WHEATON COLLEGE

Norton, Massachusetts

This is to certify that Anne Bennett has fulfilled the requirements the requirement for graduation with Departmental Honors in Environmental Science. The degree of Bachelor of Arts was awarded on May 17th, 2014.

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Small pond ecology: eutrophication, alternative stable states, and management options

BY

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Preface

This thesis was part of a collaborative effort. All fieldwork and data analysis was conducted with Sarah Moore. Together, we co-authored our methods and results sections. All other sections were written independently, though we shared references and discussed ideas.

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Abstract

Nutrient pollution is a growing problem for freshwater bodies around the world. Cultural eutrophication associated with nutrient pollution can lead to unhealthy ecosystems with a lack of oxygen and biodiversity. This study compares three small ponds in Massachusetts to determine their trophic status, measured by water chemistry and biological indicators. Based on these criteria, Gilmore and Peacock Pond have been identified as eutrophic systems, but Wildcat is not. Eutrophic ponds can enter one of two alternative stable states, and Gilmore and Peacock Ponds reflect each of these. Gilmore Pond is in a turbid, phytoplankton dominated state, and Peacock Pond is in a clear-water, macrophyte dominated state. Wildcat Pond lies in between these two extremes, with a moderate amount of both phytoplankton and macrovegetation. Management strategies to pull Gilmore Pond out of this turbid state by reestablishing bottom vegetation are described, but ultimately I would advise managers of Gilmore Pond against taking an active approach. Gilmore Pond is a satisfactory ecosystem, and the costs of management are not likely to outweigh the benefits.

1. Introduction (AB)

Eutrophication of a water body occurs when nutrients, specifically nitrogen and phosphorus, accumulate in the water column and bottom sediments. This process naturally occurs in lakes and ponds at a very slow pace as organic matter builds up during ecological succession. However, if eutrophication is accelerated through human activity it can become detrimental to ecosystems (Oertli et al. 2005). High nutrient levels promote blooms of photosynthetic life, which will eventually die and become food for aerobic bacteria. The proliferation of bacteria that follows can lead to decreased dissolved oxygen levels and a consequential drop in biodiversity (Carpenter et al. 1998). Cultural eutrophication is the term for nutrient pollution in an aquatic system caused by anthropogenic sources. Only recently have people begun to recognize the threat that this pollution is presenting to pond ecosystems (Oertli et al. 2005). Some major sources of nutrient loading into ponds include runoff from fertilizers, construction deposition, and leakage from septic systems (Carpenter et al. 1998, Velinsky 2004). These are examples of non-point sources and are much more difficult to identify and remedy than point sources, but are the major causes of eutrophication. Control depends heavily on stopping the external flow of nutrients into water systems, but even so internal cycling in the water system can make recovery difficult to attain (Carpenter et al. 1998).

1.1 Eutrophication and Water Chemistry

Water chemistry is of great importance to freshwater systems such as ponds. It has an effect on species composition and can influence which species might have competitive advantages over others to become dominant (Schindler 1974). Phytoplankton have an important presence in ponds as primary producers. The three major groups of phytoplankton are green algae, cyanobacteria, also known as blue-green algae, and diatoms. There are certain chemical compositions that would be optimal for each of these groups, and their relative abundances can reflect those preferences. All phytoplankton need nitrogen and phosphorus to survive, and their growth has been shown to correlate with the chemical concentrations of these nutrients (Soballe and Kimmel 1987). Nutrient levels have such a direct impact on phytoplankton populations because they generally exist in low concentrations in natural waters (Sze 1989). Therefore, the growth of phytoplankton populations is limited by their access to phosphorus and nitrogen for metabolic processes. On the other hand, if these nutrients are present in excess, populations can flourish. For example, ecosystem health guidelines for an oligotrophic lake dictates that it should have less than 0.01mg/L of phosphorus and less than 2 mg/L of nitrogen, correlating with phytoplankton production levels of 7 to 25 (gC)/(m^2d). By contrast, a eutrophic lake would have over 0.03 mg/L of phosphorus and over 5 mg/L of nitrogen, correlating with 350-750 (gC)/(m^2d) (Schindler 1974, Olem and Flock 1990). In this example, doubling the nutrient concentrations increased the carbon fixation rate of algae up to 30 times. However, these standards have been specifically established for lakes, which are, by definition, larger than ponds (Oertli et al. 2005). Sondergaard et al. suggest that nutrients might be of lesser importance to smaller bodies because of lower

yield of chlorophyll per unit of nitrogen or phosphorus (Sondergaard et al. 2005). This could mean that even with higher phosphorus levels, phytoplankton production levels would remain lower. Regardless of the water system, algal requirements for nitrogen, phosphorus, and carbon have been expressed in a ratio called the Redfield ratio as follows, 40C : 7N : 1P (Schindler 1974). This ratio expresses just how little nitrogen, and especially phosphorus, algae require for primary production. It is easy to see that in an environment where nitrogen or phosphorus are present in excess, phytoplankton populations have the great potential to multiply.

Phosphorus, specifically, has been identified as the most important nutrient in controlling eutrophication of freshwater systems (Correll 1998). It exists in various forms, but orthophosphate is the only type available for autotroph assimilation. Total phosphorus includes all forms of phosphorous, both organic and inorganic, found in the water column (Velinsky 2004). It is important to consider the total phosphorus inventory of a lake or pond when determining its eutrophication level, because under certain conditions particulate and dissolved phosphorus can be converted to orthophosphate (Correll 1998). These internal mechanisms that release P bound to sediments are influenced by environmental factors, such as low dissolved oxygen levels (Correll 1998). Bacteria in organic matter are largely responsible for creating conditions for the redox reactions that cause phosphorus in the sediments to be released back into the water through their metabolic activity. Under low dissolved oxygen, these redox reactions occur more readily (Wetzel 2001). This can create a positive feedback system, as low dissolved oxygen is a symptom of eutrophic waters, which would then become even more nutrient polluted due to the release of P from sediments as a result of hypoxia. On the

other hand, there are also certain conditions that act to keep phosphorus bound to sediments. Lakes with high iron concentrations have little release of P from sediments, while lakes with lower iron experience regular seasonal cycling (Smith and Schindler 2009). For this reason, it is important to consider the impacts of internal cycling when studying eutrophication of aquatic bodies.

Phosphorus is crucial to eutrophication because of how it stimulates algal growth. In a eutrophication study of Lake Washington, phosphorus was found to be the only factor that directly correlated with algal abundance (Schindler 1974). As a limiting nutrient, algae populations are able to reproduce with great success when phosphorus becomes widely available through pollution. Some cyanobacteria can benefit greatly from high phosphorus levels, because they grow heterocysts capable of nitrogen fixation from N_2 to usable NH_4^+ forms, excluding them from nitrogen limitations (Sze 1998). In a controlled study of the effects of phosphorus and nitrogen on eutrophication, both chemicals were added to lakes at various regimes to observe their effects. When phosphorus was added, without nitrogen being added as well, cyanobacteria populations were able to boom (Correll 1998). By adding phosphorus without nitrogen, nitrogen-fixing cyanobacteria secured a competitive edge over other algal species. The success of cyanobacteria in eutrophic systems can cause a reduction in biodiversity because they are able to outcompete other species unable to fix nitrogen, such as green algae and diatoms. If zooplankton are able to persist, they can play an important role in keeping the populations of these nitrogen-fixing cyanobacteria down, and put a pressure on their potential to dominate the ecosystem (Smith and Schindler 2009). Some cyanobacteria have the potential to create toxic environments for other organisms. Toxins produced

within the cells of cyanobacteria are released into animal predators and disrupt their neuron and liver function (Sze 1998). Toxic effects are not known to impact human health, but can be fatal for mammals, birds, and fishes, so control of these species becomes particularly important when trying to combat the dangers of eutrophication (Sze 1998).

Though limiting nutrients are important factors in classifying the productivity of a freshwater system, to consider them without other related factors would be overly simplistic (Schindler 1974). When predicting algal abundance based on phosphorus concentrations, other abiotic factors like residence time of the water body, water depth, and turbidity are also contributing factors (Correll 1998). History of a lake or pond can also influence its water chemistry and well as the seasonal conditions during which it is being assessed (Schindler and Fee 1974). This makes it difficult to set standards for proper nutrient concentrations for any given freshwater system, because conditions are constantly changing and differ based on unique hydraulic circumstances. Maintenance of appropriate nutrient concentrations in any lake is the product of its own geological, biological, and physical aspects, including properties such as species composition, alkalinity, nutrient concentrations and cycling, water clarity, and water renewal (Schindler 1977).

1.2 Ponds vs. Lakes

The focus of this study is specifically eutrophication of small ponds. Ponds have been largely neglected in research regarding freshwater bodies, and only recently have

they begun to be recognized and investigated as unique ecosystems, distinct from lakes, streams, and rivers (Boix et al. 2012, Oertli et al. 2002, Oertli et al. 2005). This is important to recognize in eutrophication assessment because aquatic systems will experience eutrophication differently (Correll 1998).

There is no clear-cut definition for a pond or established standards to differentiate them from lakes, but the fundamental distinction is based in size. Biggs et al. (2005) has defined ponds as being between one m^2 to two hectares in size and Oertli et al. (2005) added a depth-criteria of no more than eight meters at its maximum. Their shallowness maintains fairly regular temperature throughout the water column that changes with air temperature (Lee 1955). It also creates the potential for aquatic vegetation to grow throughout the system, typically with shorelines inhabited by vegetation, as well (Lee 1955, Oertli et al. 2005). The shallow properties of ponds are also likely to give them a longer residence time than lakes, as shallow lakes have a longer residence time than deep lakes (Olem and Flock 1990). Dissolved oxygen is highly variable throughout the day, with more being available in the daytime and then becoming largely depleted during the night (Lee 1955). A biomass pyramid for a pond would have a large base for plant-life, about 87%, with about 10% herbivore mass, and only 3% carnivore mass (Lee 1955). However, the community is dynamic and can change not only seasonally, but also daily because of variability in the shallow water column. A distinction between natural and man-made ponds does not appear to hold significance (Oertli et al. 2005).

The characteristics that set ponds apart from lakes indicate that their ecology should be distinct, as well (Table 1). In a study comparing lakes and ponds locally, researchers found that larger water bodies were able to support greater biodiversity

(Hamerlik 2013). In a study specifically measuring differences in biodiversity between ponds and lakes, Hamerlik et al. (2013) found bodies two hectares or less to demonstrate distinctly different species-area relationships than larger bodies. Lakes had a more significant positive correlation between size and diversity, while pond diversity was less dependent on size (Hamerlik 2013). However, ponds did show higher among-site diversity than lakes, and have been found to hold high aquatic biodiversity (Hamerlik 2013, Boix et al. 2012). Variability in biodiversity could be linked to the finding that ponds tend to be more variable in their physical and chemical makeup than lakes due to their small size (Schindler 1977). This would imply that species composition and also eutrophication processes are going to be different in ponds because these processes are largely influenced by physical and chemical aspects. There is still inadequate information to know how exactly pond processes might differ from similar processes occurring in lakes, but it seems unlikely that applying the same ecological standards for lakes and ponds would be satisfactory for eutrophication management.

Table 1. A comparison of lakes and ponds.

Lake	**Pond**
Large, approx. >2 acres	Small, approx.1-2 acres
Deep water, >8 m	Shallow water, <8 m
Low light penetration to depths	Light penetration to bottom throughout
Stratified waters	Un-stratified waters
Shorter residence time	Longer residence time
Positive relationship between species diversity and size	No relationship between species diversity and size

1.3 Biological Indicators of Eutrophication

Algae are often used as biological indicators to measure freshwater nutrient levels. Phytoplankton populations have a positive, linear correlation with phosphorus increase, and the Redfield ratio can be particularly useful to determine whether nutrients are available in adequate levels for growth (Soballe and Kimmel 1987, Schindler et al. 2008). However, this relationship between phytoplankton biomass and nutrient concentration has been found to be less strong in ponds, because of submerged vegetation and activity of large zooplankton (Teisser et al. 2012). Knowing the species makeup of the algal community is more telling, because the presence and abundance of certain groups can indicate different environmental conditions.

Cyanobacteria, green algae, and diatoms all flourish under different chemical parameters. Cyanobacteria's capacity for nitrogen fixation allows them to dominate freshwater systems when N:P ratios are low (Schindler 1977). Heterocyst formation is negatively correlated with dissolved inorganic N in the water, so they are easily able to outcompete diatoms and green algae that cannot fix their own nitrogen (Sze 1998, Smith 1983). However, cyanobacteria have no competitive edge in phosphorous competition, so when N:P ratios are high there is generally more equal balance of all phytoplankton groups (Smith 1983).

Species composition is also useful in studying the health of a pond, because different species flourish under different conditions. Palmer has identified certain species of algae to be indicative of clean water supplies, including *Staurastrum* and *Pinnularia* (Palmer 1959). He associates other species, such as *Euglena* (Euglenophyceae),

Oscillatoria (cyanobacteria), *Anabeana* (cyanobacteria), and *Microcystis* (cyanobacteria), with polluted waters (Palmer 1959). There have been many other species and genera of algae, since Palmer, that have been identified as common to ponds, lakes, eutrophic bodies, oligotrophic bodies, acidic waters, etc. (Wehr and Sheath 2003). If these species are present in a body of water, they can provide an indication of environmental conditions. Freshwater plankton communities also vary with seasonal succession (Hutchinson 1967, Wetzel 2001). Succession is largely driven by temperature, light penetration, and nutritional concentration (Hutchinson 1967). Algae vary in their optimum range for these conditions, and so with changing conditions, different species can proliferate.

Nevertheless, algae are not the only biological component of pond ecosystems and, therefore, should not be the sole indicators of ecosystem health. Especially because studies have shown ponds, specifically, to have a lesser association between nutrient levels and algae biomass, a whole-ecosystem, community structure evaluation is critical to assessing the state of a pond (Teisser et al. 2012, Schindler 2008, Shubert 1984). To create a standard method for surveying pond health, Oertli et al. (2005) identified five groups as principal: plants, Gastropoda (snails and slugs), Coleoptera (beetles), Odonata (dragonflies and damselflies), and Amphibia (amphibians). These groups are representative because they occupy different trophic levels within the pond, demonstrate a variety of dispersal techniques, and have some degree of information known about their environmental tolerance (Oertli 2005, Menetrey 2005). Ephemeroptera (mayfly) larvae have been used in eutrophication studies as well, because they are known to be sensitive to low dissolved oxygen levels (Menetrey et al. 2008). Aside from using biological

indicators such as these to determine the health of the ecosystem, it is also important to understand the various aspects of pond biology because any action taken to address eutrophication will inevitably impact other aspects of the food web.

This study focuses on the health of three small ponds in Massachusetts. Gilmore Pond is of critical importance to my study, because of concerns held by the pond's owners, and Wildcat and Peacock Pond are used as comparisons. The ponds were studied from late summer through the fall for different chemical and biological criteria. Upon comparing these data, the trophic state and ecological health of Gilmore Pond will be assessed and management options considered. I hypothesize that Gilmore and Peacock Pond will both be eutrophic systems, based on personal and community perception of these ponds, but that Wildcat Pond will not be eutrophic because it is a drinking water source and feeds into a lake that meets Clean Water Act criteria.

1.4 Management Strategies

Deciding how to manage a freshwater pond is the next step after assessing its trophic state. There are many different management options available that have been tested both in the scientific community and autonomously by pond and lake managers. These methods can be broadly grouped into physical, chemical, or biological control techniques. No one method is ranked highest among the others, as each case of eutrophication has to be managed uniquely based on the specific circumstances (Wagner 2004). However, understanding those circumstances can help managers to choose the

best technique for their pond. Careful treatment and understanding the repercussions of management action is critical to long-term success (Wagner 2004).

There is one course of action that has been widely acknowledged as the first step in reversing cultural eutrophication, and that is controlling the external loading of phosphorous into the water system (Lurling 2013, Hilt et al. 2006, Carpenter 1998, Schindler and Fee 1974). Without first reducing nutrient inputs, further management is not likely to yield high success, especially in the long-term. However, stopping inputs alone is often not enough for successful eutrophication reversal because it does not address internal cycling. Another element to remember in reviewing the following management plans is that they were largely created for the purpose of *lake* management. Gilmore Pond must be treated as a pond in its assessment as well as its management. Nonetheless, limiting phosphorus inputs is of utmost importance.

1.4.1 Physical

1.4.1.a Dredging

Dredging is a technique for eutrophication control when the nutrient concentrations are largely the result of internal recycling (Lurling 2013). There are several ways to dredge, for example wet vs. dry, but ultimately it involves removal of the bottom sediments that are rich with phosphorus and perpetuating the growth of algae (Wagner 2004, Olem and Flock 1990). Dredging can also have the effect of deepening a pond. This could change the dynamics of the ecosystem by altering the abiotic regimes of the water column and also act to dilute nutrient concentrations with the addition of pond

volume (Wagner 2004). Another effect of dredging is the possible exposure of an otherwise stifled seed bank (Hilt et al. 2006). If a pond has not had macrovegetation growth in several years, there is still the possibility of seeds from historic plant communities residing dormant in the bottom sediments. Dredging can expose these seeds to more ideal conditions and allow macrovegetation to reestablish, creating a photosynthetic competitor for dominant algal communities (Hilt et al. 2006).

After removing sediment, it has to be disposed of in a designated area. This area must be large enough to contain all of the nutrient-rich waters that will inevitably be removed with the spoils. Runoff from improper containment of removed sediment is a common problem associated with dredging that can be avoided with proper planning (Olem and Flock 1990). To be sure that the removed sediments are will not be hazardous, analysis for potentially toxic compounds such as heavy metals and chlorinated hyrdocarbons, must be carried out before dredging can begin (Olem and Flock 1990). Certain precautions and permitting is required if these materials are found in high concentrations, and can heavily contribute to the total cost.

Though dredging is an efficient way of removing P trapped in sediments, it does have certain drawbacks. It is a large operation that is expensive and invasive. It should only be considered if the habitat is in serious decline and reconstruction is the only option (Wagner 2004).

1.4.1.b Aeration

Aeration is a technique that acts to increase oxygen levels of a pond and minimize stratification. Oxygen is connected to eutrophication control because anoxic environments can release P from the soil to stimulate internal cycling. Therefore, with more oxygen and a more homogenous distribution, internal cycling is limited (Wagner 2004). Oxygenation by aeration can also benefit zooplankton populations by lowering the pH and creating more tolerable oxygen and temperature conditions at the depths, allowing them to spread their range (Shapiro et al. 1975). Having a wider range of suitable habitat can reduce predation on zooplankton, so that they might increase their own predation on undesirable algae populations (Shapiro et al. 1975). Because aeration management largely deals with oxygenation, it should be considered when phosphorus release is connected with low DO levels (Wagner 2004). The initial cost of installing an aeration pump is can be high, usually between $50 to $800/acre, in addition to annual costs for use and maintenance (Wagner 2004).

1.4.2 Chemical

1.4.2.a Algicide

Much like using herbicides to kill unwanted terrestrial vegetation, algicides can be used to kill unwanted algae in freshwater bodies. Copper sulfate is the most commonly used algicide (Wagner 2004, Olem and Flock 1990). Copper interferes with photosynthetic processes in algae, and thereby retards their growth (Wagner 2004). Copper algicides have been particularly effective against cyanobacteria, but, as a result of

their widespread use, certain strains of *Anabaena*, a genus of cyanobacteria classically associated with eutrophication, and a few species of green algae have developed resistance (Wagner 2004).

