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## Impacts of Shoreline Development on the Littoral Zone of Great Pond

Problems in Environmental Science course (Biology 493), Colby College

Colby Environmental Assessment Team, Colby College

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# IMPACTS OF SHORELINE DEVELOPMENT ON THE LITTORAL ZONE OF GREAT POND



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WATERVILLE, ME 04901



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Impacts of Shorline development was conducted by the students of Biology 493: Problems in Environmental Science class at Colby College in Waterville, Maine.



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# **Executive Summary**

## *Introduction*

The Colby Environmental Assessment Team (CEAT) investigated the littoral zone of Great Pond in the Belgrade Lakes region of central Maine. Data collection occurred in September and October and analysis followed in October and November of 2010. Physical, biological, and chemical parameters were assessed to examine the impacts of shoreline development on the health of the littoral community. The littoral zone and adjacent riparian areas were sampled to allow comparison of aquatic and terrestrial parameters among different levels of shoreline development.

Shoreline development disrupts ecosystem function by altering the physical conditions that determine biological community structure. CEAT categorized study sites by land use, representing a range of development levels. The five categories of land use were defined as: wetlands, undisturbed forests, residential land with no buffer between the developed area and the lake, residential land with a 1-5m wide buffer, and residential land with a buffer greater than 5m wide. CEAT sampled a total of 74 sites in the littoral and riparian zones of Great Pond. The variables measured and recorded at each site include water nitrates, total dissolved phosphorus, hardness, soil phosphorus, littoral microhabitat cover, shore grade, lake floor grade, dominant riparian vegetation, macrophyte cover, substrate composition, canopy cover, zooplankton, fish, and macroinvertebrates. Biological indices of diversity, richness, and tolerance to pollution were employed during analyses.

Data analysis indicated relationships within and among the subsets of variables studied. Significant differences in parameters among land use types supported several of



CEAT's predictions. These differences may be a result of varying levels of shoreline development. Some differences, however, may be attributed to habitat conditions that are not necessarily consequences of shoreline development. Following is a brief summary of findings from the study of Great Pond, conducted by CEAT in 2010:

### *Water Quality*

- The highest mean water nitrate concentration ( $0.6\text{ppm} \pm \text{SE } 0.12$ ) was found at sites with no buffer, while the lowest ( $0.16\text{ppm} \pm \text{SE } 0.04$ ) was found at wetland sites. Nitrate levels in samples collected on 14-October-2010 at Jamaica Point were significantly higher than nitrate levels found elsewhere in the lake. It is unclear if this discrepancy is due to sampling error or if the data reflects circumstances contributing to high levels of nitrates at this location. When the 14-October-2010 data were included in the analysis, nitrate levels at no buffer sites were found to exceed naturally occurring levels (EPA 2010b) and nitrate levels differed significantly among land use types.
- The highest mean total dissolved phosphorus (TDP) level was found at sites with no buffer ( $0.10\text{ppm} \pm \text{SE } 0.01$ ), while the lowest was found at wetland sites ( $0.02\text{ppm} \pm \text{SE } 0.002$ ). Mean TDP levels exceeded natural levels in all land use types except wetlands, but mean values were not significantly different among land use types. These findings indicate that Great Pond may be susceptible to algal blooms.
- The mean hardness value observed by CEAT was  $13.6\text{ppm}$ , indicating that the water in Great Pond is soft. Soft water is more vulnerable to algal blooms and may compromise fish health.

## *Habitat*

- Wetlands had the highest mean percentage of cover by macrophytes (10.6%), followed by sites with no buffer (2%). The physical conditions at these sites are favorable for plant growth. Additionally, it is possible that macrophyte growth at no buffer sites is enhanced by nutrient loading.
- Shore grade varied significantly among land use types and was lowest at wetland and no buffer sites. Landowners may find it difficult to maintain cleared land on a steep grade, which supports the finding that sites with greater shore grades tended to be undisturbed or have vegetative buffers.
- The distribution of sediment types may reflect shoreline manipulation by landowners, both by the creation of artificial beaches and by deforestation of the shore. The consequences of shoreline manipulation may include erosion of fine sediments and increased vulnerability of the littoral habitat to non-anthropogenic forces, such as wind, water, and ice, which can move and sort sediments.
- A significant difference in substrate type was found across land use types, with muck occurring most often in wetland sites. Wetlands, often situated in less exposed parts of the lake, accumulate nutrient rich sediment and play an important role in nutrient cycling in the lake.
- CEAT expected to see a strong correlation between terrestrial and aquatic canopy cover, but this was not observed. It is likely that two measures of canopy cover taken at the center of the site were not representative of the site as a whole. The results may also be skewed due to modification of the recommended sampling



procedure and experimenter bias. Data may not be representative of canopy cover due to seasonal changes in foliage cover.

### ***Fauna***

- Zooplankton family richness was highest in sites with a 1-5m buffer (4.47 families per site  $\pm$  SE 0.36 ) and lowest in wetland sites (2.50 families per site  $\pm$  SE 0.48). Insufficient sampling effort may have influenced these values, since densities observed at undisturbed and wetland sites were lower than at residential sites. The water column at many wetland sites was shallow, and wetland samples often contained sediment and lower volumes of water.
- The inefficiency of seining in complex habitats resulted in low fish counts that were unlikely to be representative of actual fish populations in Great Pond. These data were used only in descriptive analyses of the lake's fish populations. Three fish species that were captured had not been previously recorded in Great Pond.
- Macroinvertebrate analyses produced significant differences in Family Biotic Indices (FBIs) between different land use types. However, the percentage of families representing the pollution-intolerant orders Ephemeroptera, Plecoptera, and Tricoptera (EPT) found at sites of different land uses did not differ significantly, indicating that there may not be significant differences in biotic conditions across land use types.
- Rusty crayfish (*Orconectes rusticus*) were found in Great Pond. The presence of this invasive species merits further investigation and management plans to prevent the movement of this species throughout the Belgrade Lakes region.

## *Recommendations*

### *For Future Study*

CEAT made the following recommendations for conducting future studies of the impact of shoreline development on the littoral zone of Great Pond:

- Sample in a greater number of locations and in a more condensed timeline. This will provide results that better represent ecosystem functions and that can be compared with greater accuracy.
- Sample at a time of year with more consistent weather and lake conditions, such as the summer. In addition, longitudinal studies conducted at different times of the year and considered cumulatively to account for seasonal processes would more thoroughly reflect overall lake conditions.
- Include another land use category to assess the impacts of shoreline structures, such as docks, on littoral communities.
- Expand study parameters to include chlorophyll a, turbidity, and nutrient levels of lake floor sediment. These variables may provide details about processes impacting the lake, such as eutrophication, that may be helpful for lake management.
- Replicate water quality tests, such as nitrates and TDP, to reduce error in acquiring and testing a representative sample. Use more sensitive water quality tests, particularly for phosphorus, that can measure differences in parts per billion (ppb) rather than parts per million (ppm).

- Use multiple gear types to sample fish populations. Electro-fishing with backpack shockers and boat shockers are effective methods for fish sampling. These methods will allow researchers to more efficiently sample all microhabitat types.

These recommendations for the design and methodology of future studies will allow researchers to better understand ecosystem-scale functions and develop management plans that incorporate the physical and biological conditions of Great Pond.

#### *For Landowners and Lake Managers*

CEAT made the following recommendations for landowners to minimize the impacts of shoreline development on the littoral zone of Great Pond:

- Maintain vegetative buffers to protect the shoreline from erosion. This involves minimizing further manipulation of the shoreline, allowing woody plant growth, and not removing microhabitats, such as woody debris, from the littoral zone.
- Support wetland protection and avoid development within or adjacent to wetlands.
- Monitor non-native macrophyte, macroinvertebrate, and fish species. Learn to identify invasive species and understand their effects on ecosystem health. Work to remove them from and prevent future entry into Great Pond.
- Monitor phosphorus and nitrate levels and sources both spatially and seasonally. This can be achieved by testing water quality and conducting landowner surveys around the lake to investigate the use of phosphate-containing chemicals.
- Avoid further alteration of the natural shoreline. In addition, restrict the input of external sediments and monitor natural sediment build up.
- Promote and engage in education about lake health.

These recommendations will contribute to improved water quality and health of biological communities in the littoral zone of Great Pond.



# Introduction

## *General Nature of Study*

Understanding the dynamic processes that function within freshwater ecosystems is critical to the preservation and management of lakes and ponds, which are an important resource to the state. Maine has the greatest total area of inland surface water in the northeastern United States (Davis *et al.* 1978). Maine's lakes are an invaluable resource to its residents, visitors, and environment. They provide opportunities for recreation and industry while also playing a critical role in nutrient cycling, hydrologic cycling, and habitat for flora and fauna. The intricate balances of biotic and abiotic factors in freshwater ecosystems are threatened by increases in human activity and development, endangering the vital services provided by lakes. Land use changes, modification of aquatic environments, increases in chemical and organic inputs, and the disruption of natural cycling contribute to the degradation of water quality in Maine's lakes (MDEP 2010a; MDEP 2010d). The insects, fish, birds, and humans who depend on these ecosystems are threatened by adverse impacts of shoreline development.

In the 1970s, population growth and commercial expansion were considered threats to Maine's environmental integrity and resource availability (Davis *et al.* 1978). Decades later, the effects of shoreline development are still an important consideration for the residents of Maine. Understanding these and other underlying processes that drive environmental degradation are critical for managing landscapes and natural resources. The Colby Environmental Assessment Team (CEAT) examined the impacts of shoreline

development on the habitats, water quality, and faunal composition of the littoral zone of Great Pond, in the Belgrade Lakes region of central Maine.

## ***Background***

### **Trophic Status**

Trophic status is a metric used to characterize the condition of lakes, based on the level of nutrient enrichment of the water (Henderson-Sellers 1987). It is determined by three major factors: the rate of nutrient supply, climate, and the morphology of the lake basin (Lake Access 2005). Vegetation, soils, development, management, and the bedrock geology of the watershed influence the rate of nutrient supply. Climate affects trophic status through temperature, amount of sunlight, lake basin turnover time, and precipitation. Depth, volume, and surface area are parameters that describe the lake basin. The ratio of the watershed to the lake surface area will also determine the morphology of the lake basin (Lake Access 2005). Lake health ranges from oligotrophic, having low levels of nutrients, to eutrophic, having high levels of nutrients. Intermediate lakes with a trophic level between eutrophic and oligotrophic are described as mesotrophic (Dodds 2002). In addition to nutrient levels, oligotrophic lakes are characterized by high concentrations of dissolved oxygen, low levels of suspended particulates, great depths, and extensive light penetration. Eutrophic lakes contain low levels of dissolved oxygen, high levels of suspended particulates, shallow depths, and poor light penetration. For example, mean annual secchi disk depths, a metric that reflects suspended particulates in the water column, are greater than 6m in oligotrophic lakes and 1.5-3m in eutrophic lakes (Henderson-Sellers 1987).

The natural progression of lakes from oligotrophic to eutrophic, a process known as eutrophication, is related to geologic processes and requires thousands of years (Harper 1992; Dodds 2002). In comparison, cultural eutrophication, an anthropogenic influx of nutrients, occurs rapidly. Human activities that cause cultural eutrophication include use of fertilizers, livestock practices, watershed disturbance (e.g., deforestation), and contamination by nutrient-rich sewage. It is critical to identify and manage these causes to prevent nutrient loading, algal blooms, anoxic conditions, poor water quality and low aquatic biodiversity (Jones and Lee 1982; Wetzel 2001; USGS 2010a).

