F, Gibert J. 2003. Solute dynamics in the bed sediments of a stromwater infiltration basin. Journal of Hydrology 273:217-233.

Downing J A, McCauley E. 1992. The nitrogen : phosphorus relationship in lakes, Limnological Oceanography 37(5):936-945.

Holmes, R.M, A. Aminot, R. Kereuel, B.A. Hooker, and B.J. Peterson. 1999. A simple and precise method for measuring ammonium in marine and freshwater ecosystems. Canadian Journal of Fisheries and Aquatic Science. 56, p. 1801-1808.

Kalff J, Knoechel R. 1978. Phytoplankton and their dynamics in oligotrophic and eutrophic lakes, Annual Review of Ecological Systematics 9:475-495.

Lau S S S, Chu L M. 1999. Contaminant release from sediments in a coastal wetland. Water Research 33(4): 909-918.

Lau S S S, Chu L M. 2000. The significance of sediment contamination in a coastal wetland, Hong Kong, China. Water Research 32(2): 379-386.

Lewitus A J, Brock L M, Burke M K, DeMattio K A, Wilde S B. 2008. Lagoonal stormwater detention ponds as promoters of harmful algal blooms and eutrophication along the South Carolina coast, Elsevier: Harmful Algae 8:60-65.

Mankin K R, Ngandu D M, Barden C J, Hutchinson S L, Geyer W A. 2007. Grass-shrub riparian buffer removal of sediment, phosphorus, and nitrogen from simulated runoff. Journal of the American Water Resources Association 43(5): 1108-1116.

Manship J. 23 Feb 2012. Dew Drop Pond. [Personal Email] Accessed 23 Feb 2012.

Matson T. 1997. Earth Ponds Sourcebook: The Pond Owner’s Manual and Resource Guide. Woodstock (VT): The Countryman Press; p. 171.

Nayar S, Miller D J, Hunt A, Goh B P L, Chou L M. 2007. Environmental effects of dredging on sediment nutrients, carbon and granulometry in a tropical estuary. Environment Monitoring and Assessment 127(1-3):1-13.

Nguyen L, Sukias J. 2002. Phosphorus fractions and retention in drainage ditch sediments receiving surface runoff and subsurface drainage from agricultural catchments in the North Island, New Zealand. Agriculture, Ecosystems and Enviroment 92:49-69.

Nipp P. 16 Feb 2012. Honors Draft. [Personal Email]. Accessed 16 Feb 2012.

Omega Lake Services. 2008. The Pond Ecosystems. [Accessed 20 Feb 2012] Available from http://www.omegalakeservices.com/The_Pond_Ecosystem.html.

Palmer-Felgate E J, Bowes M J, Stratford C, Neal C, MacKenzie S. 2011. Phosphorus release from sediments in a treatment wetland: contrast between DET and EPC_0 methodologies. Ecological Engineering 37:826-832.

Patrick W H, Khalid R A. 1974. Phosphate release and sorption by soils and sediments: Effect of aerobic and anaerobic conditions. Science 186(4158):53-55.

Peterson S A. 1982. Lake restoration by sediment removal. Water Resources Bulletin 18(3): 423-433.

President, College of St. Catherine. 1931, 1933, 1961. CSC Administrative Reports, Report of the President.

Rasmussen T C, Ceballos E L. The effects of sediment removal on internal nutrient cycling and eutrophication in lake Allatoona. Proceedings of the 2009 Georgia Water Resources Conference.

Rivas Z, Medina H L, Gutierrez J, Gutierrez E. 2000. Nitrogen and phosphorus levels in sediments from tropical Catatumbo River (Venezuela). Water, Air, and Soil Pollution 117:23-37.

Schindler D W, Vallentyne J R. 2008. The Algal Bowl: Overfertilization of the World’s Freshwaters and Estuaries. Edmonton, Alberta, Canada: The University of Alberta Press; p. 334

Sondergaard M, Jensen J P, Jeppesen E. 2003. Role of sediment and internal loading of phosphorus in shallow lakes. Hydrobiologia 506-509(1-3):135-145.

Susana J, Juan P, Pablo C, Jorge C, Bernal G. 2008. Water quality and zooplankton composition in a receiving pond of the stormwater runoff from an urban catchment, Journal of Environmental Biology 29(5):693-700.

Unckless RL, Makarewicz JC. 2007. The impact of nutrient loading from Canadian geese (*Branta Canadensis*) on water quality, a mesocosm approach. Hyrdobiologia 586 (1):393-401.

Varol M, Sen B. 2012. Assessment of nutrient and heavy metal contamination in surface water and sediments of the upper Tigris River, Turkey. Catena 92:1-10.

Wang G, Liu J, Tang J. 2004. The long-term nutrient accumulation with respect to anthropogenic impacts in the sediments from two freshwater marshes (Xianghai Wetlands, Northeast China). Water Research 38: 4462-4474.

Wang S, Jin X, Zhao H, Wu F. 2009. Phosphorus release characteristics of different trophic lake sediments under simulative disturbing conditions. Journal of Hazardous Materials 161:1551-1559.

Wang S, Jiao L, Yang S, Jin X, Liang H, Wu F. 2011. Organic matter compositions and DOM release from the sediments of the shallow lakes in the middle and lower reaches of Yangtze River region, China. Applied Geochemistry 26:1458-1463.

White J S, Bayley S E, Curtis P J. 2000. Sediment storage of phosphorus in a northern prairie wetland receiving municipal and agro-industrial wastewater. Ecological Engineering 14:127-138.

Zhang S, Zhou Q, Xu D, Lin J, Cheng S, Wu Z. 2010. Effects of sediment dredging on water quality and zooplankton community structure in a shallow of eutrophic lake. Journal of Environmental Sciences 22(2):218-224.

Zhong J, You B, Fan C, Li B, Zhang L, Ding S. 2008. Influence of sediment dredging on chemical forms and release of phosphorus. Pedosphere 18(1):34-44.