Algicides should only be used as a last resort because of their unintentional negative impacts (Wagner 2004). Copper is toxic to fish as well as other microscopic organisms, such as dinoflagelletes and diatoms, and effects of chronic exposure on other organisms higher in the trophic pyramid have not yet been fully realized (Olem and Flock 1990; Wagner 2004). Algaecides also create toxic side effects when they disrupt the cells of noxious cyanobacteria and their toxins are released into the environment (Wagner 2004; Lurling 2013). Algicide treatment does not address the root causes of eutrophication, and can actually perpetuate eutrophic conditions by depleting dissolved oxygen. It is not effective in the long term, and additional applications are sometimes required (Olem and Flock 1990).

1.4.2.b Phosphorus Binding

Once external phosphorus inputs are controlled, there are chemicals that can be applied to bind phosphorus already in the system to suppress internal inputs. Flocculents made from aluminum, iron, or calcium are the most commonly used compounds to remove particulates from the water column (Wagner 2004, Lurling 2013). Aluminum sulfate, or alum, has been the most widely used and successful of these salts (Cooke et al. 1993). When aluminum sulfate enters the water, aluminum hydroxide ($Al(OH)_3$) forms and the pH of the water decreases (Cooke et al. 1993). This aluminum-based floc has the

capacity for high P adsorption, and phosphates are stripped from the water column as the compound settles to the sediments (Cooke et al. 1993). To most effectively reduce internal cycling of phosphorus, it must then be fixed, or bound, so that it cannot escape from the sediment where it accumulates. Luring and Oosterhout found this combination, that they call "Flock and Lock" (flocculent and fixative), to be the most effective way to reduce both chlorophyll-a and phosphorus concentrations of small, shallow lakes (Lurling 2013). The fixative that they suggest is lanthuim-modified bentonite Phoslock®, a product developed by Australia's Commonwealth Scientific and Industrial Research Organization (CSIRO) (Douglas 2002, as cited by Meis et al 2012). Fixation occurs when phosphorus, in the form of orthophosphate, is permanently bound to lanthanum (SePRO 2014).

The "flock and lock" method is most effective when external phosphorus loads are no longer overwhelming the ecosystem and when internal cycling becomes the main source of phosphorus to algal populations. Results from a "flock and lock" approach can be observed rapidly, but are not necessarily lasting (Lurling 2013). Treatment is generally effective for about five years (Olem and Flock 1990). Success is also dependent on the pH, alkalinity, and DO levels of the water (Lurling 2013). Phosphorus inactivation with aluminum sulfate, a popular "flocking" compound, works best in hard waters with circumneutral pH (Olem and Flock, 1990). In soft water, it is easier for pH to fall below 6, under which conditions aluminum can change into forms associated with toxic effects (Olem and Flock 1990, Cooke et al. 1993).

1.4.2.c Dyes

Dyes are a chemical treatment without toxic side effects. Dyes act to control phytoplankton by obstructing light access (Wagner 2004). This technique has been successful, but is generally used on small, ornamental ponds and not often in Massachusetts (Wagner 2004).

1.4.2.d Barley Straw

Using barley straw to control eutrophication is a new technique that continues to be researched. Barley straw management works by packing barley, about 250-lbs per acre, into loose mesh bags at the beginning of the summer, when conditions are starting to become favorable for algae blooms (McComas 2003). When barley straw sits in water, decomposition by fungi cause a chemical reaction that inhibits the growth of algae (Lembi 2002). Researchers have reported successful inhibition of nuisance cyanobacteria such as certain species of *Microcystis* and *Anabaena*, as well as green algae like *Scenedesmus* and *Spirogyra* (Islami and Filizadeh 2011). Decomposition requires a high oxygen environment, so it is best to have the bags floating on the surface near shore (McComas 2003). It can take a few weeks for the compounds that inhibit algal growth to build up, but effects are likely to last the remainder of the summer at which point the bags can be removed (McComas 2003). Though the process is not well understood, some research suggests that it might also have the effect of lowering P concentrations (Lembi 2002).

The effects of barley straw are variable, with greater success in bodies with long residence time (Lembi 2002, McComas 2003). It can require multiple applications, but research from England suggests that it can be effective in the long term (McComas 2003, Lembi 2002). The EPA has not yet assessed barley straw, so it cannot be used on public land, but is a potential management option for privately owned ponds (Lembi 2002).

1.4.3 Biological

Biomanipulation alters ecosystem characteristics to stimulate a change that will correct the problems associated with eutrophic systems. It involves complex interactions that are not completely understood and can have mixed results as a consequence.

1.4.3.a Top-Down Control

Food web interactions are a major component of ecosystem dynamics, and are closely associated with the abiotic environment. Problems with eutrophication are often centered around nuisance algal blooms. Top-down biomanipution acts to control algal populations through increased grazing by zooplankton. Large populations of zooplankton in lakes have been associated with clearer water and low algal populations (Wagner 2004). This has been seen even in lakes that maintain high phosphorus levels (Shapiro et al. 1975).

There are several different ways to go about stimulating zooplankton populations. The simplest way is to directly add zooplankton to the effected pond (Shapiro et al. 1975). Other planktivores, that are not zooplankton, could be added, but stocking in this way is less successful because they are likely to increase predation on both phytoplankton and zooplankton, which is counter-productive (Wagner 2004). In research about trophic level interactions and their effects on algal populations, it was shown that systems with an odd number of trophic levels, one or three, had higher algal biomass than systems with even numbers, two or four (Smith and Schindler 2009). To support a four level system, piscovorous fish can be added that will graze on zooplankton predators. Unfortunately, this is more of a suggestion for larger lakes, because ponds are likely already at carrying capacity for higher trophic level species, so the effects would be short lived (Wagner 2004). A two level system might be more effective for a small pond and might be achieved by removing predators of planktivorous zooplankton (Wagner 2004).

The results of biomanipulation are not reliably predictable and will certainly vary between cases. However, it is generally more successful in smaller bodies, where the environment is more easily manipulated (McComas 2003).

1.4.3.b Restoration of Submerged Vegetation

In certain cases of eutrophication, algae populations become dominant and outcompete submerged vegetation for light and photosynthesis resources. Submerged vegetation is very characteristic of ponds and, without it, reaching a clear-water state might not be possible (Hilt et al. 2006). Reestablishing aquatic plant populations that

once grew in a eutrophic pond can help keep algae blooms down by creating competition for light and nutrients (Hilt et al. 2006). After an initial treatment, such as dredging or alum application, populations may come back naturally, but intentional introduction can also be applied. Species chosen should be native and should be tested in a small area before being introduced to the entire pond. Charophytes are generally good for maintaining a healthy ecosystem (Hilt et al. 2006).

1.4.4 No action

Another management option is to take no action. To control inputs of nutrients would still be advised under this solution, but management would end there. Ponds are natural systems that go through changes during their lifespan – which is finite. A pond that appears to be degraded in the public eye, in nature might support a complex and flourishing ecosystem (Wagner 2004).

2. Methods (AB, SM)

2.1 Study Sites

Three field sites were studied and compared in this research: Gilmore Pond in Westborough, MA (Figure 1a), Peacock Pond on Wheaton College campus (Figure 1b), and Wildcat Pond in Milford, MA (Figure 1c). Peacock Pond and Gilmore Pond were chosen because they have both been targeted as potential management projects within their communities. Peacock Pond is valued for its aesthetics, but has also had many complaints of degradation. Gilmore Pond was originally built as a farm pond, but has since become enveloped by a suburban community that has also expressed concerns about the health and appearance of the pond. Wildcat Pond serves as a water source for the surrounding town of Milford and was chosen for comparative purposes, because this likely indicates good water quality. All three ponds are manmade and are comparable in dimension. Wildcat and Gilmore differ from Peacock Pond in that they are surrounded by a natural buffer, while Peacock is surrounded by buildings and a manicured lawn (Figure 2).

2.1.1 Bathymetry

In order to gain a more precise knowledge of the bottom morphology of Gilmore and Peacock Pond, depths were measured at predetermined points. Points were established every 10-20 m along the shore of one side of the pond and connected to corresponding points across the pond using a rope. The GPS coordinates of these start and end points were recorded, there were ten sets for Gilmore Pond and 19 for Peacock

Pond. Markers were placed every five meters along the rope. Depths were measured and recorded at each of these points from a kayak using a marked rod. The intention was to use this data to create a depth map of each pond. Unfortunately, these maps could not be completed to due technical difficulties.

Figure 1a. Gilmore Pond as viewed from collection site 2 (Figure 2a).

Figure 1b. Peacock Pond as viewed from weekly collection site (Figure 2b.).

Figure 1c. Wildcat Pond as viewed from weekly collection site (Figure 2c).

Figure 2. Map of the pipes flowing into Peacock Pond.

2.2 Field Collections

Ideally, samples should be collected year-round in order to observe seasonal changes in the pond, however, due to the constraints of this study, samples were collected from September through November. Each pond was visited once a week for regular collections, with additional trips for supplementary surveys. Weekly sampling took place from September 6 through November 16 for Gilmore Pond, from September 20 through November 18 for Peacock Pond, and September 13 through November 16 for Wildcat Pond. Gilmore Pond data also included samples collected by a member of the Westborough Community Land Trust from June 16 through August 30, following the same procedures.

2.2.1 Weekly Sampling

Samples were collected on a weekly basis at each pond and occasionally from inflow sources. Gilmore Pond had two regular collection sites (Figure 3a), and Peacock and Wildcat had one each (Figures 3b and 3c). Water samples for chemical analysis were collected in 150 mL plastic Nalgene bottles, pre-rinsed with pond water, and stored in a dark freezer immediately upon returning to the lab. Water samples were collected from inflow sources around Gilmore Pond on August 3 and November 23, Peacock Pond on November 18, and in Wildcat Pond on October 24 and November 16. It rained on October 4 during collections at Gilmore Pond, and a sample was taken from a stream that formed running downhill from the construction site and into the pond. A single sample was collected from the North Basin of Peacock Pond on October 10. In weekly samples,

dissolved oxygen levels were measured using a YSI85 electronic meter both at the surface of the water and a few centimeters above the substrate, this was around 0.35m deep at the sampling site for Gilmore Pond, 0.50m for Peacock Pond, and 0.60m for Wildcat. Dissolved oxygen levels were also measured from a kayak at varying depths. This measurement was taken on October 11 at Gilmore Pond and November 18 at Peacock Pond. pH readings were taken on only a few of the sampling dates with a hand-held electronic sampler. Temperature of the air and water were recorded at each sampling site.

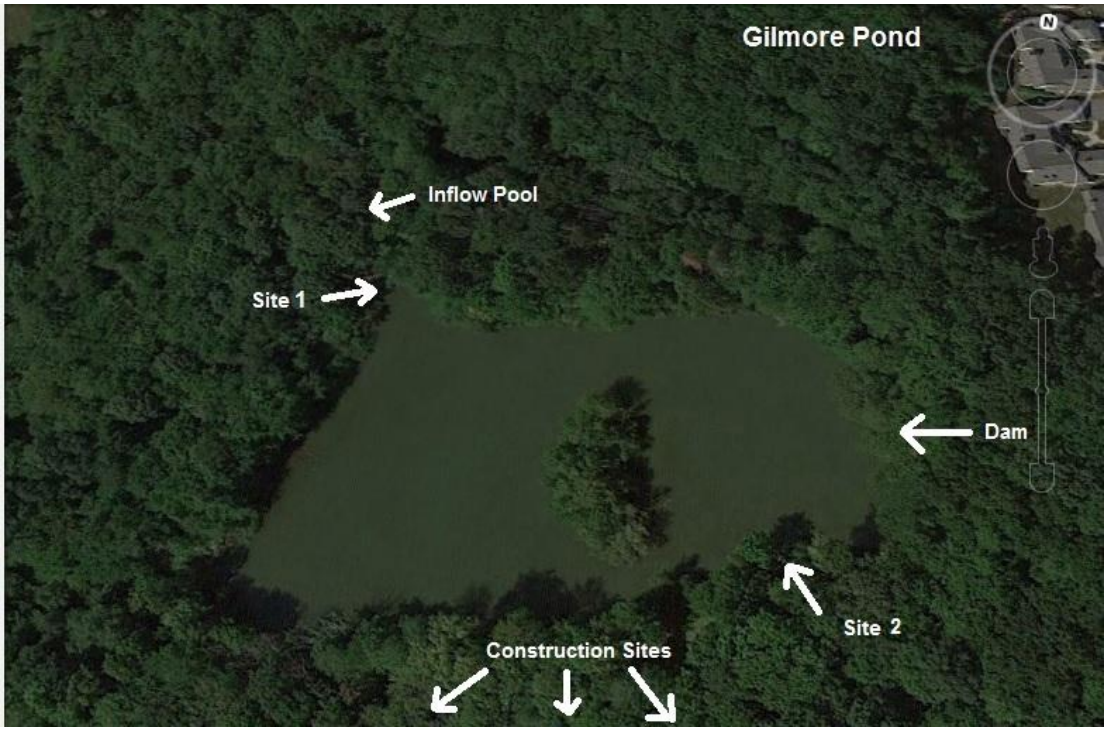


Figure 3a. Aerial view of Gilmore Pond in Westborough, MA. Site 1 and Site 2 were the two weekly collection sites, two samples were collected directly from the inflow pool, and one sample was collected from runoff coming from the construction sites. Photo taken from GoogleEarth.

Figure 3b. Aerial view of Peacock pond on the Wheaton College campus in Norton, MA. Weekly collection site and main inflow pipes are labeled. Photo taken from GoogleEarth.

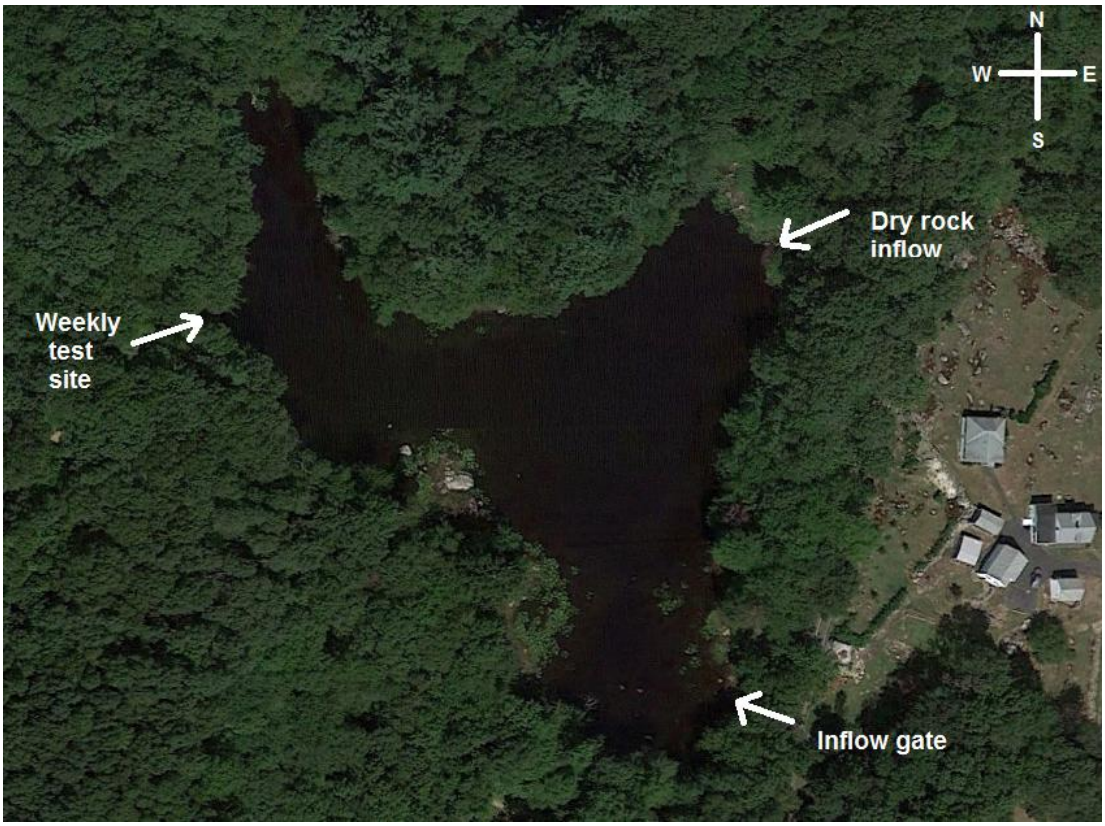


Figure 3c. Aerial view of Wildcat Pond in Milford, MA. Weekly collection site and inflows are labeled. Photo taken from GoogleEarth.

2.2.1.a. Samples for Relative Plankton Analysis

Samples of water with concentrated algal populations were collected to create a species richness count and observe the relative proportions of phytoplankton populations in the ponds. An 80micron plankton tow net was used to collect these samples in 150mL plastic bottles. The bottle was attached to the net and filled half-full with pond water to add weight before being tossed out 3-4m from shore and pulled back promptly with the attached rope to ensure that the water flow was going through the net and that the net

stayed at the surface level (Eaton et al. 2005). This process was initially done 5 times, but later adjusted to 10 to achieve a greater concentration of algae. After the final pull the net was pulsed vertically in the top 10cm of water several times in order to wash plankton off the sides of the net and into the bottle. Immediately upon returning to the lab, Lugol's solution was added to the sample until it resembled the color of tea. The preserved sample was then stored in a dark refrigerator (Eaton et al. 2005).

In addition to these horizontal tows, vertical tows were taken from a kayak in the middle of each pond by sinking the bottle attached to the net in the water and pulling it vertically up through the water column several times. In Gilmore Pond, vertical tows occurred on October 11, October 19, and November 2. A single a vertical tow was conducted in Peacock Pond on November 21, and in Wildcat Pond on October 24. Samples were preserved following the same procedure as horizontal tows.

2.2.1.b Samples for Absolute Plankton Analysis

"Absolute samples" refer to non-concentrated water samples collected to quantify phytoplankton populations in a known volume of water. These samples were collected using a Van Dorn water sampler, which was lowered into the water 5 cm below the surface before releasing the weight. 100mL of water was then decanted into a plastic 150mL bottle. Immediately upon returning to the lab, Lugol's solution was added to the sample until it resembled the color of tea. The preserved sample was then stored in a dark refrigerator (Eaton et al. 2005).

2.3 Sample Analysis

Water samples were analyzed for chlorophyll- α, phosphorus, and ion concentrations. Chlorophyll- α was analyzed as a way to determine algal biomass, which can be used to measure primary productivity, and thus trophic status. Total phosphorus levels were measured to compare with levels characteristic of eutrophic system. Dissolved reactive phosphorus was measured, as well, to determine how much phosphorus in the system was directly available for uptake. Ion chromotography was used to measure ammonium and nitrate, to determine the nitrogen levels available to primary producers. Alkalinity and water hardness were calculated using ion concentrations, in order to understand the chemical parameters of each pond for management purposes. Algal populations trends and composition were determined through algal enumeration of both concentrated and non-concentrated water samples. Population trends can reflect bloom events indicative of eutrophication. Determining species composition allow algae to be used as bioindicators of trophic status.

2.3.1 Water Chemistry Analysis

Frozen samples were thawed in a dark refrigerator to prepare them for water testing. A portion of each sample was filtered, then both filtered and unfiltered samples were refrigerated for further use.