In aquatic systems, the growth of aquatic plants and algae is limited by the availability of nutrients. Phosphorus, in particular, restricts aquatic plant and algae development in Maine lakes (Marsden 1989). In eutrophic lakes, where nutrient levels are elevated, the productivity of the lake is increased (Harper 1992). Nutrient loading can result in a sudden spike in algae growth, known as an algal bloom. Increases in primary production and the respiratory uptake of oxygen by algae can lower dissolved oxygen levels. When algal blooms decompose, aerobic decomposers further deplete dissolved oxygen levels. This can lead to hypoxic, or abnormally low dissolved oxygen levels, or anoxic, meaning depleted dissolved oxygen levels. Lack of oxygen has adverse effects on the aquatic community. In extreme cases, widespread fish mortality, termed fish kills, may result from anoxic conditions associated with cultural eutrophication (Cole 1983).

Development is a primary cause of cultural eutrophication. It can disrupt terrestrial nutrient cycling and lead to increases in nutrient loading of runoff (Harper 1992). In areas that have residential, commercial, and agricultural development, nitrogen and phosphorus are found in higher concentrations than in undisturbed areas.

Deforestation of the shoreline, a frequent effect of lakeshore development, is a significant mechanism of cultural eutrophication (O'Sullivan 2005).

## **Littoral Zone**

When considering trophic status, it is imperative to understand the structure of the littoral zone, which plays a major role in the overall productivity of a lake (Wetzel 2001). The littoral zone of a lake extends from where the influence of wave action ends at the shoreline to the depth at which light penetration is no longer sufficient for photosynthesis (Goldman and Horne 1983). Aquatic plants growing in this zone are known as macrophytes, and can be either emergent (having some parts above the water surface) or submergent (having all parts submerged; Goldman and Horne 1983). The size and depth of the littoral zone varies with the morphology of the lake basin and the water clarity. A number of schemes of varying complexity have been devised to subdivide the littoral zone (Wetzel 2001; Kalff 2002). The simpler models break the littoral zone into three sections: the upper littoral section encompasses the shoreline and the area containing emergent macrophytes; the middle littoral section extends from the upper zone to the point where rooted macrophytes are no longer found; and the lower littoral section extends from the middle littoral zone to where light is just sufficient for photosynthesis (Kalff 2002). However, the influence of the littoral zone extends beyond these borders and is vital to overall lake health (Freshwater Society 2004).

A variety of factors such as light availability, nutrient accessibility, and warmer temperatures make the littoral zone the most diverse and productive region of a lake (Kalff 2002; Ness 2006). The assemblage of organisms living and interacting in this zone



is referred to as the littoral community; the maintenance of this community's integrity is critical to the overall health of the lake (Freshwater Society 2004). The decomposition of littoral macrophytes and the algae living on them contribute to an extensive stock of dissolved and particulate organic matter. The production of this organic matter and associated detritus provides food and habitat for zooplankton, macroinvertebrates, and vertebrates such as fish and amphibians (Wetzel 2000; Freshwater Society 2004). CEAT defined zooplankton as tiny free-swimming organisms, including crustaceans, which mainly eat algae. Macroinvertebrates are defined as organisms that lack a backbone and are visible without the aid of a microscope (Voshell 2002).

The littoral zone is spatially heterogeneous, both horizontally and vertically, due to lake geomorphology, diversity of microhabitats, and varied distribution of aquatic macrophytes. This makes sampling difficult, leading to a paucity of information regarding littoral flora and fauna as well as a disproportionate emphasis on the pelagic, or open water, areas of lakes (Wetzel 2000).

The littoral zone is important not only for its extensive productivity and diversity, but also for its function as an interface between aquatic and terrestrial systems. This results in habitat coupling, linking the benthic (lake floor), pelagic, and riparian (onshore) zones (Schindler and Scheuerell 2002). These complex and varied interactions involve diverse biota and processes that allow for the transfer of energy between systems. These relationships include the movement of organic detritus from littoral areas to deeper parts of the lake, the consumption of terrestrial insects by fish, and riparian leaf litter falling into the water (Wetzel 2000; Schindler and Scheuerell 2002). The leaf litter input, as well as other riparian-littoral interactions, are of particular interest to the study of the impact

of shoreline development on the littoral zone. Destruction of riparian habitats is a frequent consequence of human disturbance. It is therefore important to understand how the riparian and littoral zones interact.

### **Littoral-Riparian Interactions**

A number of important interactions between the littoral zone and the associated riparian habitat are instrumental in maintaining lake health. The riparian zone is the transitional region of the shoreline that connects aquatic and terrestrial habitats. These interactions can be either direct or indirect. Direct interactions involve the contribution of organic materials to the lake (Schindler and Scheuerell 2002). Contributions of matter originating outside the lake from surrounding riparian vegetation can be substantial (Rau 1976; France and Peters 1995). For example, as much as 15% of the carbon supply and 10% of the phosphorus supply for large oligotrophic lakes comes from leaf litter (France and Peters 1995). In addition to being a source of nutrients, leaf litter is utilized as both food and habitat. For example, caddisfly larvae incorporate leaf litter into their casings (Wetzel 2001). Given the importance of riparian vegetation as a nutrient source, the detrimental effects associated with loss of onshore habitat due to development are not surprising (Rau 1976). Studies examining the impact of riparian zone deforestation and subsequent loss of leaf litter found that carbon and phosphorus inputs were significantly lower in deforested locations in nutrient-poor, oligotrophic lakes (France 1995; France *et al.* 1996). These effects may be negligible in eutrophic lakes, however, in which nutrients from runoff more than compensate for the loss of riparian litter (France and Peters 1995; Carpenter *et al.* 1998).

Another important direct organic input from the riparian zone is coarse woody debris (CWD). Coarse woody debris serves a number of functions in lakes, including energy input through microbial decomposition and providing complex microhabitats for juvenile fishes and macroinvertebrates (Everett and Ruiz 1993; Zimmerman *et al.* 1995; Schindler and Scheuerell 2002). Additionally, these microhabitats help facilitate critical predator-prey relationships between aquatic fauna, as they provide refuge for small prey (Crowder and Cooper 1982).

These important interactions are threatened by increased shoreline development. Anthropogenic development contributes to loss of CWD through destruction of riparian vegetative sources, as well as direct removal of CWD from the littoral zone for aesthetic and practical purposes (Christensen *et al.* 1996; Marburg *et al.* 2006). Loss of CWD habitat has been shown to negatively affect fish assemblages by decreasing prey abundance and by altering the foraging strategies of predators (Sass *et al.* 2006). The accumulation of CWD in the littoral zone is an extremely slow process that can take centuries (Guyette and Cole 1999). For this reason, human disturbance and development are threats to the maintenance of CWD in direct riparian-littoral interactions.

Indirect interactions involve protection of the littoral zone by the riparian zone. Riparian vegetation acts as a buffer to runoff, thereby reducing the risk of nutrient loading from industry and agriculture. The loading of nitrogen and phosphorus as a result of riparian forest removal has been the primary focus of studies examining the effects of lakeshore development (Christianson *et al.* 1996). The increase in runoff of these nutrients resulting from deforestation can lead to many deleterious effects, such as algal blooms, fish kills, and other effects of eutrophication (Carpenter *et al.* 1998). Loss of

riparian buffers also leads to increased erosion of the shore and consequent increased sedimentation. This buffer reduction can result in damage to the macrophyte community and negative consequences for a host of organisms in and out of the littoral zone (Ness 2006). The indirect protection offered by the riparian zone is of critical importance to the littoral community.

### **Riparian Land Use Types**

Human activities directly impact riparian-littoral coupling by changing habitat structure and introducing pollutants that can harm the fragile ecosystems. The way in which humans utilize the land and its resources is known as land use (Naiman *et al.* 2005). CEAT focused on the effects of three different land use types on fragile riparian-littoral interactions: developed, undisturbed, and wetlands. Severity of lake contamination is directly related to the type and intensity of shoreline land use (LORC 2000; Gove *et al.* 2001; Tang *et al.* 2005).

#### *Developed Riparian Land*

Developed land has been altered from its natural state and can be characterized as residential, agricultural, and industrial uses. Highly developed areas that are cleared of shoreline vegetation have detrimental impacts on water quality. A lack of shoreline vegetation increases the rate of runoff and erosion, enhancing the likelihood of contaminated sediment accumulation (LORC 2000). Deforestation in the watershed from human activities causes an increase in sediment influx into littoral communities, which negatively impacts water quality and reduces wildlife habitat (Alin *et al.* 1999). Within the developed land use category, CEAT focused on residential development. Residential areas are particularly harmful to the watershed because of the effects of household



chemicals such as detergents, herbicides, pesticides and fertilizers (Gove *et al.* 2001).

Some of these chemicals are a potent source of phosphorus, which contributes to cultural eutrophication (S.C.O.A.L.E. 1989).

### *Riparian Buffers*

A buffer is a permanent strip of woody vegetation surrounding a field, road, or body of water (LORC 2000). Their maintenance is pertinent to mitigation of environmental degradation. Riparian buffers grow along the edge of waterways, and the root systems of the vegetation absorb harmful pollutants that could leach into water sources via runoff or erosion. Root systems also stabilize the structure of the bank and limit erosion (Kalff 2002). In addition, buffers can act as wildlife refuges around water systems, increasing biodiversity (LORC 2000). Finally, vegetation in buffers provides shade to littoral fish populations. Without buffers, many aspects of the ecosystem are at risk of degradation. In the absence of a buffer, littoral communities experience a decline in water quality, increased flooding and erosion, and habitat loss (LORC 2000). Offshore, or limnetic, communities can also suffer from lack of buffers.

### *Undisturbed Riparian Land*

Undisturbed riparian areas are undeveloped and are characterized by trees and other woody vegetation that act as a buffer. Sediments in undisturbed areas are more likely to have lower contaminant concentrations (Alin *et al.* 1999; CEAT 2010). Sedimentation is also far less frequent in undisturbed regions because of the forested buffers. In some cases, there are dramatic increases in fish and mollusk species richness in undisturbed areas compared to areas where human activity is common (Alin *et al.* 1999). Undisturbed landscapes also play a critical role in providing habitat for wildlife.

Therefore, loss of biodiversity and habitat destruction is directly related to development (LORC 2000; Tang *et al.* 2005).

### *Wetlands*

A wetland is a region of land that is saturated with water and supports vegetation adapted to these conditions (EPA 2008). Wetlands are inhabited by a vast diversity of life, making them a key component in ecosystem function (Dodds 2002). The shallow water and abundance of nutrients promote growth of primary producers, which support many other species of fish, birds and mammals (EPA 2002). Organisms within wetlands play a critical role in biogeochemical cycling. Soil microbes transform nitrogen and sulfur into usable gases and plants convert inorganic phosphorus into usable organic forms (Whigham *et al.* 1988). Carbon stores in plant biomass also act as a climate regulator by preventing the release of carbon associated with global climate change (EPA 2002). Wetlands improve littoral and overall lake water quality by limiting erosion and absorbing inorganic wastes (EPA 2008; Whigham *et al.* 1988). Wetlands buffer nutrients, in some seasons acting as a sink by absorbing nitrogen, sulfur and phosphorus that in high concentration may harm the rest of the lake. Wetlands store and convert these limiting nutrients, as well as carbon, without releasing them into the atmosphere (EPA 2002). At other times, wetlands may act as a source of nutrients (Richardson 1985). Wetland habitats are crucial ecosystems that help maintain the health of limnetic systems.