2.3.1.a Chlorophyll-α

100mL of thawed sample was filtered through 25mm 0.45 porosity glass fiber filter paper into a clean plastic bottle using a 60mL threaded syringe. The filter paper with filtrate was then placed into a 15mL plastic centrifuge tube with a cap. 80% acetone was used to rinse off the filter holder into the centrifuge tube, then to fill the centrifuge tube to 10mL. The tube was then sonicated in a sonication bath for 20 seconds before being placed in a dark refrigerator overnight. These steps were repeated for each sample. The samples were then centrifuged at 500g for 20 minutes. Extract was then transferred to a 1cm cuvette and absorbance read using a spectrophotometer at 750, 663, 645, and 630 nm. Manual calculations were then carried out to determine chlorophyll-α concentration in each sample:

$$\text{Chl-}\alpha\ (\mu g/L) = \frac{[11.64\ (Abs663) - 2.16\ (Abs645) + 0.10\ (Abs\ 630)]\ E\ (F)}{V\ (L)}$$

E= volume of acetone used for extraction (10mL)

F= dilution factor (0)

V= volume of filtered sample (100mL)

L= cell path length (1cm)

22 samples were processed according to the above procedures (ESS 1991). The majority of absorbance readings were negative, and thus unusable for calculation. For this reason, chl- α sampling was discontinued for the remaining samples.

2.3.1.b Phosphorous

Phosphorus (PO_4^{3-}) standards were prepared from a certified reference standard and run through a colorimetric analysis using the ascorbic acid method outlined in the

EPA SM 4500-P E, using a spectrophotometer to measure absorbance at 880nm (Eaton et al. 2005). These readings were then used to create a calibration curve from which to calculate sample concentrations of phosphorous. Filtered samples were prepared in the same manner. Due to limited sample volume, only 10mL of filtered sample from each collection date was analyzed, and reagent quantities were altered accordingly. Unfiltered samples were digested using the persulfate digestion method outlined in the EPA SM 4500-P B 5 in order to convert all forms of phosphorus present in the sample into dissolved reactive phosphorus (Eaton et al. 2005). Again, only 10mL of sample was used due to limited volume, adjusting reagent quantities accordingly. Digested samples were then run through the same colorimetric analysis. Results were compared to standards to obtain concentration of orthophosphate (PO_4^{3-}) in each sample. Concentrations in unfiltered samples reflect total phosphorus levels and concentrations in filtered samples reflect dissolved reactive phosphorus levels.

2.3.1.c Ion Chromatography

Ion chromatography was performed to measure Na^+, NH_4^+, K^+, Mg^{2+}, Ca^{2+}, Cl^-, SO_4^{2-}, NO_3^-, and PO_4^{3-} concentrations in the water column. Mixed ion standards were prepared from certified reference standards to create a calibration curve from which to determine sample concentrations. 3mL of filtered sample was placed into 5 ml PolyVials, loaded into an ion chromatograph, and run through a 2 mm column. An autosampler Dionex ICS 2100 was used to analyze the anions and a Dionex ICS 1000 was used for cations (SM 4110).

2.3.1.d Alkalinity

Alkalinity was calculated by subtracting the sum molar concentration of anions from the sum concentration of cations. In freshwaters, this calculated difference is generally the result of bicarbonate (HCO_3^-) concentrations.

2.3.1.e Water Hardness

Water hardness is a measure of the concentration of magnesium and calcium ions in the water (Velinsky 2004). Hardness was calculated using the following formula (Water hardness calculator 2014):

$$[CaCO_3] = 2.5*[Ca^{2+}] + 4.1*[Mg^{2+}]$$

2.3.2 Algae Enumeration

2.3.2.a Concentrated plankton samples

Phytoplankton populations in concentrated water samples were analyzed to determine species richness and observe their relative proportions. Two drops of sample were pipetted from the bottom of a preserved sample, where debris had settled, and placed on a glass slide with a coverslip to be viewed under the microscope at 400x magnification. Starting with a central row of the slide, three rows were viewed and the species were recorded. It was determined in preliminary viewing that after three lengths of the slide there was not a substantial number of new species seen (Mueller-Dombois

and Ellenberg 1974). Upon completing a slide, each identified species was given a relative abundance ranking on the DAFOR scale of dominant, abundant, frequent, occasional, or rare. This ranking was based on the number of individuals seen in all three rows (Lund and Talling 1957). Two different slides made from the same sample were prepared and evaluated in this way by both researchers and then results were compared. In compiling the data, disagreements over varying abundance rankings were decided by averaging, in the case of rankings more than one letter apart, or by flipping a coin.

Species were identified using various algae identification keys and textbooks (Baker et al. 2012; Lund and Lund 1995; Needham and Needham 1957; Palmer 1959; Prescott 1970; Vinyard 1979; Wehr and Sheath 2003). A key to compile of all of the species identified was created using photos available on the internet as well as personal photographs.

2.3.2.b Non-concentrated plankton samples

Phytoplankton counts were conducted in non-concentrated water of a known volume to quantify the population structure of the phytoplankton community under normal conditions. These counts were performed using a Sedgewick-Rafter cell and modeled after SM10200 F 2 (Eaton et al. 2005). First, the sample was mixed to homogenize the 100 ml of pond water. 1 ml of water was pipetted from the bottle into the well of the cell, covered with a coverslip, and left to settle. After 15 minutes, the middle row of the slide was observed under 200x magnification and all species were recorded and counted following the natural unit/clump method, in which colonial algae are

counted as one individual rather than counting each cell as one (Eaton et al. 2005). Diversity of the algal genera found was calculated using the Shannon-Weaver Index (Sager and Hasler 1969).

$H' = \Sigma(P_i)*(-\ln P_i)$, P_i = #of organisms of a genera/total # of all organisms

2.4 Field Surveys

2.4.1 Macrophytes

A survey of macrophytes and benthic filamentous algae was performed on October 19 for Gilmore Pond, October 21 for Peacock Pond, and October 26 for Wildcat Pond to learn about their abundance within each pond. For each pond, the GPS coordinates of 16 evenly spaced points, recorded from GoogleEarth, were chosen as collection sites. A researcher then paddled out to each of these points in a kayak and used a long handled metal rake to sample an area of 1 m^2 along the sediment. Any vegetation the rake gathered was roughly identified and placed into a labeled plastic bag. The samples were then brought back to the lab for further identification (ACT 2011, "Aquatic Plants" 2014, Madsen 1999).

2.4.2 Faunal Surveys

Surveys of animal populations were performed to gain a more complete picture of each pond's ecosystem. To supplement the structured surveys, any forms of aquatic

vertebrate life observed during weekly field collections were also recorded. These data were then used to construct a food web for each pond.

2.4.2.a Macroinvertebrates

Macroinvertebrates were sampled twice during this study using a D-ring net. In Gilmore and Wildcat Pond these surveys occurred on November 8 and November 16, and in Peacock Pond they occurred September 28 and November 8. In knee to waist deep waters, the net was scraped along the bottom sediment to sample a $1m^2$ area (Kazyak 2001). The sample was dumped into a plastic tray for organism identification that took place both on site and in the lab. Identification was done to the lowest taxa possible, which generally did not extend beyond Order (Voshell and Wright 2002). The Shannon-Weaver Index was used to calculate macroinvertebrate biodiversity using the following formula:

$\Sigma(P_i)*(-\ln P_i)$, P_i = #of organisms of a genera/total # of all organisms

2.4.2.b Frog Surveys

On October 4 and October 18, 1-hour visual encounter surveys were conducted to quantify frog populations. A recorder walked around the perimeter of the pond counting frogs based on visual and audible observation. Species were noted, when possible (Manley et al. 2006).

2.4.2.c Minnow Nets

In order to survey the aquatic vertebrate population that cannot be seen from shore, collapsible, polyethylene-netted minnow traps were set in Gilmore Pond on October 4, 18, and 26, in Peacock Pond on October 9, 18, and 26, and in Wildcat Pond on October 4, 18, and 26. Four non-baited traps were fully submerged at four equidistant locations, 2-3m from shore. After 24-48 hours, the minnow traps were retrieved and their contents gently transferred into a large bucket of water (Manley et al. 2006). Each organism present was identified to the lowest taxonomic level possible, individually photographed, and released back into the water. This entire process took between 5-10 minutes, depending on the number of individuals captured. All assessment was done at the site of capture, and the organisms were handled as little as possible. This method excludes the capture of most large fish, and as a result the data gained in minnow net surveys is biased towards smaller aquatic organisms. It is advised to utilize an active survey method, such as seine-netting, to help overcome this deficiency (Ribeiro and Zuanon 2006). However, attempts at these methods were not successful in these ponds because of abundant macrophyte presence in Wildcat Pond and Peacock Pond and difficultly maneuvering the net between the kayak and shore.

2.5 Interviews

In order to obtain background information and target community grievances concerning Peacock and Gilmore Ponds, short interviews, between 30 and 60 minutes, with key community members and pond managers were conducted. Short notes were

taken during the interview and elaborated on immediately afterwards. Aside from a few questions by the interviewers to direct the conversation, exchanges were mostly conversational. Interviewees with knowledge of Peacock Pond included: Dave Nadeau, former chief mechanic at Wheaton College; Gary Pavao, current chief mechanic at Wheaton College; Steve Kelly, head of grounds at Wheaton College; and Darlene Boroviak, professor of political science at Wheaton College. Individuals interviewed about Gilmore Pond included the Westborough town engineer Carl Balduff and members of the Westborough Community Land Trust, specifically Mark Fox and Don Burn.

3. Results (AB, SM)

3.1 Field Data

3.1.1 Bathymetry

Gilmore Pond had a maximum depth of 1.98m and an average depth of 0.93m, with a standard deviation of 0.3m. Peacock Pond had a maximum depth of 2.90m and an average depth of 1.22m, with a standard deviation of 0.55m. Although an organized depth survey could not be conducted for Wildcat, it has a maximum depth that lies between that of Gilmore and Peacock Pond (pers. obs.).

3.1.2 Dissolved Oxygen (DO)

The DO readings indicate that all three of the ponds were well-oxygenated. DO concentrations in Gilmore Pond ranged from 6.5-10 mg/L (surface) and 4.5-8.4 mg/L (depth). In Peacock Pond concentrations ranged from 7.35-11.5 mg/L (surface) and 7.65-9.85 mg/L (depth). Wildcat Pond had DO concentrations ranging from 5.5-9.54 mg/L (surface) and 3.5-7.8 mg/L (depth). DO concentrations were consistently lower in readings taken near the bottom sediments than those taken near the surface of the water. Only one reading in Gilmore Pond (10/6, 4.5 mg/L) and two readings in Wildcat Pond (10/11, 3.5 mg/L; 10/18, 4.11 mg/L) from near the bottom sediments fell below the threshold of 5 mg/L DO, below which warm water fish have difficulty surviving.

The dissolved oxygen concentrations taken at varying depths on October 11th in Gilmore Pond were greatest at the surface (8.1 mg/L in the western end of the pond and 6.5 mg/L in the eastern end) and gradually decreased with depth (6.63 mg/L at a maximum depth of 1.5m in the western end of the pond and 4.7 at a maximum depth of

1m in the eastern end of the pond). In Peacock Pond DO concentrations were measured at various depths on November 18th. Concentrations increased with depth in both the North Basin and the South Basin. In the North Basin surface DO was 9.03 mg/L and 11.25 mg/L at the maximum depth of 1m. In the South Basin the DO concentration at the surface was 10.2 mg/L, and 17.3 mg/L at the maximum depth of 2m.

3.1.3 pH

In all ponds on all dates pH levels were circumneutral. In Gilmore Pond pH ranged from 7.1-8.1, in Peacock Pond 6.8-7.9, and in Wildcat Pond 6.7-7.8.

3.2 Chemistry

3.2.1 Phosphorous

All ponds had an average total phosphorous well below 0.03mg/L, the point at which a body of water is considered eutrophic (Olem and Flocks 1990). Gilmore had the highest average concentration at 0.012mg/L, follwed by Wildcat at 0.006mg/L, and the lowest concentration in Peacock at 0.002mg/L (Figure 4).

3.2.1.a Gilmore Pond

Total phosphorous in Gilmore ranged from 0.002 – 0.055 mg/L at site 1, and from 0.001 – 0.018 mg/L at site 2 (Figure 4). Overall, site 2 had lower concentrations of total phosphorous than site 1. All but 4 weekly samples, all at site 1, were below 0.03mg/L and thus below the concentration indicative of eutrophication. The two highest concentrations of total phosphorous were found in the sample from the inflow pool on Nov 23rd and in

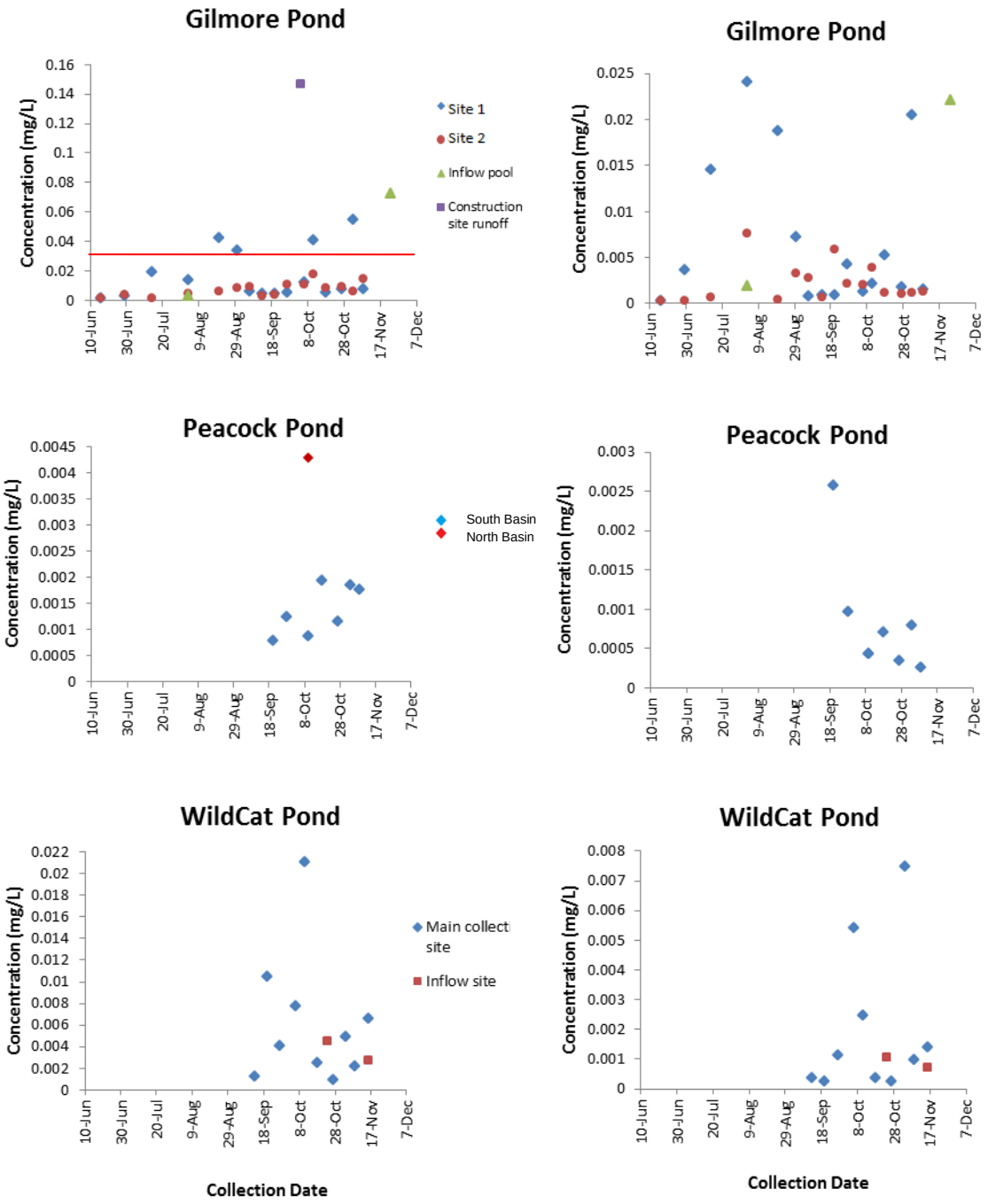


Figure 4. The concentrations of total phosphorous (left) and dissolved reactive phosphorous (right) in each pond through all sample dates. Note the difference in the magnitude on the y-axes. The red line at 0.03 mg/L indicates the threshold above which total phosphorous concentrations are considered eutrophic.

the construction site runoff on Oct 4th. The concentration of total phosphorous in the construction site runoff was nine times higher than the average concentration at site 1 and 19 times higher than the average concentration at site 2 (Figure 4).

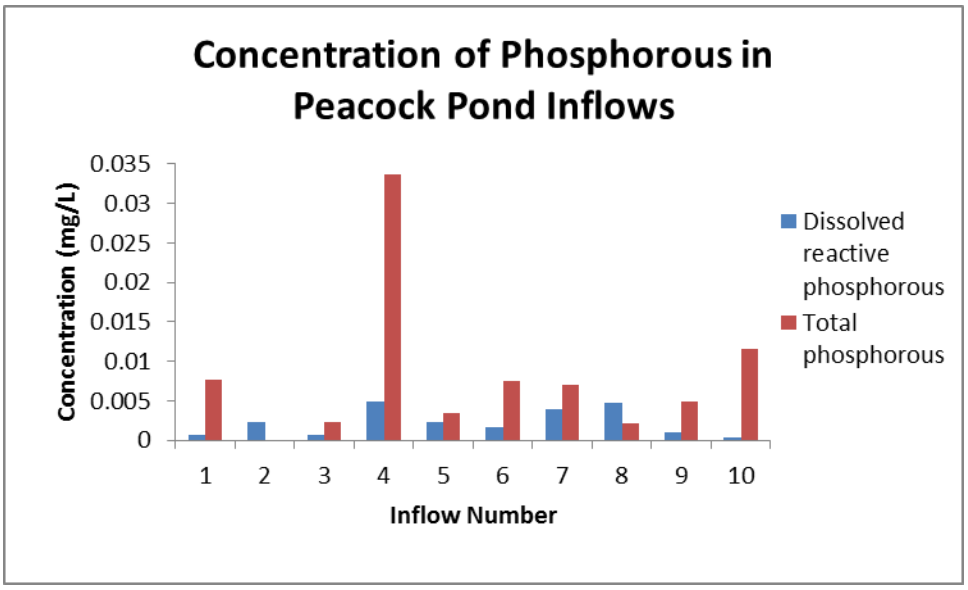
The dissolved reactive phosphorous at site 1 in Gilmore pond showed a drastic increase from Jul 13th - Aug 30th, peaking on Aug 3rd. Dissolved reactive phosphorous also peaked at site 2 on this date, though concentrations were 32% lower than at site 1. Dissolved phosphorous concentrations at site 1 ranged from 0.0004 – 0.024 mg/L, and from 0.0004 – 0.008 mg/L at site 2 (Figure 4).

3.2.1.b Peacock Pond

Weekly total Phosphorous concentrations in Peacock Pond ranged from 0.0008 – 0.0019 mg/L (Figure 4). All samples collected at the regular sample site had total phosphorous levels under 0.002mg/L, well under the level indicative of eutrophication. However, the sample collected at inflow 4 had a total phosphorous level of 0.034mg/L, which just crosses the threshold (Figure 5a). The total phosphorus found in the North Basin of Peacock was 0.0034mg higher than that found at the regular collection site in the South Basin on the same day (Oct 10th). Total phosphrous showed an upward trend as the season progressed ($y=2_{e-5}x - 0.768$, $R^2=0.496$) but drops drastically in January (Figure 4).

Dissolved reactive phosphorous in Peacock Pond showed a downward trend as the season progressed ($y=-3_{E-5}x + 1.351$, $R^2=0.555$), corresponding precisely to a decrease in temperature. Dissolved phosphorous concentrations ranged from 0.0002 – 0.0026 mg/L at the weekly collection site (Figure 4).

(A)



(B)

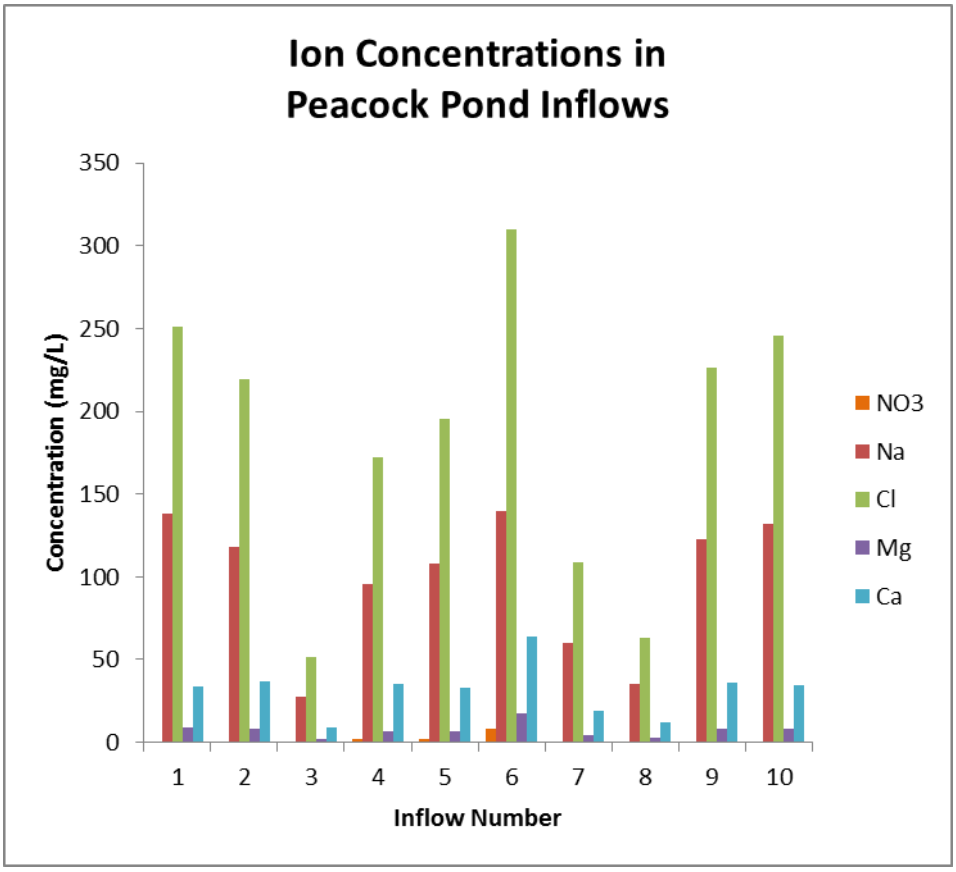


Figure 5. (A) Concentration of phosphorous at ten inflow locations in Peacock Pond (Figure 2), collected on November 18th. **(B)** Concentration of ions in samples from ten inflow pipes in Peacock Pond (Figure 3b), collected on November 18th.

3.2.1.c Wildcat Pond

Weekly total phosphorous concentrations ranged from 0.001 – 0.021 mg/L, and weekly dissolved phosphorous concentrations ranged from 0.0003 – 0.0075 mg/L (Figure 4). Dissolved reactive phosphorous concentrations shoot up on Oct 6th and Nov 3rd, while the peak concentration of total phosphorous occurs on Oct 11th. Both total phosphorous and dissolved reactive phosphorous concentrations at the inflow site fell inside the range of concentrations obtained from the main collection site (Figure 4).

3.2.2 Ion Chromatography

3.2.2.a Ammonium

Peacock Pond had undetectable levels of NH_4^+, while Wildcat Pond only had detectable levels in samples collected from inflows and Gilmore Pond and had low levels (0 - 2.151 mg/L) (Figure 6). NH_4^+ in Wildcat was higher in samples taken from the inflow gate and in front of a seasonal inflow than in samples taken across the pond at the weekly test site. Levels at Gilmore Pond site 2 stayed consistently low, showing only a small increase (0.023 to 0.269 mg/L) on August 20th. At Gilmore Pond site 1, ammonium increases from June 29th-August 30th (0.288 to 2.151 mg/L). The two data points showing concentrations of ammonium found in the inflow pool are greater than all site 2 concentrations, but lower than the peak concentrations at site 1 (Figure 6).

3.2.2.b Nitrate

Weekly samples from Peacock Pond had concentrations below 0.05mg/L throughout the study. However, concentrations from inflows 4, 5, and 6, as well as a

NH_4 Concentrations

NO_3 Concentrations

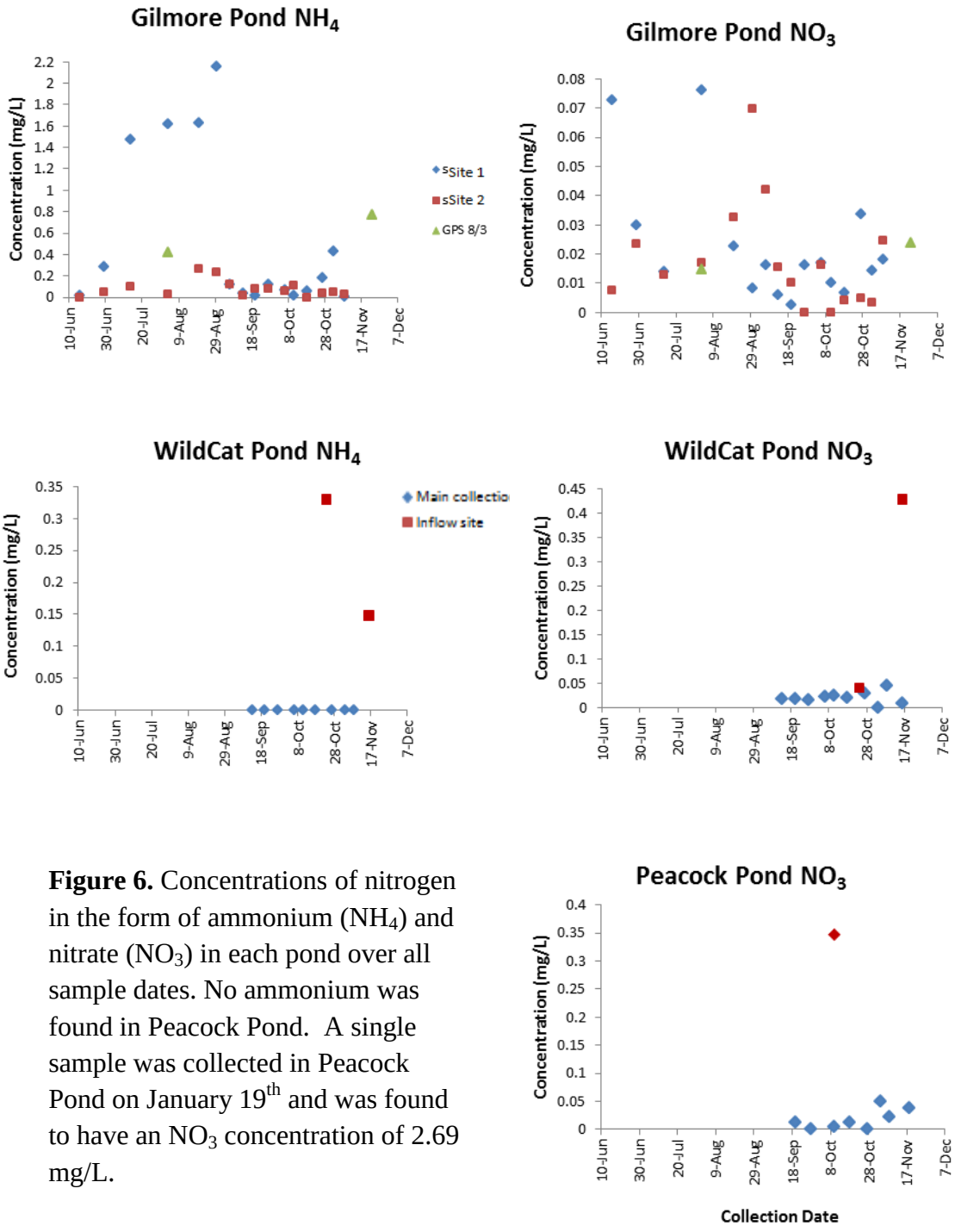


Figure 6. Concentrations of nitrogen in the form of ammonium (NH_4) and nitrate (NO_3) in each pond over all sample dates. No ammonium was found in Peacock Pond. A single sample was collected in Peacock Pond on January 19th and was found to have an NO_3 concentration of 2.69 mg/L.

sample taken from the North Basin of the pond and a sample taken from the weekly collection site in January (2.33, 2.25, 8.08, 0.35, 2.69 mg/L, respectively), were noticeably higher than concentrations in samples from the weekly collection site. Concentrations in Wildcat Pond were all below 0.05mg/L, except for one of the two samples taken from the inflow gate (0.428mg/L). All NO_3^- concentrations in Gilmore Pond fell below 0.08mg/L, with no outliers as seen in the other ponds (Figure 6).

3.2.2.c Sodium and Chloride

In both Peacock Pond and Wildcat Pond there is a direct correlation between the concentrations of sodium and chloride ions (R^2=0.722 and 0.630, respectively). No such correlation was seen in Gilmore Pond (R^2=0.074), which also had the lowest concentrations of both ions (2.8-4.7 mg/L Na^+, 2.8-5.6 mg/L Cl^-). Peacock Pond had the highest concentrations of both Na^+ (131-152 mg/L) and Cl^- (244-266 mg/L), and Wildcat Pond had intermediate levels of both ions (23-27 mg/L Na^+, 41-45 mg/L Cl^-).

3.2.2.d Calcium and Magnesium

Concentrations of calcium and magnesium were considerably higher in Peacock Pond than in Wildcat and Gilmore Pond. In the inflow samples from Peacock Pond, the concentration of these two ions showed a strong positive correlation (R^2=0.943).

3.2.2.e Alkalinity

Peacock Pond, Gilmore Pond, and Wildcat Pond were found to have alkalinities of 0.00 mg/L, 438.27 mg/L, and 21.21 mg/L, respectively.

3.2.2.f Water Hardness

Gilmore Pond had a $CaCO_3$ concentration of 456 mg/L and Peacock Pond had a concentration of 1,020 mg/L, categorizing them as 'very hard waters'. Wildcat Pond had a CaCO3 concentration of 166 mg/L, categorizing it as a 'hard water' system (Velinsky 2004).

3.3 Plankton

3.3.1 Concentrated Algal Samples

3.3.1.a Algal Species Richness

Species richness varied by pond (Table 2). Fifty-nine genera were identified in Gilmore Pond, 68 in Peacock Pond, and 80 in Wildcat Pond. However, Shannon-Weaver indices for algal populations in each pond were not drastically different. Gilmore and Peacock Pond both had a diversity value of 2.3, and Wildcat Pond had only a slightly lesser value at 2.1.

3.3.1.b Algal Abundance

The composition of algae collected in vertical tows did not appear to differ drastically from the composition of weekly horizontal tows (Table 3). The average number of genera seen in vertical tows was slightly greater than in horizontal tows in both Peacock (39.2 and 28, respectively) and Wildcat Pond (39.9 and 30, respectively), but the opposite was true for Gilmore Pond (43 and 38.13, respectively) (Figure 7). Only

one of the vertical tows was taken on the same date as a horizontal tow, allowing for direct comparison (Gilmore Pond, October 11th). Forty-seven genera were identified in the horizontal tow and 45 in the vertical tow on this date (Table 3). Of the genera seen, only four differed in abundance by more than one DAFOR rankings. These genera were: *Microcystis*, *Staurastrum*, *Dictyosphaerium*, and small dinoflagellates. In each case, the genera were recorded as frequent in the horizontal tow and rare in the vertical tow.

Animalia	**Phylum Rotifera**						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Monogonta	**Ploima**	**Brachionidae**	*Keratella sinensis*	+	+	+
				K. sp.	+	+	+
				Kellicottia	+	+	+
				Notholca	-	+	-
			Unidentified	Rotifer species 1	+	+	+
				Rotifer species 2	+	+	-
				Rotifer species 3	+	+	+
				Rotifer species 4	-	+	+
				Rotifer species 5	+	+	-
				Rotifer species 6	+	-	+
				Rotifer species 7	-	-	+
				Rotifer species 8	+	-	+
				Rotifer species 9	-	-	+
				Rotifer species 10	-	-	+
				Rotifer species 11	-	-	+
	Phylum Arthropoda						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Malacostaca	**Amphipoda**	**Unidentified**	scud	+	+	+
	Maxillopoda	**Cyclopoidea**	**Unidentified**	small copepod	+	+	+
				large copepod	-	-	+
	Brachiopoda	**Diplostraca**	**Unidentified**	small cladoceran	+	+	+
				large cladoceran	+	+	+
	Insecta	**Coleoptera**	**Gyrinidae**	Whirligig beetle	+	-	-
		Diptera	**Unidentified larva**	midge, mossquito	+	+	+
Chromista	**Phylum Bacillariophyta**						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Bacillariophyceae	**Achanthales**	**Pinnulariaceae**	*Pinnularia*	+	+	+
		Cymbellales	**Cocconeidaceae**	*Cocconeis*	+	+	+
			Surirellaceae	*Stenopterobia*	+	-	+
			Cymbellaceae	*Cymbella*	-	+	+
		Eunotiales	**Eunotiaceae**	*Eunotia*	+	+	+
				Eunotia filament 1	-	+	+
		Naviculales		*Eunotia filament 2*	-	-	+
				Peronia	+	+	+
			Naviculaceae	*Navicula*	+	+	+
		Surirellales	**Stauroneidaceae**	*Stauroneis*	+	+	+
	Coscinodiscophyceae	**Coscinodiscales**	**Gomphonemataceae**	*Gomphonema*	-	+	+
		Melosirales	**Rhoicospheniaceae**	*Rhoicosphenia*	+	+	+

Chromista	**Fragilariophyceae**	**Fragilariales**	**Coscinodiscaceae**	*Coscinodiscus*	+	-	-
			Fragilariaceae	*Fragilaria*	+	+	+
				Fragilaria filament	-	+	+
				Synedra	+	+	+
				Diatoma	-	+	+
			Melosiraceae	*Melosira*	-	+	+
		Tabellariales	**Tabellariacea**	*Tabellaria*	+	+	+
	Unidentified	**Unidentified**	**Unidentified**	pennate diatom 1	+	+	-
				pennate diatom 2	+	+	+
				pennate diatom 3	+	+	+
				centric diatom 1	-	+	-
				centric diatom 2	-	-	+
				centric diatom 3	-	+	+
	Phylum Chrysophyta						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Chrysophyceae	**Ochromonadales**	**Dinobryaceae**	*Dinobryon*	+	-	+
			Ochromonadaceae	*Uroglenopsis*	-	+	-
			Synuraceae	Synura	-	+	+
		Unidentified	**Unidentified**	Golden Brown Species 1	-	-	+
				Golden Brown Species 2	-	+	-
				Golden Brown Species 3	-	+	-
	Phylum Pyrrphycophyta						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Dinophyceae	**Phytodiniales**	**Phytodiniaceae**	*Cystodinium*	+	-	-
		Gymnodiniales	**Gymnodiniaceae**	*Gymnodinium*	+	+	+
		Gonyaulacales	**Ceratiaceae**	*Ceratium hirundinella*	+	-	-
		Unidentified	**Unidentified**	various medium-sized	+	+	+
				various small-sized	+	+	+
Monera	**Phylum Cyanophycota**						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Cyanophycea	**Chroococcales**	**Chroococcaceae**	*Microcystis*	+	+	+
		Nostocales	**Nostocaceae**	*Anabaena*	+	+	+
			Oscillotoriaceae	*Oscillatoria*	-	+	+
				Lyngbya	-	-	+
			Riculariaceae	*Gleotrichia*	-	-	+
		Unidentified	**Unidentified**	Cyanobacerial species 1	+	+	+
				Cyanobacerial species 2	+	-	-

Plantae	Phylum Charophyta						
	Class	Order	Family	Genus	G	P	WC
	Conjugophyceae	Zygnematales	Desmidiaceae	*Cosmarium*	+	+	+
				Closterium spp.	+	+	+
				C. acerosum	+	+	+
				C. setaceum	-	-	+
				Desmidium	+	-	-
				D. baileyi	-	-	+
				Hyalotheca	-	-	+
				Micrasterias	-	-	+
				Pleurotaenium trabecula	-	+	+
				Staurastrum	+	+	+
				Filamentous desmid 1	-	+	+
				Filamentous desmid 2	-	-	+
			Mesotaeniaceae	*Gonatozygon aculeatum*	-	-	+
			Zygnemataceae	*Mougeotia*	+	+	+
				Spirogyra	+	+	+
	Phylum Chlorophyta						
	Class	Order	Family	Genus	G	P	WC
	Chlorophyceae	Chlorococcales	Chlorococcaceae	*Tetraedron minimum*	+	-	+
			Coccomyxaceae	*Gloeocystis*	+	+	+
			Cylindrocapsaceae	*Cylindrocapsa*	+	+	+
			Dictyophaeriaceae	*Dictyosphaerium*	+	-	-
			Micractiniaceae	*Golenkinia*	+	+	+
		Microsporales	Microsporaceae	*Microspora*	+	+	+
		Oedogoniales	Oedogoniaceae	*Bulbochaete*	-	-	+
				Oedogonium	+	+	+
		Sphaeropleales	Hydrodictyaceae	*Pediastrum*	+	+	+
				Hydrodictyon	-	+	-
				Stauridium tetras	+	+	-
			Scenedesmaceae	*Crucigenia*	+	+	-
				Scenedesmus	+	+	+
				Selenastrum	+	-	-
				Tetradesmus	+	+	-
				Tetrastrum heteracanthum	+	-	+
		Volvocales	Chlamydomonadacea	*Chlamydomonas*	+	-	-
			Volvocaceae	*Eudorina*	-	-	+
				Pleodorina	-	+	-
	Trebouxiophyceae	Oocystales	Oocystaceae	*Ankistrodesmus*	+	+	+
				Kirchneriella	+	+	+
				Treubaria	+	+	+
				Westella	+	-	-

Plantae	**Ulvophyceae**	**Cladophorales**	**Cladophoraceae**	*Cladophora*	-	+	
				Rhizoclonium	-	-	+
		Ulotrichales		*Ulothrix*	-	+	+
	Unidentified	**Unidentified**	**Unidentified**	Unicellular green 1	+	+	+
				Unicellular green 2	+	+	-
				Unicellular green 3	-	-	+
				Unicellular green 4	-	+	+
				Colonial green 1	+	+	+
				Colonial green 2	-	+	+
				Colonial green 3	-	-	+
				Colonial green 4	-	+	+
				Colonial green 5	-	-	+
				Filamentous green 1	+	+	+
				Filamentous green 2	+	-	-
				Filamentous green 3	-	-	+
				Filamentous green 4	+	+	+
				Filamentous green 5	-	+	+
				Filamentous green 6	-	-	+
				Filamentous green 7	-	+	+
	Phylum Xanthophyta						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Xanthophyceae	**Mischococcales**	**Pleurochloridaceae**	*Pseudostaurastrum*	+	-	+
				Tetraedriella	+	+	+
Protozoa	**Phylum Euglenophyceae**						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Euglenales	**Euglenales**	**Euglenaceae**	*Euglena acus*	+	-	-
				E. sp	+	+	+
				Trachelomonas	+	+	+
				various medium-sized	+	+	+
				various small-sized	+	+	+
	Phylum Ciliphora						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Ciliatea	**Petricha**	**Vorticellidae**	*Vorticella species 1*	+	+	+
				Vorticella species 2	-	+	-
		Hymenostomatida	**Frontoniidae**	*Frontonia*	+	-	+
	Phylum Craspedophyta						
	Class	**Order**	**Family**	**Genus**	**G**	**P**	**WC**
	Craspedophyceae	**Craspedomonadales**	**Prymnesiophyceae**	*Rhipidodendron*	+	+	+