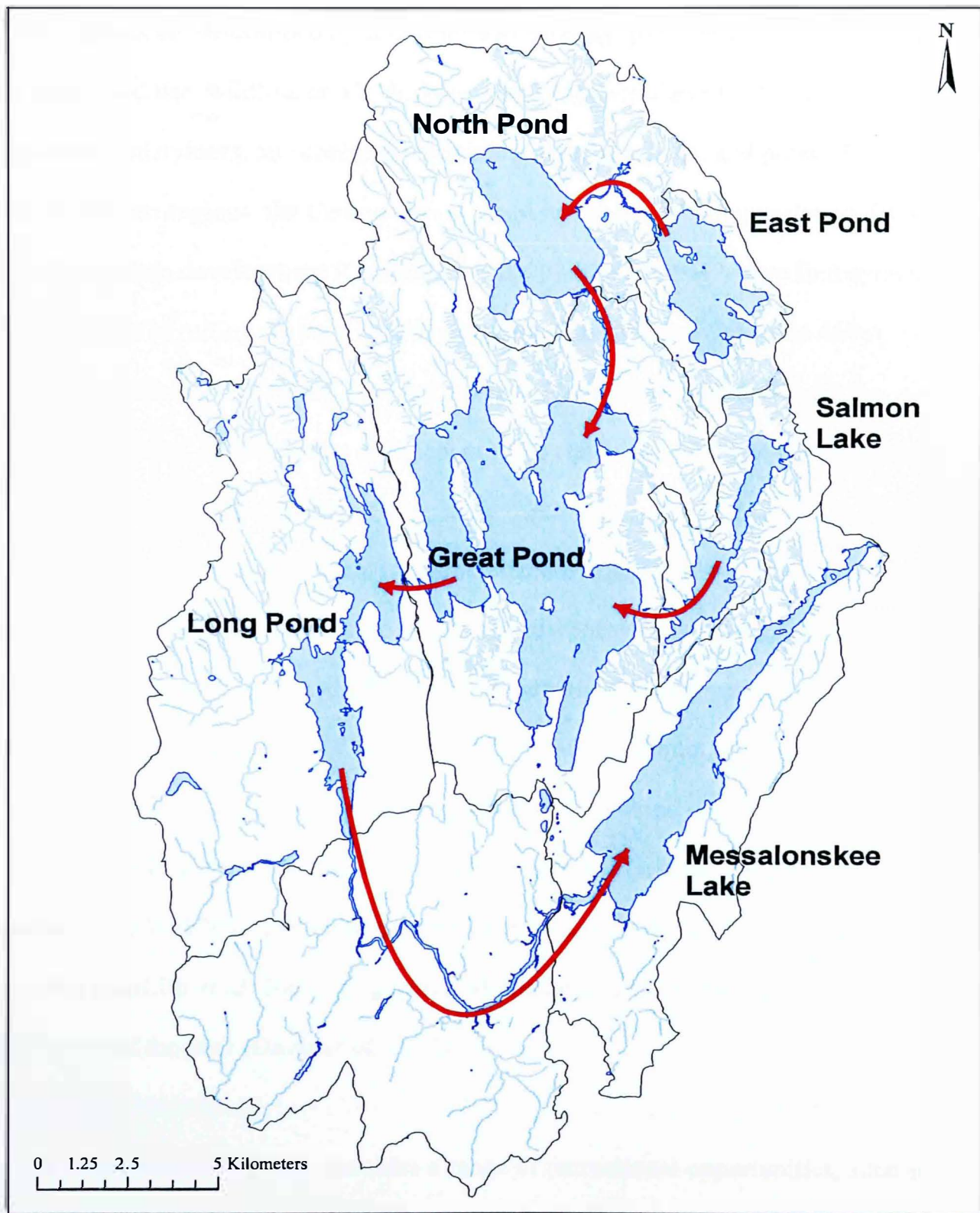
## *Area of Focus: Great Pond*

### **Physical Characteristics**

Great Pond is part of a chain of freshwater bodies known as the Belgrade Lakes. This system includes East Pond, Little Pond, North Pond, Salmon Lake, McGrath Pond, Long Pond, and Messalonskee Lake (Figure 1). The Belgrade Lakes drain into Messalonskee Stream which is part of the lower Kennebec River Watershed. Both North Pond and Salmon Lake are direct inputs into Great Pond, while Long Pond is the lake's only significant output (MDEP 2006).

Great Pond is the largest, most populated, and most heavily utilized of the Belgrade Lakes (BRCA 2000). Great Pond is located in the towns of Rome and Belgrade in Kennebec County, Maine. Its direct watershed extends into the towns of Oakland, Smithfield, and Mercer. Great Pond has a surface area of 3,453 hectares, a perimeter of 74,230m, and a maximum depth of 21m. It has a total drainage area of 215km<sup>2</sup> and a direct drainage area of 83km<sup>2</sup> (PEARL 2005).

The 1886 construction of the Great Pond Storage Dam on Belgrade Stream gave Great Pond its current shape. The dam, measuring 4.3m in height, was initially constructed for power generation. The dam was rebuilt to its current structure in the early 20<sup>th</sup> century (P. Kallin, pers. comm.). This dam regulates Great Pond water levels and is controlled by the Belgrade Area Dams Committee. The committee consists of representatives from the towns of Rome, Belgrade, and Oakland. Optimum fall and winter drawdown levels are 0.46 to 0.61m below the spillway and drawdown levels are not to exceed 0.80m below the spillway (BADC 2008). The maintenance and operation of the dam impacts the health, potential uses, and economic value of Great Pond (MDEP 2003).



**Figure 1.** Directional waterflow in the Messalonskee Stream Watershed.





## **Ecoregion**

Ecoregions are determined by differences in geology, physiography, vegetation, climate, soils, land use, wildlife, and hydrology. The Belgrade Lakes are located in the Central Maine Embayment, an ecoregion containing numerous lakes and ponds. Relative to Maine's other ecoregions, the Central Maine Embayment has a high population density and extensive human development (Griffith *et al.* 2009). The Central Maine Embayment is a diverse region of rolling plains and hills with elevations ranging from 6 to 460m above sea level. The mixed bedrock geology of the region includes pelite, sandstone, limestone, and granite. Glaciomarine sediments of silt, clay, sand, and gravel cover much of the lowlands (Griffith *et al.* 2009).

The Central Maine Embayment is characterized by the following forest types: white pine-mixed conifer forest; red oak-northern hardwoods-white pine forest; beech-birch-maple forest; oak-pine forest; silver maple floodplain. These forests are found in the watershed and riparian areas of Great Pond. They provide habitat, food, and environmental services to the organisms inhabiting the Great Pond ecosystem.

The region is between coastal and inland climates with a mean annual precipitation of 102–127cm. The mean number of frost-free days ranges from 140 to 170 days annually (Griffith *et al.* 2009). In general, Maine lakes are completely frozen for 4 to 5 months out of the year (Davis *et al.* 1978).

## **Recreation and Wildlife**

The Great Pond ecosystem provides a range of recreational opportunities, such as fishing, hunting, boating, and swimming. These activities form the base of the local tourism industry, which provides revenue and employment for residents. Approximately

80% of shoreline residences on Great Pond are seasonal (CEAT 1999), reflecting the importance of seasonal recreational activities to the people living in the region. All of these activities are dependent on the health of the region's lakes. For example, algal blooms change the color of the water and produce mats of decomposing organic matter that are undesirable for swimmers and boaters. Thus decreases in water quality have been shown to decrease property values (Michael *et al.* 1996)

Maine hosts a more diverse and abundant group of lake-dependent wildlife than any other northeastern state (Parkin 1989). Wildlife depend on the environmental health of the Belgrade Lakes region, in both terrestrial and aquatic landscapes. Wading birds and waterfowl rely on wetland ecosystems for habitat and food sources. Fish are important to piscivorous birds such as the great blue heron, common loon, hooded merganser, osprey, and bald eagles. The Belgrade Lakes can also provide crucial habitat for a number of bird species (Parkin 1989). Terrestrial organisms such as the white-tailed deer and moose rely on wintering sites in the Belgrade Lakes region. In addition, muskrat, beavers, river otters, mink, and fisher rely on healthy ecosystems throughout the Belgrade Lakes. Below the surface of the water, organisms from fish to microscopic zooplankton are directly dependent on lake water quality. Smallmouth bass, largemouth bass, landlocked alewife, perch, pike, and bullhead are found in Great Pond (PEARL 2005). Currently, brown trout and brook trout are also stocked in Great Pond (MDIFW 2010). Great Pond is host to a variety of human activities and animal inhabitants, and therefore should be protected.

## **Water Quality of Great Pond**

Riparian land use directly impacts water quality. In Great Pond, changes in land use have had consequences on nutrient loading, impacting the limnetic community (CEAT 1999). Anthropogenic activities, such as deforestation and development, increase the amount of pollutants and sediment runoff into littoral communities. Increased runoff can lower dissolved oxygen levels through algal blooms and other ecological phenomenon. In 2000, the Belgrade Regional Conservation Alliance was concerned with decreasing levels of dissolved oxygen in Great Pond. Low oxygen levels are detrimental to fish populations and all other aquatic wildlife. Furthermore, anoxic conditions release phosphorus from lake-floor sediments, exacerbating the nutrient-feedback cycle, resulting in decreased oxygen levels (BRCA 2000). Phosphorus is a limiting nutrient that human activity can artificially increase within an ecosystem, leading to algal blooms (Gove *et al.* 2001). Great Pond has not experienced any problematic algal blooms in the past. The total phosphorus concentration, however, has increased from 5-6ppb in 1977 to 6-8ppb in 2000 (BRCA 2000). To decrease phosphorus concentrations, anthropogenic activities must be altered. Monitoring Great Pond's water quality yields the most accurate indication of ecosystem health and is important in maintaining biodiversity in Maine's lakes.

## **Trophic Status of Great Pond**

Great Pond is currently considered a mesotrophic lake (PEARL 2005), though it was historically reported to be oligotrophic (Davis *et al.* 1978). This change in trophic status is a result of changes in rates of nutrient loading and primary production over the past three decades. One potential mechanism for increased cultural eutrophication in



Great Pond is deforestation of shoreline areas. Shoreline vegetation reduces runoff velocity and sediment-carrying capacity as well as preventing erosion by binding together soil particles (Henderson-Sellers 1987).

In 1999, CEAT predicted the total phosphorus inflow for the year 2015 based on population growth, land use, and development. They predicted an inflow of between 3224kg/yr and 8402kg/yr, which corresponds to a phosphorus concentration range of 5.8ppb to 15.1ppb. This range includes 15ppb, the value at which total phosphorus is considered to indicate a high potential for algal blooms (MDEP 2002). Monitoring indicators of trophic status, such as total phosphorus and nitrogen as well as bioindicators contribute to the assessment of changing lake conditions in Great Pond.

### **Habitats of Great Pond**

The shoreline habitats of Great Pond vary and perform a number of different functions in the Great Pond ecosystem. Mixed hardwood and conifer forest types offer significant habitat for wildlife and provide essential ecosystem services in the watershed, such as filtering runoff, moderating littoral temperature, and preventing shoreline erosion (Naiman *et al.* 2005). Other shoreline habitats consist of developed and transitional sites which are areas recovering from previous disturbance

Littoral habitats in Great Pond are comprised of sand, rock, CWD, small woody debris (SWD), rootwads (exposed tree roots extending from the riparian zone into the littoral zone), and macrophytes (personal observation 2010). These microhabitats are influenced by adjacent terrestrial ecosystems. For example, a forested site will contribute twigs and branches to CWD and SWD microhabitats in the littoral zone, whereas a non-forested site may lack these microhabitats. Similarly, a shoreline area with high nutrient

run-off will likely correspond to a littoral community with a greater abundance of macrophytes, which are often limited by phosphorus (Wetzel 2001). Littoral habitats are important to overall ecosystem health because they provide important shelter and food sources to aquatic organisms such as fish and macroinvertebrates. When assessing the populations of these organisms, it is critical to consider the available littoral habitats and their interactions with adjacent riparian habitats.

### ***Study Goal***

The goal of this study was to assess impacts of shoreline development on the littoral zone of Great Pond. CEAT investigated terrestrial and aquatic habitats, zooplankton, macroinvertebrate, and fish communities, as well as several water quality parameters. The sites in the study ranged from undisturbed (forested) to no buffer (lawn or beach). The objective of the study was to determine how land use may influence littoral biological communities and water quality. This assessment will contribute to an understanding of the health of Great Pond's littoral communities and guide future management of shoreline development.

# Methods, Results, and Discussion

## *General Sampling*

### *General Sampling Methods*

CEAT sampled 74 sites around Great Pond across five different land use types (Figure 2). This study was designed to assess the effects of shoreline development on the littoral zone of Great Pond. Therefore, the designated categories of land use that determined sampling site selection reflect differing levels of development. The five categories were defined as follows: wetlands, undisturbed forests, residential land with no buffer between the developed area and the lake, residential land with a 1-5m wide buffer between the developed area and the lake, and residential land with a buffer of greater than 5m. These categories were developed from land use maps of Great Pond and modified shoreline disturbance classifications used by Butler (2008). Buffers were defined as areas of woody vegetation at least 1m in height. CEAT sampled 14 wetlands, 15 undisturbed, 15 1-5m buffer, 14 >5m buffer, and 16 no buffer sites. All sites were visited between 1pm and 5pm. Sites were accessed on foot. Due to time constraints, certain sites were clustered spatially.

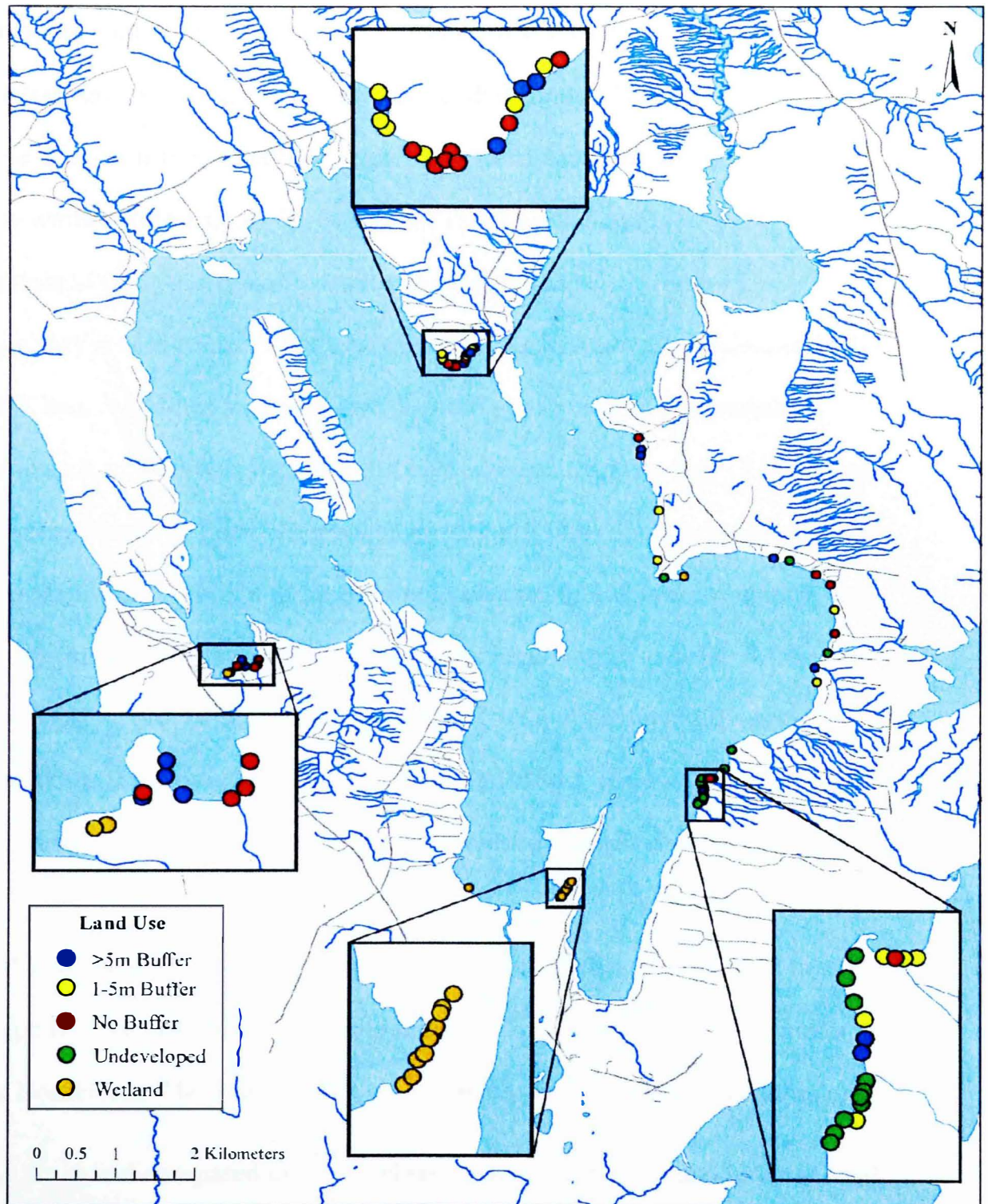
Sampling sites were selected to obtain similar numbers of sites in each of the five land use categories. Width of buffer was determined by perpendicular distance from the shoreline. Since the continuity of buffers may be more important than their width (Kalff 2002), the effects of neighboring land use types on the littoral community were minimized by placing each site at least 10m away from differing land use categories. Each site consisted of two parts: an aquatic portion, where measurements and samples were taken in the littoral zone, and an adjacent terrestrial portion where measurements

and samples were taken from the riparian zone. The aquatic portion of each site was 10m long and extended 3m from the shore into the water. Terrestrial sampling extended up to 2m inland from the water's edge along the length of the aquatic portion. Before data collection, the boundaries and center point of each site were marked with flags to ensure that sampling occurred within the specified area.

Once sampling sites were established, water quality, habitat structure, and fauna assemblages were assessed in a specific order. The order was determined to minimize the impacts of each sampling method on the subsequent ones. For instance, zooplankton tows are impacted by introduction of sediment to the water column, potentially from a beach seine, and fish can be chased out of the site by sampling for macroinvertebrates. Therefore, the first procedure was visual inspection of terrestrial plants and then a water sample was taken. pH and temperature were measured using a handheld meter, but due to meter inconsistencies this practice was stopped before completion of the study. The shore grade was measured and an on-shore soil core was collected as zooplankton samples were taken. This was followed by beach seining to sample fish. Next, all present microhabitats were sampled for macroinvertebrates and macrophyte quadrats were placed for visual estimation and identification. Finally, the canopy cover was estimated and the lake floor slope was measured.



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**Figure 2.** Seventy four sites were sampled along the littoral zone of Great Pond; these are color coded to show land use categories associated with each sample site.



## General Statistical Methods

Multivariate analysis of variance (MANOVA) was used to test for differences in littoral parameters among land use types and spatial distribution. The MANOVA was chosen because of its ability to handle multiple dependent variables and rigorously test for differences while controlling for Type I error (Tabachnick and Fidell 1996; Green *et al.* 2000; Scheiner 2001). Additionally, testing several dependent variables simultaneously may reveal complex patterns missed by univariate tests (Scheiner 2001). The MANOVA was conducted to investigate if water quality and habitat variables, organism abundance, and diversity metrics differed between land use type (5 levels) and spatial distribution (clusters of sampling points around the lake). In order to ensure that all variables used in the MANOVA were truly independent, a bivariate correlation was conducted to determine collinearity. If variables were significantly correlated, one was selected for the MANOVA. Wilk's lambda was used to calculate the multivariate F-statistic (Stata 2009). If the overall MANOVA was significant, separate analyses of variances (One-way ANOVAs) were conducted to investigate each dependent variable separately.

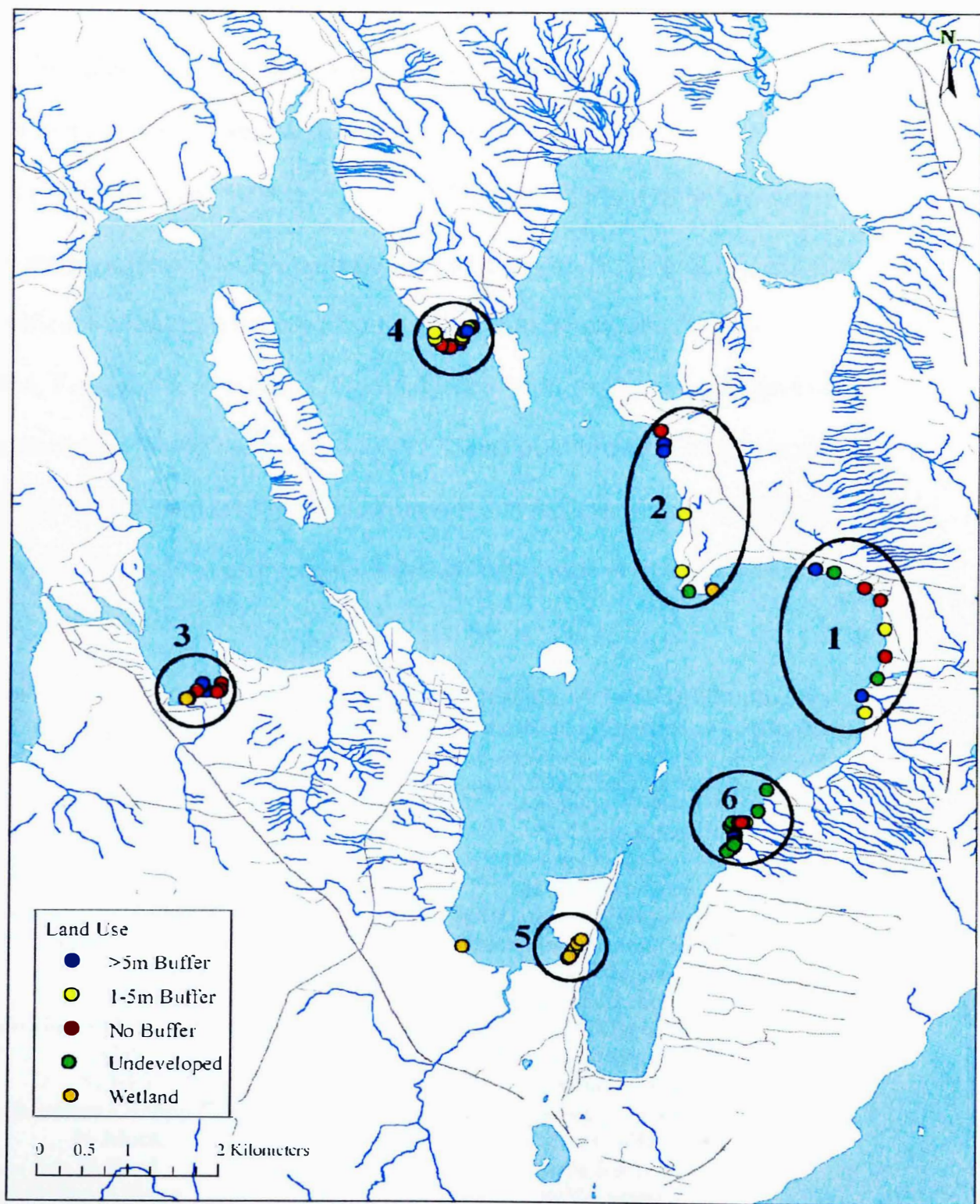
The false discovery rate (Benjamini and Hochberg 1995) was employed to control Type I and Type II error rates for the multiple individual one-way ANOVAs. Similar to the sequential Bonferroni (Hochberg 1988), the p-values were ranked in ascending order ( $p_{(1)} < p_{(2)} < \dots < p_{(m)}$ ) and compared to  $\frac{\alpha * i}{m}$ , where  $i$  = rank of p-value and  $m$  is the total number of tests. The null hypothesis ( $H_0$ ) was rejected when  $p \leq \frac{\alpha * i}{m}$  (Benjamini and Hochberg 1995; Verhoeven *et al.* 2005; Strakosh 2009). Type III sums of square were