Table 2. Species richness data collected during the analysis of concentrated algal samples (ITIS 2014)

Table 3: DAFOR Abundance of Algal Genera

3a. GILMORE POND																		
	6/16	6/29	7/14	8/3	8/20	8/30	9/6	9/20	9/27	10/6	10/11	10/18	10/27	11/2	11/8	10/11	10/19	11/2
DIATOMS																		
Pinnularia						R	R	R		R	R	R				O	R	
Cocconeis		R								R	R							
Stenopterobia	R											R						
Eunotia																R		R
Peronia								R	R	R		R	R			R		
Navicula		R	R		R	R	R	R		R	R	R	R	R	R	O	R	R
Stauroneis						R	R	R			R					R		
Rhoicosphenia						R			R	R		R						
Coscinodiscus			R	R	R										R			
Fragilaria	R	F	O	R	R	O	F	A	F	F	F	F	R	F	F	F	F	F
Synedra	R	O	R	R	R	O	R	R	R	O	O	R	R	O	R	F	O	O
Diatoma										R								
Tabellaria	R	R	R	R		R		R	R		R	R			R	R	R	
pennate diatom 1		R	R						R		R	R		R		R	R	
pennate diatom 2		O	R			R												
GOLDEN-BROWN ALGAE																		
Dinobryon	O	O								R	O	O	O	R	R	R	O	R
DINOFLAGGELATES																		
Cystodinium					R			R				R		R				

Gymnodinium			R		R	O	F	O		R	R	R				R		
Ceratium hirundinella		R																
medium-sized	R	A	R	O	O	F	R	R	R	R	O	R	R	F		R	R	R
small-sized	R	R	R	R	R	F	R	O	R	O	F	R		R		R	R	R
CYANOBACTERIA																		
Microcystis	R	R	R	F	R	O	R	R	F	F	F	R	R	R	O	R	O	R
Anabaena	R	F	D	D	O	O	R	R		R	R	R				R		R
Cyanobacerial species 1	R	O	R	R	O	F	O	R	R	R	R	R	R	O	R	R	O	O
Cyanobacerial species 2									R									
GREEN ALGAE																		
Cosmarium		R	R		R	R	R	R	R							R		
Closterium spp.							R									R	R	R
C. acerosum								R			R				R			R
Desmidium sp.								R				R			R		R	R
Staurastrum	R	R	O	R	R	F	R	R		R	F	R	R	O	O	R	O	O
Mougeotia											R				R	R		R
Spirogyra												R						
Gloeocystis									R			R	R	R	R		R	R
Dictyosphaerium	O	F	F	O	O	F	O	R	F	O	F	R	R	R	R	R	O	O
Oedogonium								R										
Pediastrum	R	R	R	R	R	O	O	O	O	R	O	R	R	R	R	R	R	R
Stauridium tetras			R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Crucigenia	R	O	O			R	R	R		R	R	R	R	O	O	R	R	O
Scenedesmus	O	F	F	R	O	F	R	R	R	O	O	O	R	R	O	R	O	O
Selenastrum		R	R		R	R	R	R	R	R	R	R		R	R	R	R	R
Tetradesmus		R	R			F	R	R	R	R	O	O	R	R	R	R		O

Tetrastrum heteracanthum		R	F			R	R	O	R	R	O	R	R	R	R	R	R	R
Ankistrodesmus	O	F	O	O	F	O	R	O	R	R	R	R	R	R	R	R	R	R
Kirchneriella		R	O	R	R	R	R	R	O	R	R	R		R	R	R	R	R
Treubaria	R	R		R	F												O	R
Westella	R	R	O	R	R	R	R	R	R	R	R	R		R	R	R	R	R
Tetraedron minimum					R	R	R	R	R	R	R	R		R	R	R	R	R
Unicellular green 1						R	R	R			R	R		R	R	R	R	R
Unicellular green 2							R	F	R	R	O	O		R	R	R	O	O
Colonial green 1	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R	R
Filamentous green 1									R									
Filamentous green 2														R				
Filamentous green 4								R										
Pseudostaurastrum		R	R	R	R	R		R		R		R		R		R	R	
Tetraedriella	R	R	O	R			R	R	R	R	R	R		R	R	R	R	R

3b. PEACOCK POND										
	9/20	9/23	9/28	10/7	10/10	10/18	10/27	11/8	11/13	10/21
DIATOMS										
Pinnularia	R	R				R				
Cocconeis	F	O	O	R	R	R	R	R	R	R
Cymbella		R	R		R					
Eunotia	R	R	R							
Eunotia filament 1			R	R	R	R	R		R	
Peronia		R								
Navicula	O	O	O	R	R	R	R	R	R	
Stauroneis			R							
Gomphonema	R	R	R	R	R	R	R	R	R	R
Rhoicosphenia	R	R	R		R					
Fragilaria	O	O	R	R	R	R	R		R	
Fragilaria filament		R	R	R		R	R	R	R	
Synedra	O	O	O	R	O	R	R	R	O	R
Diatoma		R	R	R	R		R		R	R
Melosira	R	R	R	R	R		R		O	R
Tabellaria	R	R	F	R	R		R	R	R	R
pennate diatom 1	R	R	F	R			R			
pennate diatom 2	R				R					
pennate diatom 3		R								
centric diatom 1									R	
centric diatom 3	R	R	R	R	R	R	R	R	R	R
GOLDEN-BROWN ALGAE										
Dinobryon							R			
Uroglenopsis	R		R	R	R		R		R	R
Synura				F	R					
Golden Brown Species 3	R									
DINOFLAGGELATES										
Gymnodinium				R		R	R			
medium-sized	R	R	R	R	R	R	R		R	R
small-sized	R	O	R	R	R	R	R	R		R
CYANOBACTERIA										
Microcystis	R	R	R	R		R	R			
Anabaena			R						R	
Oscillatoria			O	F	O	O	F	R	O	R
Cyanobacerial species 1	R	R	R	R	R	R	R	R	R	R

GREEN ALGAE										
Cosmarium	R	R	R	R	R		R		R	R
Closterium spp.			R	R	R	R	R		R	
C. acerosum				R	R	R	R			R
Pleurotaenium trabecula					R				R	
Staurastrum	R	R	R							R
Filamentous desmid 1		R		R	R				R	
Mougeotia	R	R	O	R			R		R	R
Spirogyra		R	R	R	R	R	R		R	
Gloeocystis		R	R	R	R	R	R			R
Cylindrocapsa		R	R	R	R					
Microspora			R							
Oedogonium	R	R	R	R	R				R	
Pediastrum	R	R	R		R					R
Hydrodictyon									R	
Stauridium tetras	R	R	R							
Scenedesmus	R	R	R	R	R	R		R	R	R
Tetradesmus	R	R								
Pleodorina		R	O	R	R	R	R			R
Ankistrodesmus	R	R								
Treubaria	R									
Cladophora			R							
Unicellular green 1		R		R			R		R	
Unicellular green 2		R	R	R	R			R	R	
Unicellular green 4	R	R								
Colonial green 1	R	R	R	R			R			R
Colonial green 2	R		R	R			R	R	R	R
Colonial green 4	R	R	R		R			R	R	
Filamentous green 1			R					R	R	R
Filamentous green 3	R		R							
Filamentous green 4			R		R	R	R		R	
Filamentous green 5								R		
Filamentous green 7		R	R			R		R		R

3c. WILDCAT POND										
	9/13	9/20	9/27	10/6	10/11	10/18	10/27	11/3	11/8	10/24
DIATOMS										
Pinnularia	R	R	R	R	R	R	R	R	R	R
Cocconeis							R			
Stenopterobia							R			R
Cymbella	R		R				R		R	
Eunotia	R	O	R	R	R	R	O	R	O	R
Eunotia filament 1	F	R	R	O	R	R	R	R	R	
Eunotia filament 2	F	R								
Peronia		R					R		R	
Navicula	R	R	R	R	R	R	O	R	R	R
Stauroneis		R			R				R	
Gomphonema	R						R			
Rhoicosphenia					R		R		R	R
Fragilaria				R	R					
Fragilaria filament							R		R	
Synedra	F	R	R	R	O	R	R	R	R	R
Diatoma	O	R	R	R	R	R	O	R	O	R
Melosira		R	R		R	R	R			R
Tabellaria	A	R	R	R	R	R	R	R	O	R
pennate diatom 1		R								
pennate diatom 2					R					
pennate diatom 3		R			R		R	R		R
centric diatom 1							R	R	R	R
GOLDEN-BROWN ALGAE										
Dinobryon	R	F	**A**	R	**D**	O	**A**	**D**	**A**	F
Synura				R						
DINOFLAGGELATES										
Gymnodinium	R									
Ceratium hirundinella							R		R	R
medium-sized	R	R	R	R	R	R				
small-sized	R	R	R							
CYANOBACTERIA										
Microcystis					R	O		R		
Anabaena		R								
Oscillatoria	R			R						

Cyanobacerial species 1	R									
GREEN ALGAE										
Cosmarium	R			R	R	R				
Closterium spp.	R				R					
C. acerosum		R	R		R	R	R			R
C. setaceum	O	O	R	R	R		R			
D.baileyi					R	R	R			
Hyalotheca	R									
Micrasterias	R	R	R	R	R	R				
Pleurotaenium trabecula		R			R					R
Staurastrum	R			R	R				R	
Filamentous desmid 1	R	R	R	R	R					
Filamentous desmid 2	R		R	O	F	R				
Gonatozygon aculeatum		R	R	R		R				
Mougeotia	R	R		R	R	R	R	R	R	R
Spirogyra	R		R						R	R
Gloeocystis	R	R				R				
Cylindrocapsa				R			R			
Microspora	O	R	R	R	R	R	R			
Bulbochaete	R		R		R	R				
Oedogonium	R			R	R	R	R	R		R
Pediastrum	R	R	R		R					
Scenedesmus	R		R							
Eudorina	R	R	R		R					
Rhizoclonium		R			O	R	R	R	R	R
Ulothrix										R
Tetraedron minimum	R									
Unicellular green 1	R								R	
Unicellular green 3	R	R	R							
Colonial green 1	R									
Colonial green 2		R								
Colonial green 3		R								
Colonial green 4	R		R	R		R				
Colonial green 5				R	R	R				
Filamentous green 1	R		R	R			R			
Filamentous green 3							R			
Filamentous green 4			R	R	R	R	R		R	
Filamentous green 5		R								
Filamentous green 6		R								

Filamentous green 7		R								
Pseudostaurastrum							R			
Tetraedriella	R	R	R		R					

Table 3. The results of DAFOR assignments for phytoplankton identified in weekly concentrated water samples. Shaded cells represent data from vertical plankton tows. **(3a)** Species list for Gilmore Pond.**(3b)** Species list for Peacock Pond. **(3c)** Species list for Wildcat Pond.

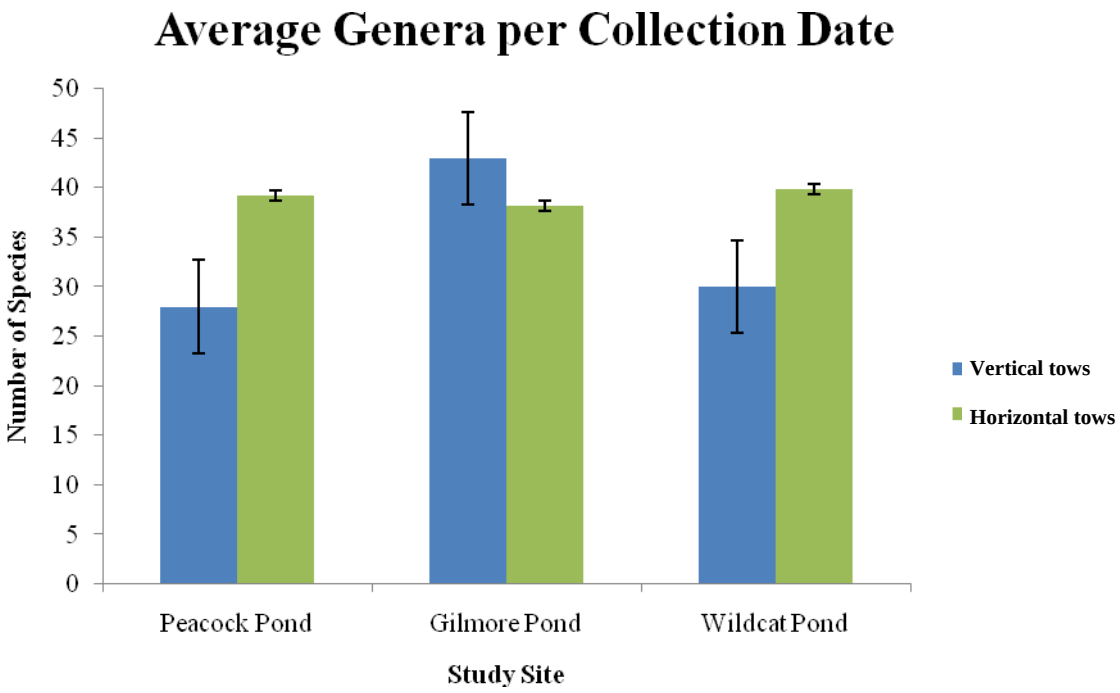


Figure 7. Comparison of the average number of genera observed per collection date in concentrated algal samples in horizontal tows from land (weekly) and in vertical tows from a kayak (kayak) with standard error bars. Weekly samples: Peacock n=9, Gilmore n=15, Wildcat n=9. Kayak samples: Peacock n=1, Gilmore n=3, Wildcat n=1.

3.3.2 Non-Concentrated Algal Samples

3.3.2.a Algal Composition

The ponds in this study were similar in the number of algal genera within each taxonomic grouping (Figure 8). In all three ponds, the most diverse group of algae was the green algae, followed by the diatoms. However, Golden-brown algae were lacking from Peacock Pond, and Gilmore pond had a greater diversity of cyanobacteria species than the other two ponds (9% compared to 5% and 3%). Conversely, the ponds differed greatly in the number of individuals counted within each taxonomic grouping. Peacock Pond was dominated by diatoms (62%) while Wildcat Pond was dominated by golden-brown algae (55%). Gilmore Pond showed an almost even distribution between diatoms and green algae (42% and 37%, respectively). Gilmore Pond also had a substantially higher amount of cyannobacteria than the other two ponds combined (17%) (Figure 8). However, the difference is not as drastic when the data collected from Gilmore Pond in the summer is separated from the data collected in the fall (Figure 9). In both the fall and the summer there is still an almost even proportion of green algae and diatom individuals, but the amount of cyannobacteria is only 7% of the total number of individuals counted. In the summer, Gilmore is dominated by cyannobacteria, making up 49% of the individuals counted. An average of 24,705; 1,682; and 660 total individuals per mL were counted in Gilmore Pond, Peacock Pond, and Wildcat Pond, respectively. In concentrated samples, a total of 59 algal genera were identified in Gilmore Pond, 68 in Peacock Pond, and 80 in Wildcat Pond.

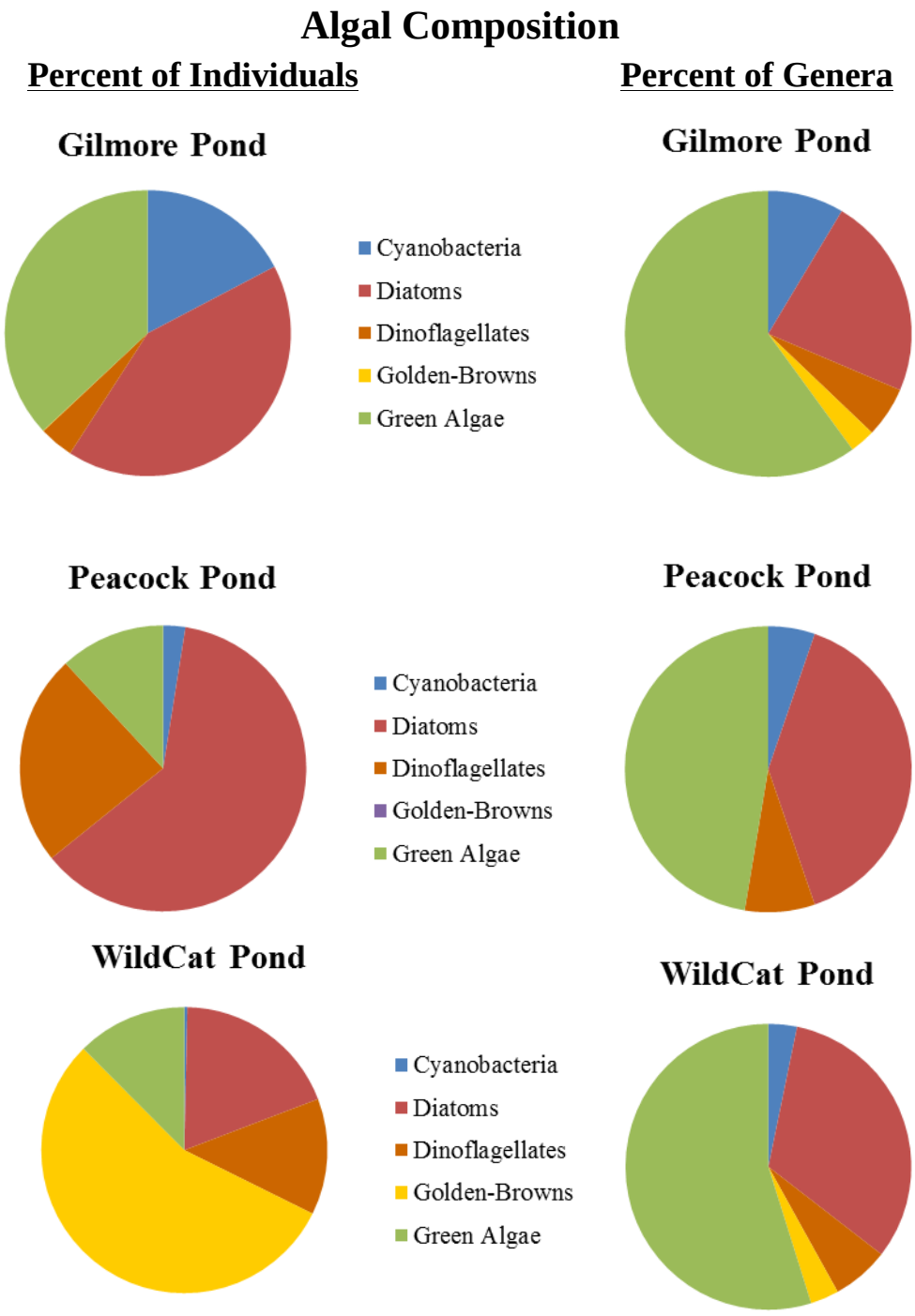


Figure 8. The percent of individuals (right) and genera (left) observed in non-concentrated plankton counts within each higher taxonomic grouping.