used in all analyses. All tests, including individual one-way ANOVAs, were conducted in Stata (2009).

Since this study was conducted by sampling on foot from several access points, the sites are clustered on a spatial basis. If several sites of the same land use type were situated in close proximity to each other, these sites may not have produced truly independent samples, resulting in pseudoreplication (Hurlbert 1984). The clustering of sites may indicate differences in water quality, habitat, or biodiversity that are results of larger-scale processes such as watershed development, prevailing winds, or currents in the lake rather than direct impacts of shoreline land uses.

In order to control for these effects, each site was assigned a cluster value between one and six based on the geographic location of the site (Figure 3). These clusters were analyzed independently and by land use type in the overall MANOVA.



**Figure 3.** Clustered sites sampled along the littoral zone of Great Pond.



## Results

Colinearity was found 41 times between variables; therefore, a total of 18 variables were included in the MANOVA (Table 1). Results from the MANOVA test for different water quality, habitat, and biotic metrics found a significant difference among land use types (Wilk's lambda= 0.0424,  $F_{54, 102} = 3.57$ ,  $P = 0.0098$ ). However, cluster analyses were not significant (Wilk's lambda=0.5145,  $F_{18,34}=1.78$ ,  $P=0.0717$ ), and there was not a significant interaction between land use and cluster analyses (Wilk's lambda=0.5664,  $F_{18,34}=1.45$ ,  $P=0.1729$ ). The analyses of differences between land use types were investigated through one-way ANOVAs, and are discussed in other sections. After all one-way ANOVAs were run, a rejection value of 0.03 for an  $\alpha$ -level of 0.05 was calculated. Therefore, all p-values from one-way ANOVAs were rejected if they were above 0.03.

**Table 1.** All of the variables tested in the bivariate regression. A total of 20 variables were included in the MANOVA analyses and 18 were excluded because of collinearity.

Variables in MANOVA	Variables Excluded
Temperature	Zooplankton Diversity
Nitrates	% Cyclopidae
Hardness	% Daphnia
Phosphates	% Polyphemidae
Zooplankton Density	Macroinvertebrate Abundance
Zooplankton Richness	Macroinvertebrate Richness
% Chydoridae	% Filterers
% Sididae	% Scrapers
Macro-invertebrate Diversity	% Clingers
FBI	Aquatic Canopy Cover
% EPT	Maximum Depth
% Shoreline Canopy Cover	Aquatic Slope
% Muck	% CWD
% Sand	% SWD
Shore Slope	% Vegetation
% Gravel	% Root wads
% Cobble	Macrophyte Richness
% Rock	Large and Small Woody Debris
% Boulder	Rootwads
% Macrophyte Cover	



## *Discussion*

The significant differences found by the MANOVA indicate that there are differences between land use types. These differences may be due to inherent habitat variations (see section III.D.) or may be due to shoreline development (see section III.B.). If the MANOVA is reflecting natural habitat variations, such as those between wetlands sites with high nutrient loads and abundant macrophytes, then it is likely that shoreline development does not have a significant impact on the health of the lake. However, the significant differences could also be reflecting negative impacts of shoreline development on parts of the lake. The relative importance of natural habitat versus development variations is discussed in greater detail in the following sections.

Because many of CEAT's sampling sites were located near each other, a cluster analysis was necessary to determine if all of the sites were statistically independent. The lack of significant interactions between land use and cluster analyses indicates that all of the sites can be considered to be independent. However, there were some significant cluster interactions within individual variables (e.g., zooplankton), which are discussed in other sections.

## *Water Quality*

### **Introduction to Water Quality**

As development and other human activities continue and pollution accumulates, natural systems and cycles are damaged. One third of Maine's threatened water systems are lakes. Of these lakes, 77% are threatened by non-point source pollution, such as runoff and disturbance of vegetation and soils in the watershed (Amirbaham 2003). Lake water quality can be an indicator of pollution levels in the surrounding ecosystems. Water quality measures the purity of the water source and its ability to sustain life (USGS 2006). The most significant threat facing Maine lakes is phosphorus loading, which leads to eutrophication (VLMP 2010). The health of a watershed can be determined by the relationships between temperature, acidity (pH), dissolved oxygen, chlorophyll, phosphorus and nitrogen (USGS 2006). Measuring and monitoring water quality is vital to maintaining the health of littoral ecosystems. Identifying the sources of water contamination is necessary to understand and combat continued degradation of water quality.

### *General Methods*

CEAT measured water quality parameters to compare the ecological health of the littoral zone at areas of different land use types on Great Pond. In the center of each 10m by 3m sampling area, 1 L of water was collected 1.5m from shoreline and approximately 0.10m below the surface (DW 2009). Each water sample was placed in an ice slurry to slow metabolic processes. Water samples were taken back to Colby College, Waterville, ME, and refrigerated before testing. Soil was also collected 2m from the shoreline at each site.

## Nitrates

### *Introduction*

Nitrates are naturally occurring nitrogen-containing compounds in aquatic and terrestrial ecosystems. They are composed of nitrogen and oxygen bound together to form nitrate ( $\text{NO}_3^-$ ). Nitrates typically are present in low concentrations, below 0.30ppm (WLPR 2010). When nitrate concentrations are above 0.30ppm, phytoplankton, algae, and certain plants can experience rapid growth, which can lead to lake algal blooms and eutrophication (EPA 2010b).

Sources of excessive nitrate contamination include fertilizers, animal waste, fossil fuels, industrial discharge, wastewater, and poorly functioning septic systems (Murphy 2007). Nitrates dissolve more easily in water than other nutrients, and therefore serve as the best indicator for possible pollution because they end up in water bodies more quickly and in greater concentrations (EPA 2010b).

The rate of nitrogen fixation, the process by which free nitrogen ( $\text{N}_2$ ) is converted to ammonia, increases at the air-water interface (Stumm 1985). Ammonia is then fixed by oxygen to form nitrate, which is a usable form of nitrogen available to plants (Krebs 2001). Nitrogen is crucial to many biological processes and therefore expected to be found in our samples. We are primarily concerned with values exceeding typical nitrate values of 0.30ppm, as this may indicate poor water quality and eutrophication (Das *et al.* 2009).

Nitrate levels may be artificially increased through sewage, industrial wastes, storm drainage and common fertilizers (Das *et al.* 2009). Therefore, residential sites (1-

5m buffer, >5m buffer, no buffer) are expected to experience elevated levels of nitrates in comparison to undeveloped sites (undisturbed, wetlands).

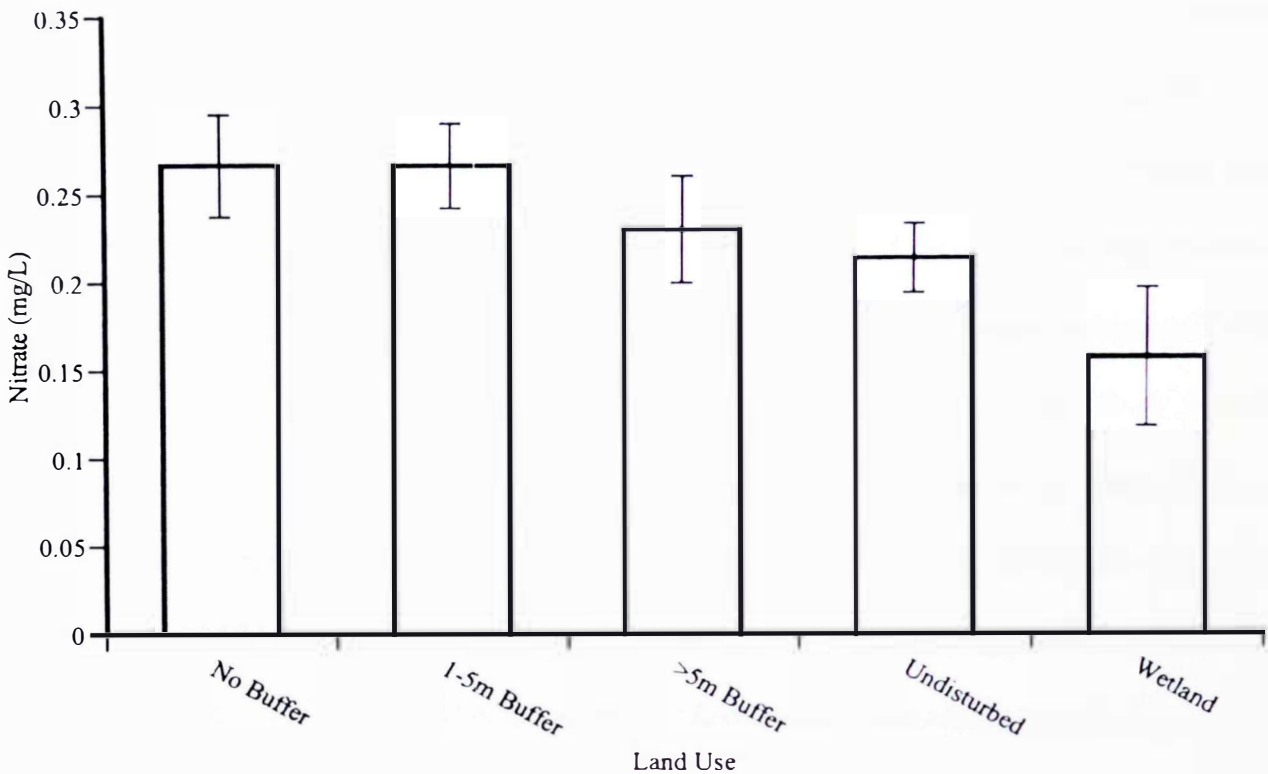
### *Methods*

More reliable results are obtained when samples are analyzed promptly (HACH 2005), so nitrate testing was conducted in the CEAT lab within 24 hours of water collection. The test was conducted using the EPA approved Cadmium Reduction Method 8039. The method uses the HACH Nitrate DR/800 Series, Pocket Colorimeter II, Product 5870002. In this method of nitrate measurement, cadmium metal reduces nitrate in the sample to nitrite. The nitrite ions react in an acidic medium with sulfanilic acid to form a diazonium salt, which couples with gentisic acid. The diazonium salt and gentisic acid compound has an amber color that is then measured by the colorimeter (HACH 2005).