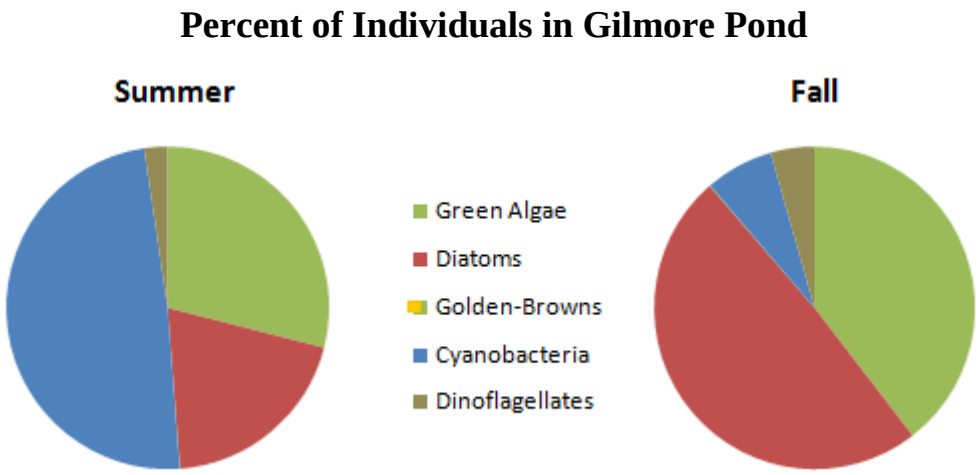


Figure 9. The percent of individual plankton counted in non-concentrated algal samples in Gilmore Pond by season. Summer samples represent collection dates between June 16th and September 6th, and fall sample dates include September 13th through November 8th.

3.3.2.b Change in Density Over Time

The majority of the species encountered in all three ponds did not vary greatly in their abundance over the course of the study, with a few notable exceptions (Figure 10). Numbers of *Fragilaria* per mL showed a dramatic increase in both Gilmore and Peacock Pond in late September (28 – 5,413 and 1-249 individuals per mL, respectively). On the same date that *Fragilaria* numbers peaked in Peacock Pond, so did the number of *Synedra* and dinoflagellates (September 28th). Gilmore Pond experienced a large increase in the number of *Anabaena* per mL in mid-July (0 – 4,033 individuals per mL). *Fragilaria*, *Synedra*, and *Anabaena* are all planktonic species that are known to be indicators of eutrophication. In Wildcat Pond, the number of *Dinobryon* seen per mL increased drastically at the beginning of November (2-83 individuals per mL) (Figure 10). The average density of all algae in Gilmore Pond was 24,705 individuals per mL, in Peacock 1,682, and in Wildcat 660.

Algal Abundance

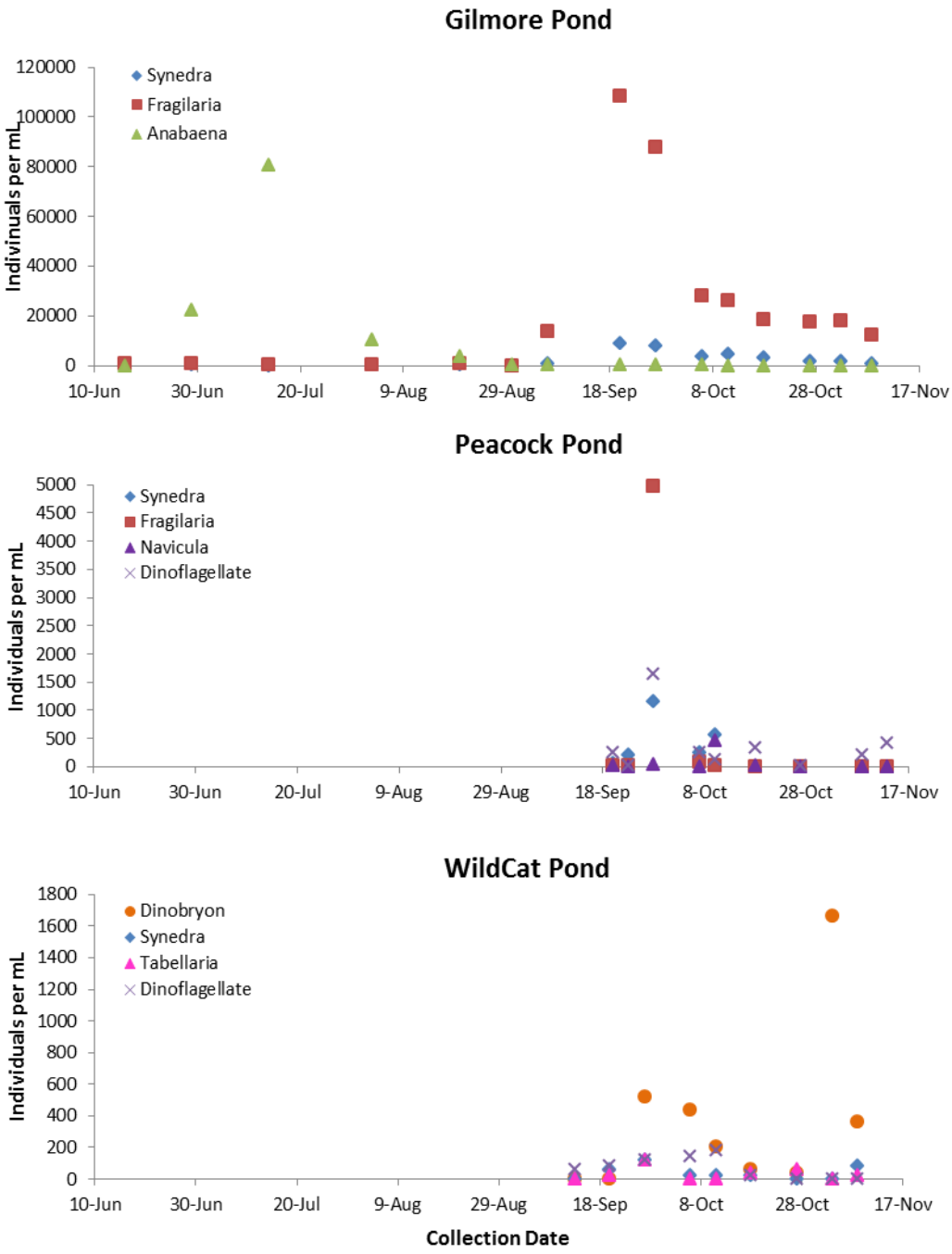


Figure 10. Abundance of the most common algal species in each pond. Note the large difference in values on the Y-axis for each pond. *Fragilaria* has a large increase in numbers in both Gilmore and Peacock Ponds. *Anabaena* numbers jump in Gilmore Pond in early August, and *Dinobryon* numbers jump in Wildcat Pond in early November.

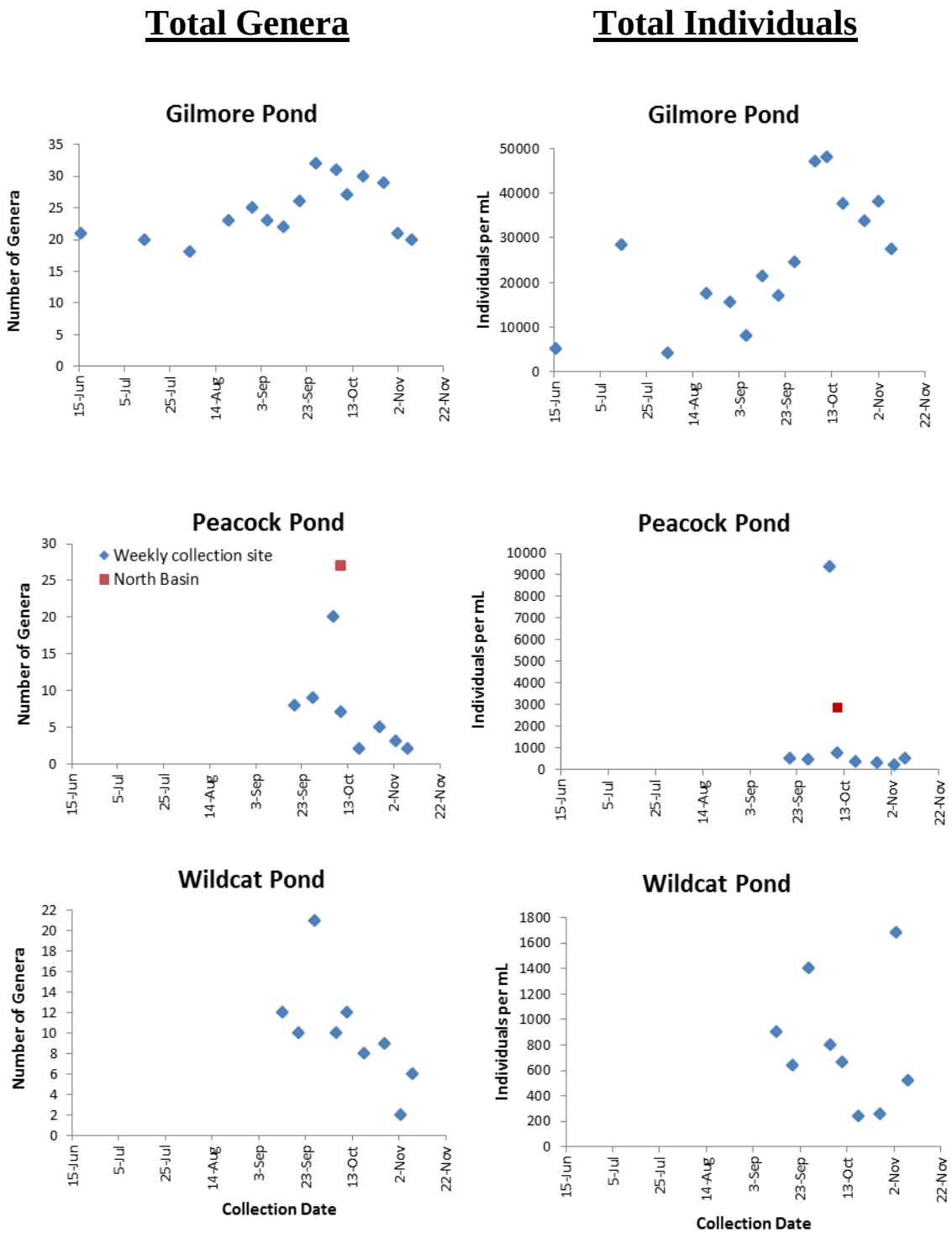


Figure 11. The total number of genera and the total number of individuals seen per sample in non-concentrated algal samples.

Gilmore Pond exhibited a slight upward trend in total number of individuals per mL seen over time (Figure 11). The total number of individuals per mL and total number of genera seen per non-concentrated sample in Peacock Pond did not fluctuate, except for a single outlier on October 7th. The lowest number of genera seen in Wildcat Pond and the highest number of individuals per mL occurred simultaneously on November 3rd (Figure 11).

3.3.3 Top Genera—Indicators of Eutrophication

In Gilmore Pond, all of the genera that were seen in 100% of concentrated samples, all of the genera that had an overall relative abundance greater than rare, and the top 10 most abundant genera per mL of sample were eutrophic indicators. The top genera were *Synedra, Fragilaria*, *Scenedesmus, Ankistrodesmus, Anabaena, Microcystis*, and Dinoflagellates (Wehr and Sheath 2003). In Peacock Pond all genera in these three categories except for the diatoms *Navicula* and *Cocconeis* were also indicators of eutrophication (Wehr and Sheath 2003). Wildcat Pond had only two genera in these categories that characterize eutrophic systems (*Synedra* and dinoflagellates). In Peacock Pond, the top genera were *Synedra, Fragilaria, Navicula, Cocconeis,* an unidentified cyanobacteria (species 1), and dinoflagellates. In Wildcat Pond the top genera were, *Synedra*, *Tabellaria*, and *Dinobryon* (Table 2).

3.4 Field Surveys

3.4.1 Macrophytes

The bottom of Peacock Pond was found to have an extremely high density of

benthic vegetation comprised mainly of filamentous algae and *Elodea canadensis* (pondweed) (Table 4). The category of filamentous algae was comprised of seven genera, including; *Moueogtia, Oedogonium, Spirogyra, Zygnema, Ulothrix, Cladophora,* and *Oscillatoria*. Of the seven genera, *Mougeotia* and *Oedogonium* dominated. The bottom of Wildcat Pond was patchily covered in *Utricularia vulgaris* (bladderwort) with a single patch of filamentous algae. A single aquatic moss was found at one sample site in Gilmore Pond (Table 4). Frequency of occurrence for these surveys was calculated by dividing the number of sample plots in which a species occurred by the total number of sample plots (16).

Benthic Vegetation		
	Species	**Frequency of Occurrence (%)**
Peacock	Filamentous Algae	100
	Elodea canadensis	93.75
	Typha sp.	6.25
	Brasenia schreberi	6.25
	Nymphaea odorata	6.25
WildCat	*Utricula vulgaris*	87.5
	Filamentous algae	6.25
Gilmore	Aquatic moss	6.25

Table 4. The benthic vegetation in Peacock pond was multi-layered, with mats of filamentous algae covering dense growths of *Elodea*. The bottom of WildCat was covered with thick patches of *Utricula vulgaris* and small patches of filamentous algae. In Gilmore, no bottom vegetation was found except for a single aquatic moss.

3.4.2 Vertebrates and Invertebrates

Macroinvertebrates were found in all ponds during D-ring sampling (Table 5). However, based on the results of Shannon-Weaver calculations, Gilmore Pond had considerably lower diversity. Gilmore Pond contained only three types of invertebrates, yielding a diversity value of 0.6. Wildcat Pond had the greatest diversity, with 15 different types of invertebrates and a Shannon-Weaver value of 2.3. Peacock Pond had 10 different invertebrates and a diversity value of 2.0. Red oligochetes were the most common organisms in both Peacock and Gilmore Ponds, though more dominant in Gilmore. In Wildcat, prong gilled mayfly nymphs were the most common organism (Table 5). Three bluegills (*Lepomismacrochirus*) were also captured during D-ring sampling in Wildcat Pond, but these individuals were not included in Shannon-Weaver calculation.

D-ring Sampling			
Organism	**Gilmore**	**Peacock**	**Wildcat**
Scud (Order: Amphipoda)	-	1	11
Aquatic sow bug (Order: Isopoda, Family: Asellidae)	-	15	4
Red Oligochete (Class: Oligochaeta)	38	17	1
Cladocera (Order: Cladocera)	-	-	1
Flatworm (Class: Turbellaria)	-	-	2
Hydrobiid snail (Family: Hydrobiidae)	8	-	-
Planorbid snail (Family: Planorbidae)	-	16	-
Physid snail (Family: Physidae)	-	7	-
Fingernail clam (Family: Sphaeriidae)	-	2	-
Skimmerdragonfly nymph (Order: Odonata, Family: Libellulidae)	-	3	4
unID Dragonfly nymph (Order: Odonata)	-	2	1
Broadwinged damselfly nymph (Order: Odonata, Family: Calopterygidae)	-	1	-
unID Damselfly nymph (Order: Odonata)	-	14	2
Small square gilled mayfly nymph (Order: Ephemeroptera, Family: Caenidae)	-	-	9
Prong gilled mayfly nymph (Order: Ephemeroptera, Family: Leptophlebiidae)	-	-	22
unID Mayfly nymph 1 (Order: Ephemeroptera)	-	-	15
unID Mayfly nymph 2 (Order: Ephemeroptera)	-	-	2
Stonefly nymph (Order: Plecoptera)	-	-	9
Alderfly nymph (Order: Megaloptera, Family: Sialidae)	-	-	1
unID nymph	2	-	-
Watermite (Class: Arachnida, Order: Trombidiformes)	-	-	4
Bluegill (*Lepomis macrochirus*)	-	-	3

Table 5: The results of D-ring sampling in each pond. Gilmore Pond had a total of 3 species, Peacock Pond had 10 species, and Wildcat Pond had 16 species.

Results of frog surveys were very different in all three ponds. Wildcat Pond had the greatest number of individuals counted, with 323 frogs and more than 110 tadpoles.

Gilmore Pond had a total of 62 frogs and 2 tadpoles. Peacock Pond had no frogs or tadpoles (Table 6).

Frog Surveys			
Frog	**Gilmore**	**Peacock**	**Wildcat**
Bull (*Rana catesbeiana*)	7	-	1
Green (*Rana clamitans melanota*)	30	-	322
Pickeral (*Rana palustris*)	2	-	-
Wood (*Rana sylvatica*)	1	-	-
unID	22	-	-
Tadpole (*Rana sp.*)	2	-	>110

Table 6: The results of frog surveys in each pond.

Gilmore Pond had the most minnow net captures. Organisms captured included brown bullheads (*Ameiurus nebulosus*), shiners, tadpoles (*Rana sp.*), and crayfish (Cambaridae). Minnow nets in Wildcat Pond contained shiners and tadpoles (*Rana sp.*), as well. In Peacock Pond bluegills(*Lepomis macrochirus*)and beetles (Gyrinidae)were the only organisms found in the nets (Table 7).

Minnow Nets			
Organism	**Gilmore**	**Peacock**	**Wildcat**
Brown Bullhead (*Ameiurus nebulosus*)	31	-	-
Bluegill (*Lepomis macrochirus*)	-	17	-
Shiner	3	-	4
Tadpole (*Rana sp.)*	12	-	7
Crayfish (Order: Decapoda, Family: Cambaridae)	5	-	-
Beetle (Order: Coleoptera, Family: Gyrinidae)	-	2	-

Table 7: The results of minnow net captures.

Several different organisms were observed during fieldwork. Kingfishers were observed at Gilmore Pond and a great blue heron was observed at Peacock Pond, both piscivorous birds of prey, and Peacock Pond was the only pond with waterfowl. Gilmore

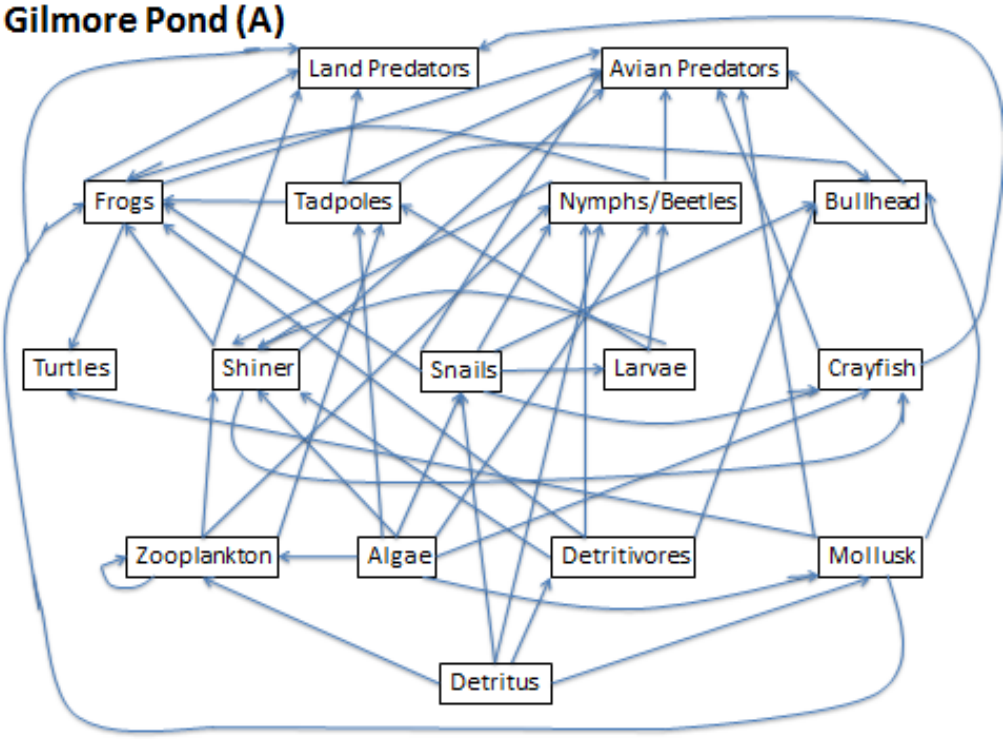
Pond was the only pond where snakes (*Thamnophis sirtalis sirtalis*) and toads (*Bufo sp.)* were observed. Largemouth bass (*Micropterus salmoides*)were recorded at Peacock Pond, though never captured in minnow nets. Insects such as dragonflies (Odonata), backswimmers (Notonectidae), and water striders (Gerridae) were reported in Wildcat Pond. The only organisms observed across all three ponds were painted turtles and songbirds (Table 8).

The results of all of these captures and observations are presented in a food web for each pond that shows potential interactions between organisms (Figures 12a-c).

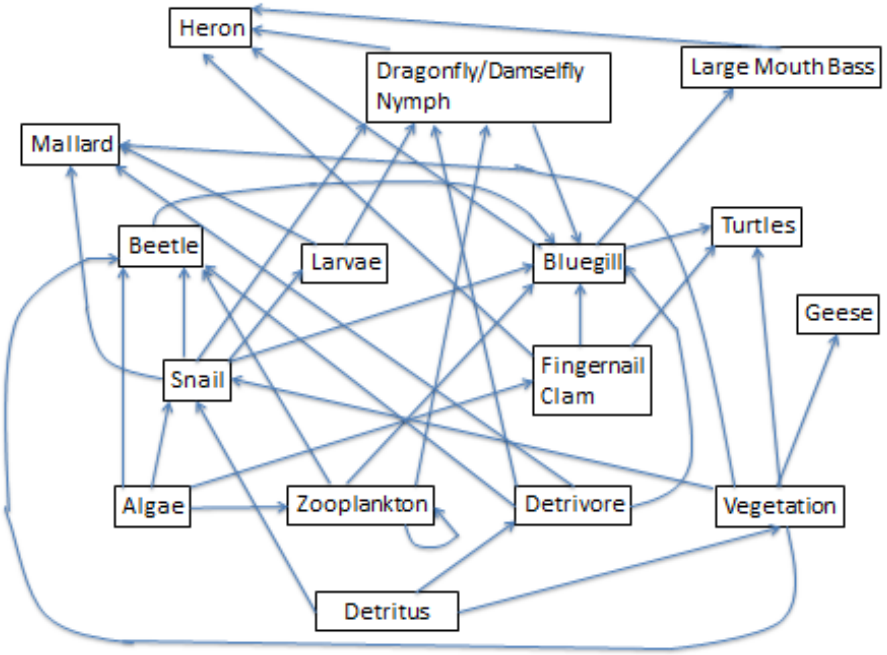
General Observations			
Organism	**Gilmore Pond**	**Peacock Pond**	**Wildcat Pond**
Great Blue Heron (*Ardea herodias*)	+	+	-
Belted Kingfisher (*Megaceryle alcyon*)	+	-	-
Canada Goose (*Branta canadensis*)	-	+	-
Mallard (*Anas platyrhynchos*)	-	+	-
Songbirds (Order: Passeriformes)	+	+	+
Raccoon (*Procyon lotor*)	+	-	+
Muskrat (*Ondatra zibethicus)*	-	-	+
Garter snake (*Thamnophis sirtalis sirtalis*)	+	-	-
Painted turtle (*Chrysemys picta picta*)	+	+	+
Toad (*Bufo sp.*)	+	-	-
Largemouth bass (*Micropterus salmoides*)	-	+	-
Mollusk (Order: Unionoida, Family: Unionidae)			
Dragonflies (Order: Odonata)	-	-	+
Water strider (Order: Hemiptera, Family: Gerridae)	-	-	+
Backswimmer (Order: Hemiptera, Family:	-	-	+

Table 8: The results of general wildlife observations made throughout the study

Gilmore Pond (A)



Peacock Pond (B)



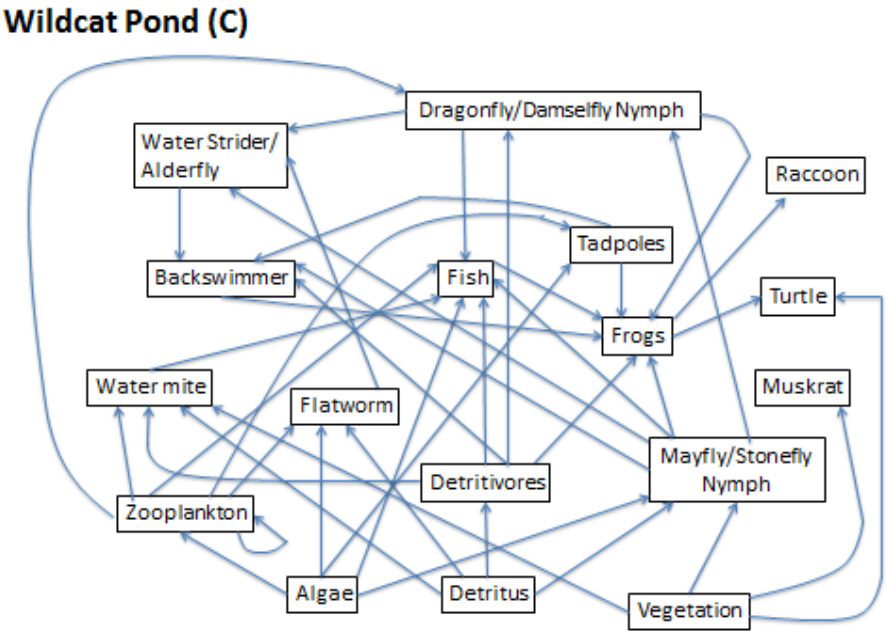


Figure 13: Represents the potential food web interactions in each of the ponds based on organisms observed throughout the study period and the knowledge of a general natural history of the organisms. Data is based on minnow trap surveys, frog surveys, macroinvertebrate sampling, and general observations. Arrows point in the direction of energy transfer. (**13a)** Food web for Gilmore Pond. Vegetation is notably absent. **(13b)** Food web for Peacock Pond. **(13c)** Food web for Wildcat Pond.

3.5 Interviews

3.5.1 Gilmore Pond

3.5.1.a Public Opinion

In the summer and early fall the pond's water becomes discolored and occasionally smells, according to complaints made by local residents. The Westborough Community Land Trust has spent a large amount of time and money on Gilmore Pond remediation, including the construction of a new dam on the eastern end (Figure 4). The Land Trust wants the pond to be an aesthetically pleasing place for recreation, with a

minimal amount of effort and expenses necessary for management. According to members on the trust, some of the residents that live around the pond want more than that, and desire a clear-water system (D. Burns, M. Fox, Westborough Community Land Trust, pers. interview).

3.5.1.b Pond History

Gilmore Pond was excavated in the 1950's to serve as a farm pond for the surrounding apple orchard. The farmer regularly washed off his equipment in the pond, including chemical and fertilizer apparatuses. The hill on which it sits is known to have poor, rocky soil that does not allow for much leaching, and there is little outflow from the pond (C. Balduff, pers. interview). Construction began spring up around the pond in the 1990's, replacing the orchard. This construction has continued on and off through the present day. Of the houses that surround the pond, only one remains on septic, and is in a position downhill from the pond (Figure 4) (C. Balduff, pers. interview).

3.5.2 Peacock Pond

3.5.2.a Public Opinion

Based on interviews with the staff at Wheaton College, it was determined that the main concerns about Peacock Pond revolved around the overabundance of vegetation that grows within the pond due to its detrimental effects on the overall aesthetics of the pond (D. Boroviak, D. Nadeau, S. Kelly, pers. interview). According to Prof. Boroviak, as difficult as it is to manage an artificial water body, the minimum effort required to keep Peacock in its pond state should be. It was also noted that the surrounding community

makes frequent use of the pond for recreational fishing.

3.5.2.b Pond History

In the interview with the chief mechanics the layout of the pond was described, including description of the cyclone separator located in the North Basin, the outflow gate, and the sources of all the pipes that flow into it (Figure 5). In addition to inflow from pipes around campus, Peacock Pond is spring-fed, and this combination gives the pond a very short retention time compared to most small ponds (D. Nadeau, pers. interview). Peacock Pond originated when the stream that used to run in its place was dammed. Over the years the pond began to fill with organic muck, and in 2004 the decision was made to dredge the South Basin. In order to aid in oxygenation and decomposition as a way to prevent the re-buildup of organic material, several aerators were placed throughout the pond (D. Nadeau, pers. interview).

Fertilizer is used as needed on the lawns throughout campus, but this fertilizer does not include any phosphorous, with an N:P:K ratio of 20:0:4. Rock salt (NaCl) and 'snow melt' ($CaMg[C_2H_3O_2^-]$) are used in the winter to keep the pathways ice-free (S. Kelly, pers. interview).

4. Discussion (AB)

After studying the water chemistry and biology of Gilmore Pond, Peacock Pond, and Wildcat Pond, there was strong support that Gilmore and Peacock Pond were eutrophic, and that Wildcat Pond was not. However, this state of eutrophication is not accompanied by a panicked call for action. Gilmore Pond, though eutrophic, still appears to be a functioning ecosystem that supports an abundance of life.

4.1 Chemistry

Gilmore and Peacock Pond were determined to be eutrophic, despite having average total phosphorus levels that fell beneath the accepted eutrophication threshold of 0.03mg/L (Olem and Flocks 1990). In Gilmore Pond, the average concentration was .012 mg/L, and in Peacock Pond the average was a whole magnitude of ten lower at 0.002 mg/L (Figure 4). These levels indicate that even if phosphorus were the limiting nutrient of these ponds, concentrations did not exist in such abundance to stimulate the excessive primary productivity associated with eutrophication. In 2005 and 2012 the Westborough Community Land Trust, the group that owns the conservation land that contains Gilmore Pond, hired Aquatic Control Technology (ACT) to test the pond for water quality. ACT reported total phosphorus concentrations of 0.06mg/L 2005 and 0.15 mg/L in 2012, both of which exceed the average concentration found in this study (ACT 2012, Table 9). ACT's measurements, especially the more recent report in 2012, are also higher than the threshold level for eutrophication.

Comparison with Results of ACT					
Nutrient	**ACT 2005 (average)**	**ACT 2012 (August 21)**	**2013 (August 20)**		**2013 (average)**
			Site 1	**Site 2**	
Total phosphorus (mg/L)	0.06	0.15	0.042	0.006	0.012
Nitrate (mg/L)	<0.05	ND	0.02	0.03	0.020
Total Nitrogen (mg/L)	1.3	0.3	N/A	N/A	N/A

Table 9. This table provides a comparison of Gilmore Pond nutrient concentrations measured in this study with those of ACT measured in 2005 and 2012. ACT data from 2012 was taken from a single water sample on August 21, so the comparative concentrations from 2013 are from the August 20 sample as well as the average of all samples.

However, there were four samples taken from collection site 1 of Gilmore Pond that did contain phosphorus concentrations greater than 0.03 mg/L. The samples were taken between early August and early October, with the lowest being 0.034 mg/L and the highest being 0.055 mg/L – both of which remain lower than even the 2006 value measured by ACT (Figure 4). Site 1 is directly in front of a pool that sometimes connects to the pond to act as an inflow source, so this presents the pool as a possible source of phosphorus loading that should be further explored. To further support this possibility, the sample taken from the actual pool during November also showed high phosphorus levels, of 0.072 mg/L. Throughout the sampling period, site 1, generally, had higher

phosphorus levels than site 2. Elevation maps and the town's septic history do not strongly point to septic leaks as a likely source of this phosphorus, but it is a possibility (Balduff 2014). The other sample from Gilmore Pond that indicated high levels of phosphorus was in a sample taken from a puddle downhill from the construction site on a rainy day. The concentration of this sample was well over the eutrophication threshold at 0.147 mg/L, strongly suggesting construction runoff as a source of phosphorus loading into the pond.

It is curious that measured levels of total phosphorus were drastically lower in Gilmore Pond than when they were measured in both 2012 and 2005 (ACT 2012). One reason for this could be due to the nature of phosphorus cycling in ponds. The results found in this study do not include phosphorus locked in sediments, though sediments are capable of storing a portion of the entire phosphorus inventory of a freshwater system. Though that phosphorus is not directly available to organisms and is not measurable in the water column, it is still an active part of the phosphorus cycle and can be released under conditions of low oxygen (Correll 1998). It would be advised to take these measurements in order to more fully understand the condition of phosphorus in Gilmore Pond.

There was only one sample in Peacock Pond that contained phosphorus levels greater than the eutrophication threshold, at a concentration of 0.0337 mg/L (Figure 4). This sample was from an inflow site, but there are no strong indicators of what might cause this site in particular to have elevated phosphorus concentrations. Fertilizers from ground maintenance would seem a probable source, but was ruled out because fertilizers used by Wheaton College groundskeepers do not contain phosphorus (Kelly 2013).

The phosphorus samples from Wildcat Pond were consistently below the eutrophication threshold, with an average concentration 0.006 mg/L (Figure 4).

Nitrogen is the other major nutrient that can influence phytoplankton populations, although it is often considered to be more important in ocean systems than in freshwater (Moss et al. 2013). This study measured ammonium and nitrate, because they are the primary forms of nitrogen used by freshwater algae (Sze 1998, Figure 6). Total nitrogen concentrations include these forms plus organic nitrogen and, like total phosphorus, this would encompass all nitrogen potentially available to primary producers (Velinsky 2004). Unfortunately, total nitrogen was not measured due to time constraints. This information would have been useful in determining, with certainty, the limiting nutrients of each pond. In Gilmore Pond, there were four samples where ammonia levels stood out from the general trend that was around 0.2 mg/L, and these were all at site 1 ranging from 1.47 to 2.15 mg/L. Once again, this identifies the pool near site 3 as a potential source of nutrient pollution. The pool, itself, had higher ammonium levels than most other samples, at 0.41 and 0.77 mg/L. Nitrate concentrations in Gilmore did not show any obvious trend or outliers.

In 2012, ACT determined ammonia and nitrate to be under the detection limit (ACT 2012, Table 9). In 2005, ammonia was measured the be <0.08 mg/L and nitrate <0.5 mg/L. The level of nitrate they measured is comparable to our samples (Table 9). This study measured ammonium, and not ammonia, preventing a direct comparison to be made with ACT. ACT was also able to determine total nitrogen and in both years found levels to be lower than those indicating eutrophication. The threshold level of total nitrogen for eutrophication is 5 mg/L, compared to their results of 1.3 mg/L in 2005 and

0.3 mg/L in 2012 (Wetzel 2001, ACT 2012). Total nitrogen is the form of nitrogen used to determine trophic state, so based on this information Gilmore Pond does not appear to be in danger of excessive nitrogen loading. The EPA has set the acceptable level of nitrate in drinking water to be 45 mg/L, which is far beyond the highest level of nitrate measured in any pond (Carpenter 1998).

Peacock Pond had undetectable levels of ammonium. This can most likely be attributed to the aeration fountain causing high dissolved oxygen levels, because ammonium is usually low where oxygen is high (Sze 1998). There was one sample in Peacock Pond that reflected a much higher level of nitrate than the others. This sample was taken from the north basin on October 10 and had a concentration of 0.347 mg/L, compared to the sample taken from the south basin on that same day, which had a concentration of 0.005 mg/L – a value much closer to all of the other sample points (Figure 6). This sample is different from the others in location, only, so it is possible that the north basin is receiving inputs of nutrient pollution that the south basin is not directly exposed to.

Wildcat Pond has one sample for both ammonium and nitrate that are clear outliers from the rest of the samples, as they contain concentrations much higher than the general trend. They are not from the same day, but both are from inflow points and could indicate a source of nutrient pollution.

To reiterate a major point of this study, the total phosphorus concentration used as a baseline level for eutrophication was set for lakes, specifically. The small size of ponds, relative to lakes, puts them in a position to "magnify" their process (Teisser et al 2011).

Furthermore, a eutrophication study comparing natural lakes, rivers, and reservoirs found lakes to have the most pronounced increase of algal concentrations following phosphorus additions, with rivers having least pronounced effect (Soballe and Kimmel 1987). This pronounced increase was attributed to the longer residence time of lakes. Shallow lakes generally have shorter residence time than larger lakes, which would logically give ponds the shortest residence time of all of these bodies (Olem and Flock 1990). Therefore, based on these results it seems reasonable to conclude that lower total phosphorus and nitrogen concentrations should be established to indicate eutrophication of ponds. Further investigation would be required to set these levels, and would probably rely most heavily on field-testing.

The high levels of Na^+, Cl^-, Mg^{2+}, and Ca^{2+} in Peacock Pond are reflective of runoff from winter icing (NaCl) and snow-melt compounds (MgCa), which is made clear by the strong correlation between Na^+/Cl^- and Mg^{2+}/Ca^{2+} (Kelly 2013, Figure 5b). After Peacock, Wildcat Pond had the next highest levels of Na^+/Cl^-, which could also be attributed to wintering ice compounds because there is a major highway up elevation from the pond only 100 meters away. Gilmore Pond had the lowest concentrations of Na^+/Cl^-, with the weakest correlation between them, and both Gilmore and Wildcat had minimal Mg^{2+}/Ca^{2+} concentrations. These compounds are not generally the focus of eutrophication studies, but they do have impacts freshwater systems. Salts can change the density of water and prevent full water turnover in the spring, which can cause dissolved oxygen levels to suffer near the sediment (Ramakrishna and Viraraghavan 2005). Sodium might also promote the growth of cyanobacteria, a group of algae that is dominant in eutrophic systems (Briggins and Walsh 1989, as cited by Ramakrishna and Viraraghavan

2005). While sodium can promote cyanobacteria populations, most other populations are negative impacted by salts, including zooplankton and macroinvertebrates (Findlay and Kelly 2011). Ca^{2+} and Mg^{2+} are thought to be less disturbing to ecosystems (Wetzel 2001).

4.2 Phytoplankton

The phytoplankton communities of Gilmore Pond and Peacock Pond provide strong evidence that these ponds are eutrophic. The phytoplankton community of Wildcat Pond is less conclusive, but does not align strongly with eutrophic systems.

In comparing the three ponds, Gilmore and Peacock Pond reflect greater productivity than Wildcat. The Shannon-Weaver values calculated for each pond were very similar, making them unhelpful for comparing diversity between ponds. This is not too disappointing because indices for phytoplankton communities have been found to be of small significance in reports of eutrophication (Wetzel 2001). Nonetheless, the greater number of species found in Wildcat Pond may indicate that it is the least fertile, because of the idea that slower growth rates in less productive ponds allows for more species to develop with overlapping niches (Wetzel 2001). Wildcat Pond was found to have 80 species, with 68 and 59 in Peacock and Gilmore respectively (Table 2).

It is interesting that Gilmore Pond had the least number of species, because it unquestionably had the greatest density of individuals. The shear abundance of algal individuals in Gilmore Pond when compared with the other two ponds strongly suggest eutrophic conditions, as phytoplankton density increases with water fertility (Hutchinson

1967, Wetzel 2001). In Peacock Pond, average algal density was 1,682 individuals/mL, including a sample that captured a *Fragilaria* bloom when the density went up to almost 10,000 individuals/mL, bringing the average up considerably (Figures 8 and 10). Densities were generally greater in Wildcat Pond, although the average was 660 individuals/mL. Samples from Gilmore Pond had concentrations far greater than these, with values ranging from 4,260 individuals/mL to 48,140 individuals/mL and an average of 24,705 individuals/mL. The density of algae in Gilmore Pond reported by ACT is even greater, at 313,000 cells/mL (ACT 2012). However, this is not directly comparable because they enumerated algae by cell and the figures in this study reflect algal units as individuals. Counting each algal cell as one unit could potentially produce greater density values even if the concentration of algae was actually less in ACT samples, however 313,000 cells/mL is still far greater than the individuals/mL reported for Peacock and Wildcat Ponds, supporting the conclusion that the algae concentrations were higher in Gilmore, and therefore reflect greater eutrophication. Because Gilmore Pond did have such a higher density of algae, more algae species were found on a regular basis in non-concentrated samples (Figure 11). Overall, however, Gilmore Pond did not support as many species as the other two ponds.