Statistical analyses were conducted to compare nutrient levels between land use types. Descriptive statistics (e.g., means) and one-way ANOVAs were run, followed by Scheffe post-hoc tests if significance was found.

### *Results*

A one-way ANOVA found there was a significant difference between land use types for nitrate values ( $F_{4,64}=6.76$ ,  $p=0.0001$ ). The greatest mean nitrate level,  $0.6\text{ppm} \pm \text{SE } 0.12$ , was found at sites with no buffer ( $n=15$ ). Wetlands had the lowest nitrate level,  $0.16\text{ppm} \pm \text{SE } 0.04$  ( $n=14$ ) (Figure 4). There was a significant difference in nitrate values between no buffer areas and undisturbed areas (Scheffe post hoc test,  $p=0.012$ ). There was also a significant difference between no buffer areas and wetlands (Scheffe post hoc test,  $p=0.005$ ). Additionally, there was a significant difference between 1-5m buffer and no buffer land use types (Scheffe post hoc test,  $p=0.033$ ).

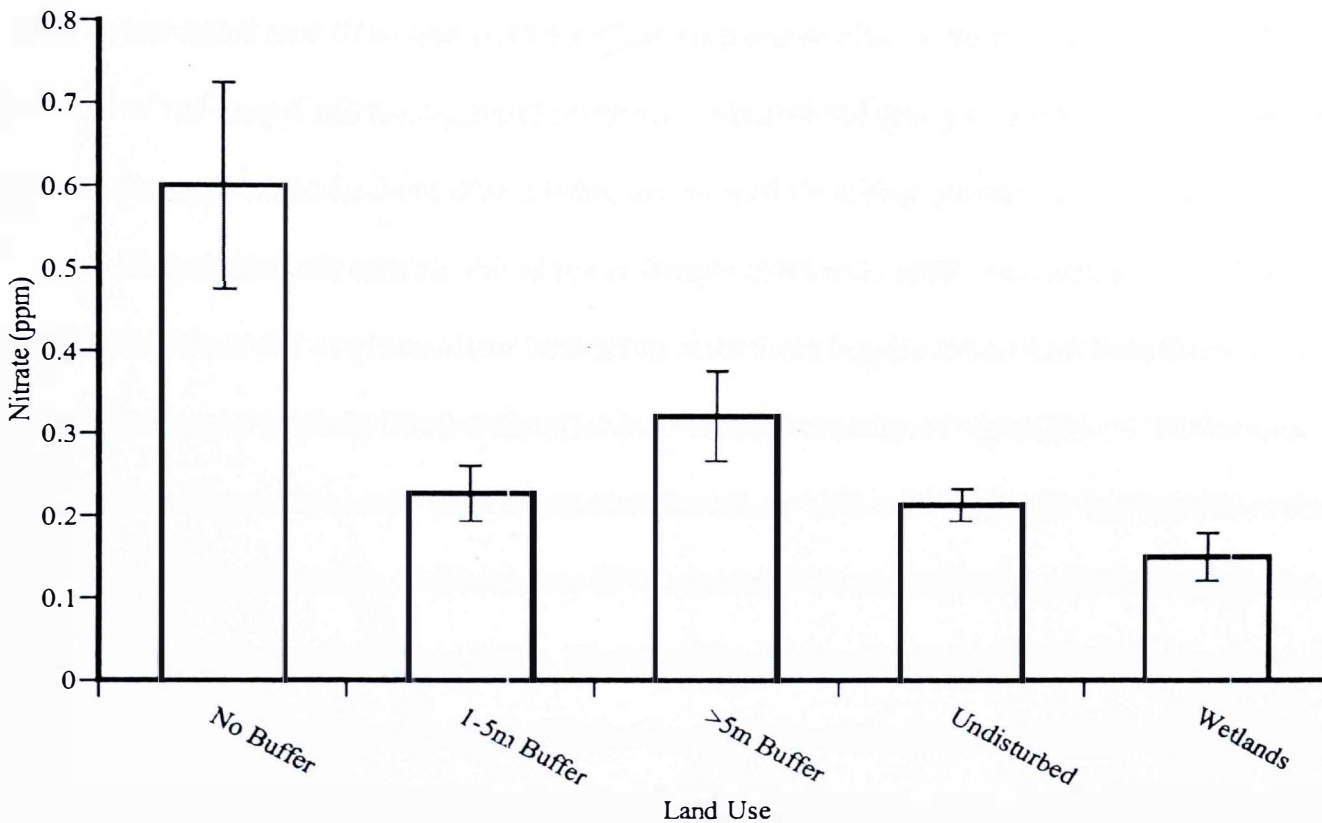


**Figure 4.** Nitrate values (ppm) for water samples taken from all 74 sites taken 1.5m from shore in Great Pond, with respect to land use type.

A comparison of spatial clusters showed that nitrate values for samples collected from sites on 14-October-2010 were generally higher than for samples collected from all other days. Sampling on 14-October-2010 included primarily no buffer and 1-5m buffer land use types.

When data from 14-October-2010 was excluded from analyses, there were no significant differences across land use types ( $F_{4,52}=2.35$ ,  $p=0.0665$ ) (Figure 5). The greatest mean nitrate level was found at no buffer and 1-5m buffer land use types,  $0.27\text{ppm} \pm \text{SE } 0.03$  ( $n=9$ ) and  $0.27\text{ppm} \pm \text{SE } 0.02$  ( $n=9$ ), respectively.





**Figure 5.** Nitrate values (ppm) for water samples taken 1.5m from shore in Great Pond, excluding sites from 14-Oct-10, with respect to land use type.

### *Discussion*

Nitrate analysis, including data from 14-October-2010, supported the prediction that residential sites would have elevated levels of nitrates. As stated in the prediction, the higher mean ( $\pm 1$  SE) nitrate concentrations found in residential sites could be a consequence of runoff from sewage, storm drainage, industrial waste, and fertilizers. The mean ( $\pm 1$  SE) nitrate concentrations from no buffer sites exceed the safe amount of 0.30ppm. Since these sources of contamination are not present in undisturbed sites, it may explain why a significant difference in nitrate concentrations was found between residential and undisturbed sites.

Results changed dramatically when data from 14-October-2010 was removed from the analysis, eliminating any significant difference among land use types. In addition, mean ( $\pm$  1SE) nitrate levels no longer exceed the safe amount of 0.30ppm without data from this date. This change in significance indicates that the differences between residential and undeveloped sites were attributed exclusively to the data from 14-October-2010 sampling. The data collected on 14-October-2010 showed higher nitrate concentrations than the rest of the data collected over the course of this study. It should be noted that the sites examined on 14-October-2010 were spatially clumped. Therefore, these elevated values could be a localized effect. There could be some source of nitrates that affects that region more than others. Since most of the sites sampled on 14-October-2010 were residential, inclusion of this data could skew the results into showing that residential sites have significantly higher nitrate concentrations, when really this may be attributed to a localized effect. This increase in values may also have come from the nitrate testing procedure itself. As analysis was not performed until all data was collected, and sampling for nitrates is sensitive to expiration within 48 hours of water collection. Therefore the anomalous data was not discovered until well after the 48-hour period.

## **Total Dissolved Phosphorus**

### *Introduction*

Phosphorus is a naturally occurring element in soil and rocks, used by plants for growth. Plants use phosphorus in photosynthesis, nutrient transport, and energy transfer, so a lack of phosphorus will result in stunted growth and a decline in fruit production. Aqueous phosphorus occurs as both organic and inorganic phosphates (Murphy 2007). The phosphate ion has the form  $\text{PO}_4^{3-}$ . Organic phosphates are released through decay of

plant or animal tissue. Inorganic phosphates are released into bodies of water through erosion of rocks and other inorganic materials. Eroded sediment is the dominant source of natural phosphate loading for many lakes (Amirbaham 2003).

Phosphates are naturally found at concentrations between 0.01ppm and 0.03ppm (HACH 1983). At these concentrations, phosphates are not harmful to ecosystem health. However, human activity has artificially increased phosphate concentrations creating devastating impacts to some aquatic environments. Levels exceeding 0.03ppm may lead to increased algal growth, and high levels of phosphate may induce an algal bloom and eutrophic conditions (Murphy 2007). The maximum phosphate level to avoid eutrophication is 0.10ppm (HACH 1983). Phosphate enrichment in the environment can be attributed to the use of fertilizers, pesticides, and herbicides. Phosphates are also common components of detergents, sewage, and other industrial wastes. Increased phosphate concentrations in water bodies can lead to algal blooms, causing temporary or long-term eutrophication. Since decomposition of algae requires oxygen, algal blooms reduce dissolved oxygen. This can lead to hypoxic or anoxic conditions, which result in the death of organisms throughout the ecosystem.

Higher total dissolved phosphorus (TDP) levels were expected at no buffer sites than at other land use types. Other developed land use types (1-5m buffer, >5m buffer) may also have elevated phosphorus levels.