In analyzing the algal community of a pond, it is important to consider both dominant and more rare species, as both can provide information about the condition of the ecosystem (Hutchinson 1967). Gilmore Pond was largely dominated by cyanobacteria, diatoms, and green algae (Figures 8 and 9). Cyanobacteria and green algae have been categorized as groups that tend to dominate in eutrophic or hypereutrophic systems (Hutchinson 1967, Wetzel 2001). The top genera that dominated Gilmore Pond

were identified as *Anabaena* (cyanobacteria), *Ankistrodesmus* (green algae), *Fragilaria* (diatom), *Microcystis* (cyanbactiera), *Scenedesmus* (green algae), *Synedra* (diatom), and unidentifiable dinoflagellate species. With the exception of the dinoflagellates, all of these genera are eutrophication indicator species (Wehr and Sheath 2003). Furthermore, many of the genera that were commonly seen, but in lesser numbers have also been identified as eutrophication indicators, specifically in shallow lakes or ponds (Wehr and Sheath 2003). These include green algae such as *Pediastrum*, *Selenastrum*, *Kirchneriella*, and *Dictyosphaerium*. Of these top genera and less seen genera, ACT algae count data also reported having seen *Anabaena*, *Microcystis*, *Synedra*, *Scenedesmus*, *Kirchneriella*, and *Dictyosphaerium* (ACT 2012). Out of the 59 genera identified, only three of the more rare genera are recognized as common to oligotrophic water. These genera include, *Desmidium* (desmid), *Gloeocystis* (green algae), and *Tabellaria* (diatom). ACT only reported one oligotrophic genus, *Gloeocystis* (ACT 2012).

In Peacock Pond, there were far fewer individuals per mL than in Gilmore Pond, but the community structure still reflected that of a eutrophic system. Peacock Pond was dominated by diatoms (Figure 8). The dominant genera were mostly eutrophication indicators, including *Fragilaria*, *Synedra*, and an unidentifiable cyanobacteria species (Wehr and Sheath 2003). There were also more rarely found genera that are common to eutrophic waters such as the cyanobacteria *Oscillatoria* and *Microcystis* (Wehr and Sheath 2003).

In temperate regions, the phytoplankton populations of ponds are greatest in the spring and then again in late summer and early fall (Welch 1952). In the first maxima, diatoms and smaller green algae will generally flourish (Wetzel 2001). A past study of

Gilmore Pond that took place over four weeks in March and April in 2011 provided some evidence of this trend. Cyanobacteria were dominant at first, but as the study progressed diatoms took over as the dominant group (Larson-Whittaker et al. 2011). Among their stop species, *Microcystis* stands out as a eutrophication indicator.

This study (Bennett) was able to observe the second phytoplankton maxima of the year in Gilmore Pond, which is often categorized by larger green algae and, in the case of more productive ponds, cyanobacteria (Wetzel 2001, Hutchinson 1967). Bloom events were influential in determining the dominant group of algae, measured in terms of individuals. In Gilmore Pond, the cyanobacteria species, *Anabaena*, bloomed in the summertime. Summer cyanobacteria blooms are typical of eutrophic lakes and ponds, because their ability to fix nitrogen puts them at a competitive advantage to other phytoplankton (Wetzel 2001, Moss et al. 2013). Then, the diatom *Fragilaria* bloomed in the fall. When the summer and fall data are looked at independently, it is clear to see the reflection of these events (Figure 9). Cyanobacteria are far more abundant in the summer, as a result of the *Anabaeana* bloom. Diatoms then become dominant in the fall, again, a reflection of the *Fragilaria* bloom that occurred. In both seasons, green algae have a strong presence, and genera common to shallow, eutrophic waters are well represented. *Fragilaria* also bloomed during the fall in Peacock Pond. This is reflective of eutrophication, as *Fragilaria* is a eutrophication indicator species (Wehr and Sheath 2003).

Wildcat Pond experienced a bloom of *Dinobryon* that made Chrysophaceae the dominant group when considering the number of individuals, but not when considering the distribution of genera (Figure 8). In looking at genera, green algae dominate, as they

do in both Gilmore and Peacock Ponds. Chrysophaceae has been identified as being typically dominant in oligotrophic lakes (Hutchinson 1967, Wetzel 2001). However, the dominant genera of Wildcat Pond, Chlorophyceae, tend to dominate in hypereutrophic systems (Hutchinson 1967, Wetzel 2001). Looking at the genera, *Dinobryon*, specifically does not help to clarify things because certain species are common to eutrophic bodies while other are common to oligotrophic bodies (Wetzel 2001, Wehr and Sheath 2003). The *Dinobryon* that bloomed in Wildcat could not be identified to species.

To more fully understand the phytoplankton communities of these ponds, it would be useful to regularly survey their populations. Phytoplankton populations will very greatly seasonally, but should remain generally consistent from year to year. If seasonal population patterns are not staying consistent, this can be reflective of a disturbance such as nutrient loading (Wetzel 2001).

4.3 Macrophytes

Eutrophic ponds are sometimes dominated by submerged macrophytes (Scheffer and Van Ness 2007). For the purpose of this study, the term "macrophyte" includes mats of filamentous algae as well as true aquatic vascular plants.

The bottom of Peacock Pond was completely dominated by macrophytes (Table 4). There was some diversity, but the plant that dominated throughout the entire pond was *Elodea canadensis*. *Elodea* is a genus that can grow over a range of nutrient levels, but is commonly found in mesoeutrophic to eutrophic waters (Grime 1988, as cited by Thiebaut 2005). It is once again advised to measure phosphorus concentrations in sediments

because this phosphorus is directly available to many macrophytes, including *Elodea*, and because *Elodea* populations have been observed to increase with increasing nutrient levels (Thiebaut 2005). The dense mats of filamentous algae are also characteristic of eutrophic waters, and a few of the species found within these mats are common to eutrophic ponds (Wehr and Sheath 2003).

Gilmore Pond was strikingly devoid of vegetation, except a single patch of moss (Table 4). A previous study from 2007 found abundant plant life in Gilmore Pond (Young 2007). This study found a native coon-tail species (*Ceratophyllum demersum*) growing in six out of eight vegetation sampling locations and Nuttal's mudflower (*Micrathemum micranthemoides*) in one of these eight locations (Young 2007). What likely happened is that a disturbance event altered the nutrient concentrations in the pond and allowed phytoplankton to take over. Dense phytoplankton populations in the water column then out-shaded submerged macrophytes and eliminated their populations (Scheffer and Van Ness 2007).

Ulticularis vulgaris was abundant in Wildcat Pond (Table 4). This is a species that is often found in low-nutrient lakes or ponds, but can grow over a wide range of nutrient concentrations (Kibriya and Jones 2007). This is an interesting plant because it does not have true roots and, therefore, cannot use phosphorus contained in the sediment, like many other macrophytes (Kibriya and Jones 2007). Instead, this species supplements its nutrient intake through carnivory (Kibriya and Jones 2007). The single patch of filamentous algae found in Wildcat Pond was not alarming because of its low distribution. Wildcat Pond appeared to be in a state of mutual competition between phytoplankton and submerged vegetation.

4.4 Alternative Stable States

Limnologists have recently recognized two alternative stable states that characterize eutrophic shallow-lakes and ponds (Scheffer and Van Ness 2007). One state is that of clear water with abundant submerged macrophytes, and the other is characterized by turbid water with minimal or no submerged macrophytes (Scheffer and Van Ness 2007). The ponds in this study were ideal models for observing these ecological states. Gilmore Pond exhibited characteristics of a eutrophic pond with turbid water dominated by phytoplankton. Peacock Pond exhibited characteristics of a clear water eutrophic pond with macrophytes dominating the bottom. Wildcat Pond had observed characteristics less extreme than either of these two stable states, with phytoplankton and submerged macrophyte populations coexisting.

A eutrophic pond will generally reach the turbid state once a certain threshold nutrient concentration has been reached (Scheffer and Van Ness 2007). Depth, size, and climate can also be important factors, but since our ponds were purposefully similar in these aspects and because Gilmore Pond had the highest levels of phosphorus, nutrient composition likely drove Gilmore Pond into a turbid state (Scheffer and Van Ness 2007).

Peacock Pond was a clear water pond dominated by macrophytes, perhaps because it had not yet surpassed its nutrient threshold. *E. canadensis* is also tolerant of a wide range of light intensities, making it a suitable species to dominate in a clear-water pond (Erhard and Gross 2006). Another competitive advantage that *E. canadensis* has over phytoplankton is allelopathy. Studies have shown that *E. cadanensis* is capable of limiting cyanobacteria and epiphytic algae growth (Erhard and Gross 2006). Peacock

Pond contained a smaller proportion of cyanobacteria than Gilmore Pond, which could be a reflection of inhibition by *Elodea*.

5. Management

The following is a discussion of management options for Gilmore Pond. Management plans cannot be guaranteed and results will vary.

In any management plan, the first step is always to identify and eliminate the nutrient inflow source (Smith and Schindler 2009; Lurling 2013; Hilt et al. 2006; Carpenter 1998). From the results of water testing, it appears that phosphorus may be entering through the site 1-inflow pool. There could be other inflow sources as well, so sampling over more areas would be beneficial in finding other inflow sites. The two sites used for this study were selected by the community member who initiated sampling during the summer. He identified sites he thought were likely phosphorus sources and determined those two sites to be of the greatest interest (Breecher 2013). Any inflows caused by construction will be discontinued when the development is complete, but may unfortunately be replaced by lawn fertilizer runoff.

5.1 No Action

The easiest option in management is to do nothing at all. For Gilmore Pond, this might be the best course of action. Two main objectives for Gilmore Pond, based on communication with concerned community members, were that the pond be aesthetically pleasing and healthy (Fox 2013, Burn 2013, Westborough Community Landtrust 2013). Aesthetics cannot be objectively determined, but after spending many days collecting data at Gilmore Pond I do not believe that it is lacking in beauty (Figure 1a).

Unfortunately, society can have high aesthetic expectations of freshwater bodies that are particularly hard on small ponds. As previously stated, eutrophication in its essence is a natural process of filling in, which would still be happening even without human disturbances to the watershed (Olem and Flock 1990, Hutchinson 1967). It is not necessarily cause for distress. Even with extensive management some lakes and ponds will never be able reach a clear water state, though it is an objective many lake users desire (Olem and Flock 1990). It is when eutrophication is culturally enhanced that it can be problematic for ecosystems. As cultural eutrophication occurs at a more rapid pace, it can overwhelm ecosystems and lead to depletion of dissolved oxygen and a consequential decrease in biodiversity. However, if these effects are not being observed, eutrophication in a pond is probably natural and harmless.

"Health" is also difficult to determine. In beginning this research, health was largely regarded as equivalent to eutrophic: a healthy pond would not show signs of eutrophication. In addition to using algae as bioindicators of eutrophication, macroinvertebrate and amphibian life were also sampled to be used as bioindicators, as suggested by limnologists (Menetrey 2005, Oertli et al. 2005). Unfortunately, to be reliable eutrophication indicators, organisms needed to be identified to species, which was not possible. Nonetheless, the results of the surveys were still useful in answering the question of health. A diversity of macroinvertebrates, especially mayfly, damselfly, and dragonfly nymphs, would be represented in a non-eutrophic system (Menetrey 2005). These populations were underrepresented at Gilmore Pond, compared to the Wildcat and Peacock Pond (Table 5). This is likely the consequence of Gilmore being devoid of

submerged macrophytes, which provide important habitat and refuge for macroinvertebrates (van Donk and van de Bund 2002).

One symptom of cultural eutrophication that Gilmore Pond did not exhibit was dissolved oxygen depletion. With the exception of one data point, the dissolved oxygen levels were consistently above the 5 mg/L concentration, the optimum concentration for fish (Francis-Floyd 2013). Furthermore, Gilmore Pond had a higher overall range of dissolved oxygen than Wildcat Pond, which was not identified as eutrophic.

To look at health in a different way, a foodweb was constructed as a means of evaluating the interactions within the ecosystem (Figure 13a). A healthy pond would be expected to have a functioning food web, so applying the reverse logic does not seem unreasonable. Aside from the lack of macrophytes and nymphs, the food web created for Gilmore Pond appears to be full. It supports first level consumers such as zooplankton, which are important predators of algae, as well as detritivores, such as snails that perform important nutrient cycling roles (Voshell 2002). Second level consumers, such as beetles, crayfish, and fish were present as well indicating that they are able to find enough food to subsist on while still being preyed upon by higher predators like hawks and snakes (Reid 2001, "Cornell Lab of Ornithology: All about the birds", Voshell 2002).

Even without management in terms of treatment, further pursuit of phosphorus inflow sources is advised.

5.2 Active Treatment

If a clear-water state is desired, macrophyte reestablishment is essential (van Donk and van de Bund 2002). Macrophytes have an important role in pond ecosystems. As primary producers, they create competition for resources with phytoplankton and provide food for fish (van Donk and van de Bund 2002). They provide shelter for fish eggs and macroinvertebrates, structural support for periphyton, and refuge from predation for zooplankton and small fish (van Donk and van de Bund 2002). Gilmore Pond used to have a population of native coontail, and it is likely that viable seeds of these plants are still present in the sediment (McComas 2003). To allow these seeds to germinate, turbidity would need to be decreased by reducing phytoplankton populations so that sunlight can penetrate deeper into the water column (Scheffer and van Ness 2007).

Stopping inflows may not be enough to achieve this goal, because once a pond reaches a stable state of turbidity, it can be difficult to fall out of it (Phillips et al. 1978 and Meijer et al. 1989, as cited by Scheffer and Van Ness 2007). The following sections will address the ultimate management option recommended by ACT and offer an alternative suggestion.

5.2.1 Phosphorus binding

The final word against aluminum sulfate, or alum, treatment in Gilmore Pond is that it targets phosphorus concentrations, which was not identified as a major concern in this study.

ACT recommended that Gilmore Pond be treated with application or alum in 2012. Alum acts by binding phosphorus and inactivating it in the sediment (Cooke et al. 1993, Gensemer 1999). When ACT made this recommendation, total phosphorus concentrations were measured at much higher levels, 0.15 mg/L compared to 0.012 mg/L. At the levels of phosphorus measured in this study, phosphorus does not present itself as a top priority because it is lower than the eutrophication threshold of 0.03 mg/L (Olem and Flock 1990). However, these levels do not include phosphorus in the sediments, which can greatly contribute to internal phosphorus loads. Alum is effective at binding phosphorus in the sediment, and may be more useful if sediments do have high concentrations. Sediments should be tested for phosphorus before any management strategy is selected. Perhaps most importantly, successful alum treatment requires that nutrient inputs be addressed and eliminated, or action will have been for nothing (Cooke et al. 1993).

Alum has been successfully used in many instances of lake management (Olem and Flock 1990, Cooke et al. 1993, Welch and Shrieve 1994). Though there is little research about the use of alum in ponds, specifically, it has seen high success in non-stratified, shallow lakes (Cooke et al. 1993; Welch and Shrieve 1994). Gilmore Pond is a suitable candidate for alum treatment because it has hard water and circumneutral pH. In acidic water (pH<6), aluminum can change into soluble forms that are toxic to biota. Hard water ensures that the pH drop that occurs when aluminum sulfate disassociates into $Al(OH)_3$, the compound that binds phosphorus, is not so significant as to allow further aluminum transformation to take place (Cooke et al. 1993). Alum has successfully reduced phosphorus concentrations and phytoplankton populations in many lakes and has

not been reported to bioaccumulate in fish populations or cause significant fish mortality (Cooke et al 1993). There have even been reports of macrovegetation populations expanding as a result of increased water clarity (Welch and Shrieve 1994).

Despite the successes of alum, it is still not highly recommended as a management strategy for Gilmore Pond. The strongest argument against the use of alum is financial. Effects of alum are not permanent and may only last around five years (Olem and Flock 1990, Cooke et al. 1993). It is an expensive management plan that would cost upwards of $6,500 by ACT's estimates for one year's treatment (ACT 2012). Multiple applications may also be required, to ensure the appropriate amount is used to reduce phosphorus levels to the desired concentrations (ACT 2012). Although Gilmore Pond should be protected against toxic effects associated with alum treatment in soft and low pH waters, aluminum does not naturally occur in large quantities in freshwater bodies, so unknown toxic effects are a potential danger (Gensemer 1999). There have also been reports of decreased invertebrate diversity as an effect of treatment (Cooke et al. 1993). Furthermore to look at the larger picture, which is especially important in this age of exhaustive resource use and high waste production, to produce and apply alum is likely to be more environmentally damaging than the positive environmental effects that Gilmore Pond will experience as a result of its application. Factory production and shipping industries both have high ecological footprints that are devastating many of the earth's ecosystems, including freshwater bodies. In 2011, the EPA reported the Chemical Manufacturing Sector as one of the largest natural gas consumers (EPA 2011). This sector was also responsible for emitting 1.5 tons of criteria air pollutants, and disposed

500 million tons of chemicals in the water or air (EPA 2011). For Gilmore Pond, total phosphorus concentrations do not merit treatment with alum.

5.2.2 Barley hay

A less invasive and less expensive treatment that aims to achieve a similar outcome to alum is the application of barley hay. Barley hay acts as a natural algae control technique by releasing chemicals that inhibit the growth of algae (hUallachain and Fenton 2010). Studies have shown inconsistencies in results, but the most success has been recorded in Europe, where the majority of research on ponds has been focused (Lembi 2002). In a lab setting, barley hay as caused growth inhibition of *Microcystis*, a top cyanobacteria species in Gilmore Pond (Ferrier et al. 2005). Gilmore Pond is also suitable for barley hay treatment because it has high dissolved oxygen levels and long retention time (pers. obs.). Like alum, barley straw would likely require multiple applications, but at a much lower cost and without fear of lingering toxic impacts (McComas 2003).

Ultimately, barley hay has had mixed results in pond and lake management, but it is very low risk (hUallachain and Fenton 2010,). In a pond, like Gilmore Pond, that has not shown symptoms of an ecological catastrophe, barley hay would be a good management option if action to improve conditions were desired.

6. Conclusion

Gilmore and Peacock Pond are among the many freshwater systems in the world that have become impacted by cultural eutrophication. Shallow lakes and ponds often take on one of two stable states when they have become eutrophic. Gilmore Pond is in a turbid state, dominated by phytoplankton, and Peacock Pond is in a clear-water state, dominated by submerged vegetation. However, both ponds also have food webs with the characteristics of a typical pond. The turbid state can be difficult to overcome, and with ecosystem in the state that is, an intensive management program for Gilmore Pond is not recommended. Active management is not required to transform the pond into a healthy ecosystem, because it is satisfactory in its present state. If a clear-water state is desired, barley straw would be the recommended course of action, because it is low risk and inexpensive.

This study contributes to the pool of knowledge that is developing about pond ecology and eutrophication. Based on water chemistry analysis, this study presents the possibility that the nutrient threshold level indicating eutrophication in lakes should be altered for the study of ponds, because of their unique properties. It may be important to pursue this possibility with haste, because eutrophication is among the many current environmental issues associated with global climate change (Scheffer and van Ness 2007; Paerl and Paul 2012). Biota of shallow lakes and ponds are predicted to be impacted by rising temperatures. Rising temperatures would favor cyanobacteria populations, because their optima growth temperature is higher than that of other phytoplankton, augmenting the problem of harmful cyanobacterial blooms in eutrophic systems (Paerl and Paul 2012, Roberts and Zohary 1987). Extreme weather events associated with climate change are

also likely to impact the stability of alternative states in eutrophic ponds, because extreme events can prompt shifts between turbid and clear water states (Scheffer and van Ness 2007). With the uncertain environmental future posed by climate change, there is a need for continued research on ponds and the unique way they experience cultural eutrophication.

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