### *Methods*

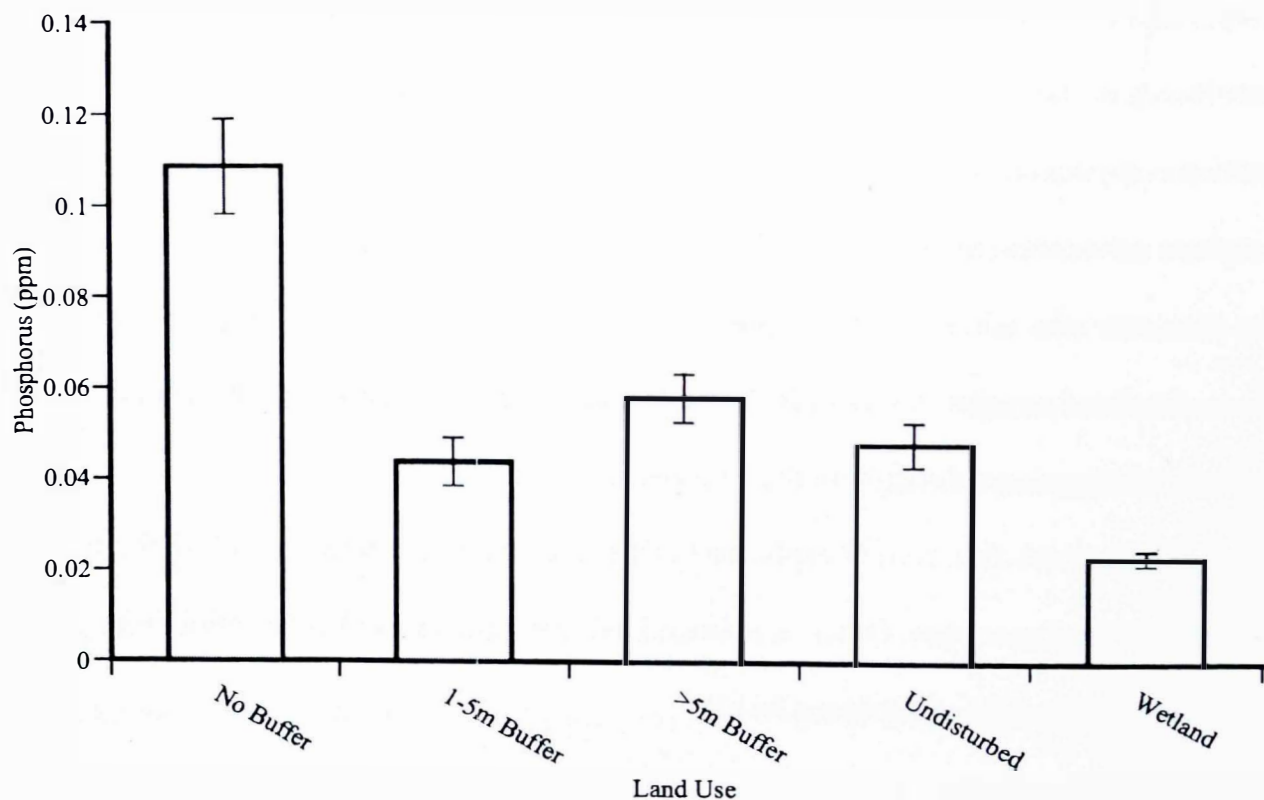
Total dissolved phosphorus testing was conducted within 24 hours of water collection. The test was conducted using Hach Total Phosphate DR/800 Series, Pocket Colorimeter II, Product 587006. This test measures both inorganic and organic forms of

phosphorus.

Statistical analyses were conducted to compare nutrient levels between land use types. Descriptive statistics (e.g., means) and one-way ANOVAs were run, followed by Scheffe post-hoc tests if significance was found.

### Results

The highest mean TDP concentration,  $0.10\text{ppm} \pm \text{SE } 0.01$ , was found at no buffer sites and the lowest mean TDP concentration was found at wetland sites ( $0.02\text{ppm} \pm \text{SE } 0.002$ ; Figure 6). No significant difference existed between land use types for phosphorus values (One-way ANOVA,  $F_{4,67}=0.33$ ,  $p=0.8600$ ).



**Figure 6.** Phosphorus values (ppm) for water samples taken 1.5m from shore in Great Pond, with respect to land use type.

## *Discussion*

Total dissolved phosphorus levels in the littoral zone were much greater than natural levels of 0.01 to 0.03 ppm, and therefore have the potential to induce excess algal growth that may contribute to eutrophication. Great Pond TDP levels should continue to be monitored to observe potential increases that could be cause for concern.

Because there is no significant difference across land use types, this study suggests that there may not be any effects of human development on phosphorus concentrations by sample site.

## **Hardness**

### *Introduction*

In water, hardness is a measure of the concentration of calcium ions ( $\text{Ca}^{2+}$ ) from calcium carbonate ( $\text{CaCO}_3$ ) (Wetzel 2001). The weathering of rocks, particularly limestone, is the major contributor of calcium ions to lakes (Chapman 1992). Water with low hardness, referred to as soft water, has calcium concentrations below 60ppm (USGS 2009).

Low levels of hardness may impact fish health by causing a loss of important nutrients. Fish must expend extra energy to reabsorb lost nutrients (Wurts 1993). Additionally, when calcium ions bind to phosphorus in the water, it renders the phosphorus unusable by plants. Therefore softer water, with fewer calcium ions, has more free phosphorus and is more susceptible to algal blooms (CEAT 2010).

Water in Maine is typically soft (Briggs and Ficke 1977), so CEAT expected to see uniform hardness values between the different land use sites.



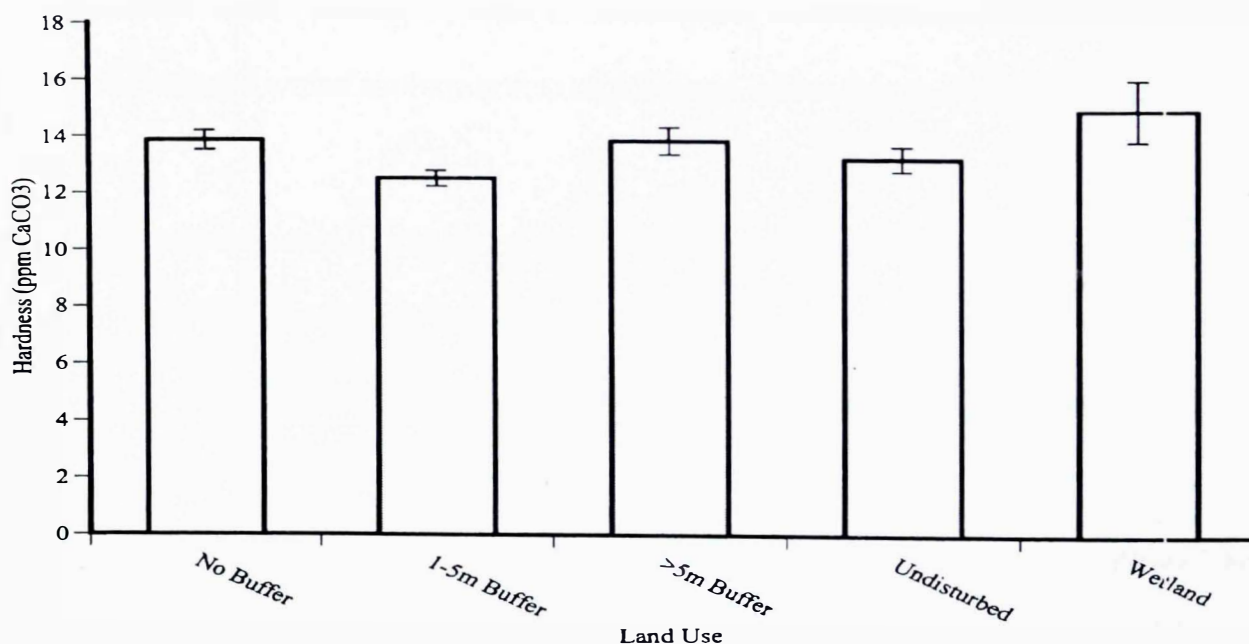
## Methods

Water samples were tested for hardness within two months of collection. The test was conducted using HACH Hardness (Total) Test Kit, Model HA-71A, Drop Count Titration, Dual Range, Product 145201.

Statistical analyses were conducted to compare nutrient levels between land use types. Descriptive statistics (e.g., means) and one-way ANOVAs were run, followed by Scheffe post-hoc tests if significance was found

## Results

No significant differences were found between land use types for hardness values (One-way ANOVA,  $F_{4,63}=2.11$ ,  $p=0.0903$ ; Figure 7). The greatest mean hardness value,  $15.05\text{ppm} \pm \text{SE } 0.98$  ( $n=68$ ), was found in wetland sites. The land use type 1-5m buffer had the lowest mean hardness values,  $12.58\text{ppm} \pm \text{SE } 0.28$  ( $n=68$ ). We found no significant difference among hardness values at different land use types.



**Figure 7.** Hardness values (ppm CaCO<sub>3</sub>) for water samples taken 1.5m from shore in Great Pond, with respect to land use type.

## *Discussion*

Hardness levels can affect the health of fish and influence algae bloom rates in lakes (Wurts 1993; CEAT 2000). Soft water benefits fish growth by improving the absorption of nutrients, but may increase susceptibility to toxins and contaminants (CEAT 1999).

The water in Great Pond is soft, which corresponds to the New England and Northeast Region trend (Briggs and Ficke 1977). This is an advantage to fish growth but can also lead to increased algal growth, which can cause lake eutrophication. Phosphorus tests also indicated susceptibility to increased algal growth; the combination of several factors such as soft water and high phosphorus may accelerate eutrophication.

## **Soil Phosphorous**

### *Introduction*

Sediment is the dominant natural source of phosphate for many lakes (Amirbaham 2003). Phosphorus levels in soil may determine those in adjacent littoral areas. Inorganic phosphates are released into bodies of water through erosion (Murphy 2007). Therefore, areas of high erosion would likely contribute more phosphorus to a lake. Human activity has artificially increased phosphorus levels in the soil from fertilizers, pesticides, herbicides, detergents, sewage, and other industrial waste. Therefore, CEAT expected phosphorus levels to be highest in residential land use types.

### *Methods*

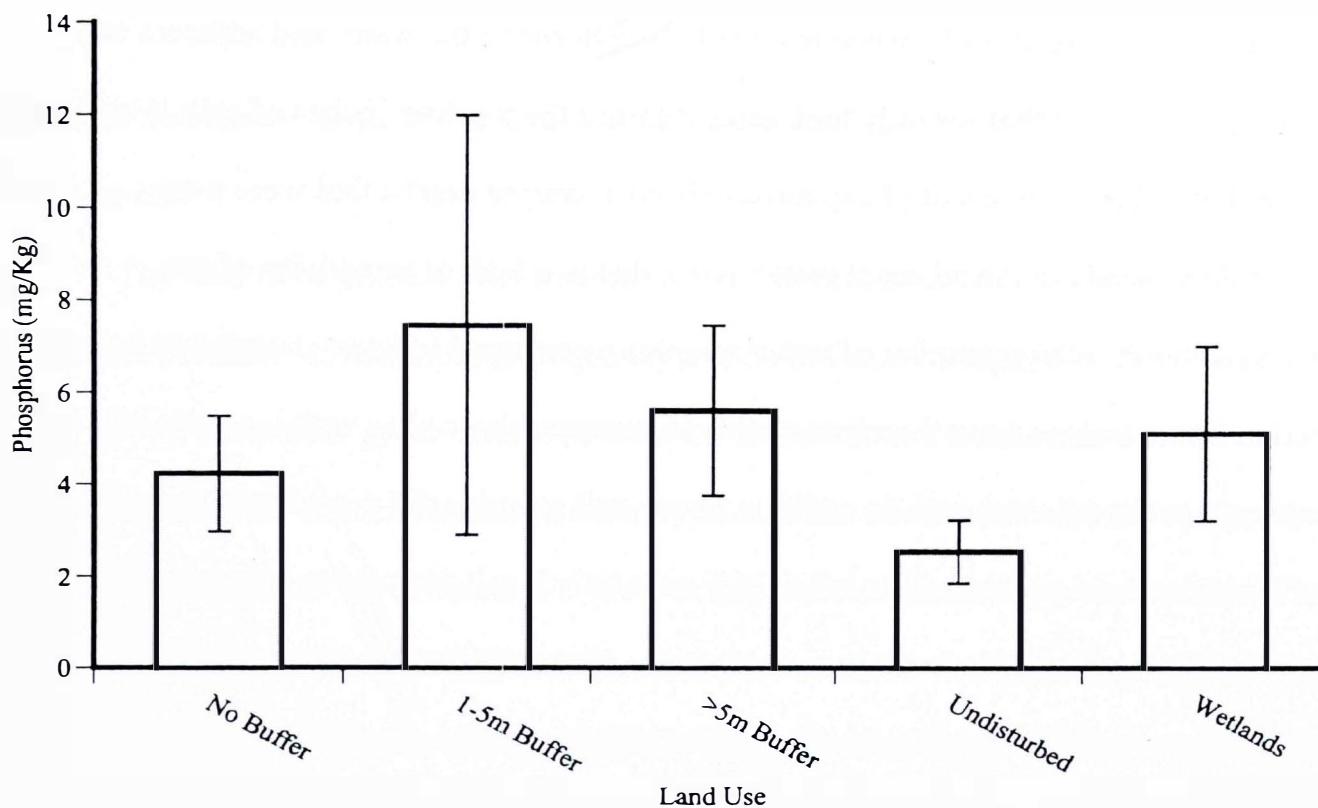
CEAT compared the amount of phosphorus in the soil of the lakeshore with the amount of phosphates in the lake. A terrestrial soil sample was taken between 1 and 3m

from the shore (depending on where the rocky substrate ended) in the middle of the sample site. CEAT collected soil 10cm deep with a soil core sampler. Soil samples were stored in a refrigerator and later dehydrated in an oven at 60° C. The procedure for manual determination of phosphorus described by Ziadi and Tran (2006) was used to analyze samples.

Statistical analyses were conducted to compare nutrient levels between land use types. Descriptive statistics (e.g., means) and one-way ANOVAs were run, followed by Scheffe post-hoc tests if significance was found.

### *Results*

The highest mean soil phosphorus concentration, 7.458ppm  $\pm$  SE 4.553, was found at 1-5m buffer sites. The lowest mean soil phosphorus concentration, 2.54ppm  $\pm$  SE 0.685, was found at undeveloped sites. A one-way ANOVA found no significant difference between land use types for soil phosphorus values ( $F_{4,31}=0.62$ ,  $p=0.6547$ ; Figure 8). There was no correlation between soil phosphorus levels and TDP in water ( $r^2=0.1472$ ,  $p=0.3482$ ).



**Figure 8.** Phosphorus values (mg/Kg) for soil samples taken 1.5m from shore in Great Pond, in five different land use types.

### Discussion

Soil phosphorus levels in the northeastern United States tend to fall between 0.9 and 56.2ppm (Ohno *et al.* 2006), putting amounts of phosphorus in the soil around Great Pond on the lower end of the spectrum compared to other soils in the northeastern United States. We found higher soil phosphorus levels were seen in 1-5m buffer sites, which may be because of the presence of a buffer, which prevents runoff of phosphorus from the soil. In contrast, the smallest amount of soil phosphorus was found at undeveloped sites. The fact that no significant difference was found between land use types may be due to a lack of phosphorus rich agents present in the soil such as fertilizers, herbicides, and pesticides. Landowners may understand that these lawn amendments would have detrimental effects on the pond or simply do not feel the need to use them.

A lack of correlation between levels of phosphorus in the water and adjacent soil may be due to the fact that we only took samples from the top four inches of soil. It is possible that different levels of phosphorus existed at deeper depths that were more comparable to levels in the adjacent water. Also, due to a lack of sensitivity of our analysis methods, a large number of water samples were found to have phosphorus below detection limits and we were therefore unable to compare these sites with amounts of phosphorus in the adjacent soil. In addition, many soil samples did not contain enough mass to render them testable. This left CEAT with only 9 soil samples to compare to water phosphorus data; this small sample size may also contribute to lack of significance. Had we taken deeper samples and had a greater weight of soil to compare to water samples, we may have found different results. Because of these limitations, further testing to compare soil to water phosphorus necessary.



## *Habitat*

### **Littoral Substrate: Lake Floor Grade and Sediment Type**

#### *Introduction*

The lake floor grade, or the steepness of the littoral zone (Kalff 2002), is an important predictor of macrophyte frequency and abundance in large lakes (Duarte and Kalff 1986). Lake floor grade affects how far the littoral zone extends into the lake. It also determines the accumulation and stability of lake floor sediments. Additionally, lake floor grade influences wave action, a critical determinant of macrophyte success (Kalff 2002). Sediment type and the nutrients present in the substrate also influence the success of macrophytes (Kalff 2002; Van der Putten *et al.* 1997; Strakosh 2009).

CEAT expected to find shallower lake floor grades at no buffer and wetland sites. Finer sediments and higher macrophyte cover were also expected at these land use types. Specifically, muck was expected to be present more frequently at wetland sites. Less sand was expected at sites with less development.

#### *Methods*

A 2m staff with 10cm interval markings was used to measure depth. Measurements were taken at distances 1.5m and 3m from the water's edge in the center of the site. Depth measurements were divided by the appropriate horizontal distances to calculate lake floor slope. This was then converted to lake floor grade by multiplying by 100. The two grades for each site were averaged to obtain overall grade.

The aquatic portion of each site was subdivided laterally into four 2.5m x 3m sections. A 0.5m x 1m quadrat was placed within each section to select a sampling subsite. A randomly generated number, 1, 2 or 3, was used to determine distance from

shore (Haahr 2010). Quadrats were tossed into the sections of the site that corresponded with the randomly generated numbers. A “1” indicated that the quadrat was tossed between the shore and 1m away from the shore, while a “2” indicated that the quadrat was placed in the center, 1m from each edge. A “3” indicated that the quadrat was tossed between 2 and 3m away from the shoreline. The researcher arbitrarily determined lateral placement within each of the four subsections of the site. Each quadrat was subdivided into 0.1m by 0.1m squares with a string grid. The dominant sediment types in each quadrat were then identified. The classifications of sediment were muck (organic material), sand, gravel, cobble, rock, and boulder. Percent cover was recorded for each sediment type present. Appendix C displays the particle size of each sediment type (adapted from Mackay *et al.* 2003). Mean percent cover for each sediment type was calculated. Descriptive and comparative statistics were performed.

## **Microhabitats**

### *Introduction*

The microhabitats present may affect the types of organisms found in a given area. Fish are particularly influenced by different types of microhabitat and are found to increase in diversity as macrophyte diversity increases (Dewey *et al.* 1997). Fish prefer areas with a mixture of non-vegetated and vegetated habitats, as this minimizes risk of predation (Dewey *et al.* 1997). Bacteria and algae that grow on macrophytes are a food source for fish. Macrophytes also provide shelter and food for zooplankton (Kalff 2002). Some studies have found that zooplankton density is positively correlated with macrophyte density; others have found that zooplankton prefer non-vegetated areas when fish predation is concentrated in macrophyte beds (Jeppensen *et al.* 1998). These trends

change daily and seasonally (Jeppensen *et al.* 1998). Greater microhabitat variety can lead to greater zooplankton diversity, as zooplankton species richness increases with an increase in macrophyte structure diversity. In some cases, macrophytes suppress phytoplankton productivity, which can alter zooplankton abundance (Declerck *et al.* 2007). Macrophytes and their decomposition influence the substrate that supports benthic organisms (Strakosh 2009). Macrophytes modify their environment by collecting organic matter (Barko and Smart 1983), and their decomposition adds organic material and nutrients to the sediment (Hynes 1960). Their roots aerate the sediment and provide food sources for some benthic organisms (Magnuson *et al.* 2006). The distribution and abundance of benthic macroinvertebrates are used to predict water quality; therefore, available microhabitats must be considered for correlative and causal analysis.

CEAT expected to find less woody debris at sites with development. CEAT also expected to observe higher zooplankton density and diversity at sites with higher macrophyte cover and diversity.

### *.Methods*

CEAT conducted a visual inspection of aquatic microhabitat types present in each site using four categories: coarse woody debris (diameter >5cm, CWD), small woody debris (diameter <5cm, SWD), rootwads, and macrophytes (Strakosh 2009). Percent cover of each microhabitat type was estimated and recorded, using six cover classes adapted from Leonard *et al.* (2008) and Roman *et al.* (2001): less than 1%, 1%–10%, 11%–25%, 26%–50%, 51%–75%, and 76%–100%. Within each quadrat, the percent cover of each macrophyte type was measured using the same six cover classes as those used to classify microhabitats (Leonard *et al.* 2008, Roman *et al.* 2001). All macrophyte

types were identified to the family level. Macrophytes that could not be identified in the field were collected for later analysis, and pictures of the plants were taken. The specimens were refrigerated in plastic bags until analysis, when they were identified using Borman *et al.* (1997).

Mean cover by each microhabitat type was calculated. Macrophyte richness, or the number of families found, was also calculated. One-way ANOVAs and post-hoc Scheffé tests were used to compare differences between land use types, while linear regression was used to determine strength of relationships.

### **Riparian Zone: Shore Grade, Canopy Cover, and Dominant Vegetations**

#### *Introduction*

The transitional area between aquatic and inland terrestrial habitats is known as the riparian zone (Kalff 2002). Although they are distinct habitats, the composition of the riparian zone may impact the littoral zone. Shore grade, canopy cover and vegetation will all influence the littoral area. Low grades tend to store more groundwater than steep grades, which are dominated by surface runoff. Surface runoff can carry large amounts of organic and inorganic particles, as well as nutrients and contaminants (Kalff 2002). Increased nitrogen and phosphorus may result in increased primary production in freshwater lakes, leading to eutrophication (Krebs 2001). CEAT measured shore grade to determine if it was related to abundance and distribution of macrophytes, macroinvertebrates, and zooplankton as a result of varying levels of surface runoff.

Although it is not part of the littoral zone, riparian vegetation may have a variety of impacts on water quality (Krebs 2001). Canopy cover affects the microhabitats present. It may affect the number and distribution of macrophytes found in a given area

(Mackay *et al* 2003). Organic material falling from the canopy may contribute to the macrophyte, CWD, and SWD microhabitats used by macroinvertebrates and fish. Decaying leaf litter can also contribute to accumulation of organic sediment. As canopy cover and littoral shade decrease, water temperature will rise. The increase in available light will also cause primary production by algae and macrophytes to increase. A potential consequence may be increased abundance of fish and macroinvertebrates. However, effects of reduced canopy cover may be mitigated by an increase in turbidity, a measure of suspended particles in the water (Kalff 2002).

CEAT expected to observe a decrease in terrestrial vegetation diversity at sites with greater development. A strong relationship between terrestrial and aquatic canopy cover was expected. Greater canopy cover was expected at sites with buffers and undisturbed sites. CEAT expected to find a relationship between canopy cover and macrophyte presence, though the direction of this relationship was uncertain.

### *Methods*

The slope of the shore was measured using an adaptation of Emery's methods (1961). A 2m staff with 10cm interval markings was placed vertically, directly at the water's edge. A measuring tape and level were used to measure the height of the shore above the water's edge 1m inland and 2m inland, respectively. The two measurements for each site were divided by the appropriate horizontal distance, averaged, and multiplied by 100 to obtain a measure of shore grade. To measure canopy cover, CEAT used concave spherical densimeters (Forestry Suppliers Inc., Model-C). CEAT adapted the procedure specified by the manufacturer for the purposes of this study, so that two measurements were taken: one in the center of the shoreline and one in the center of the aquatic edge of

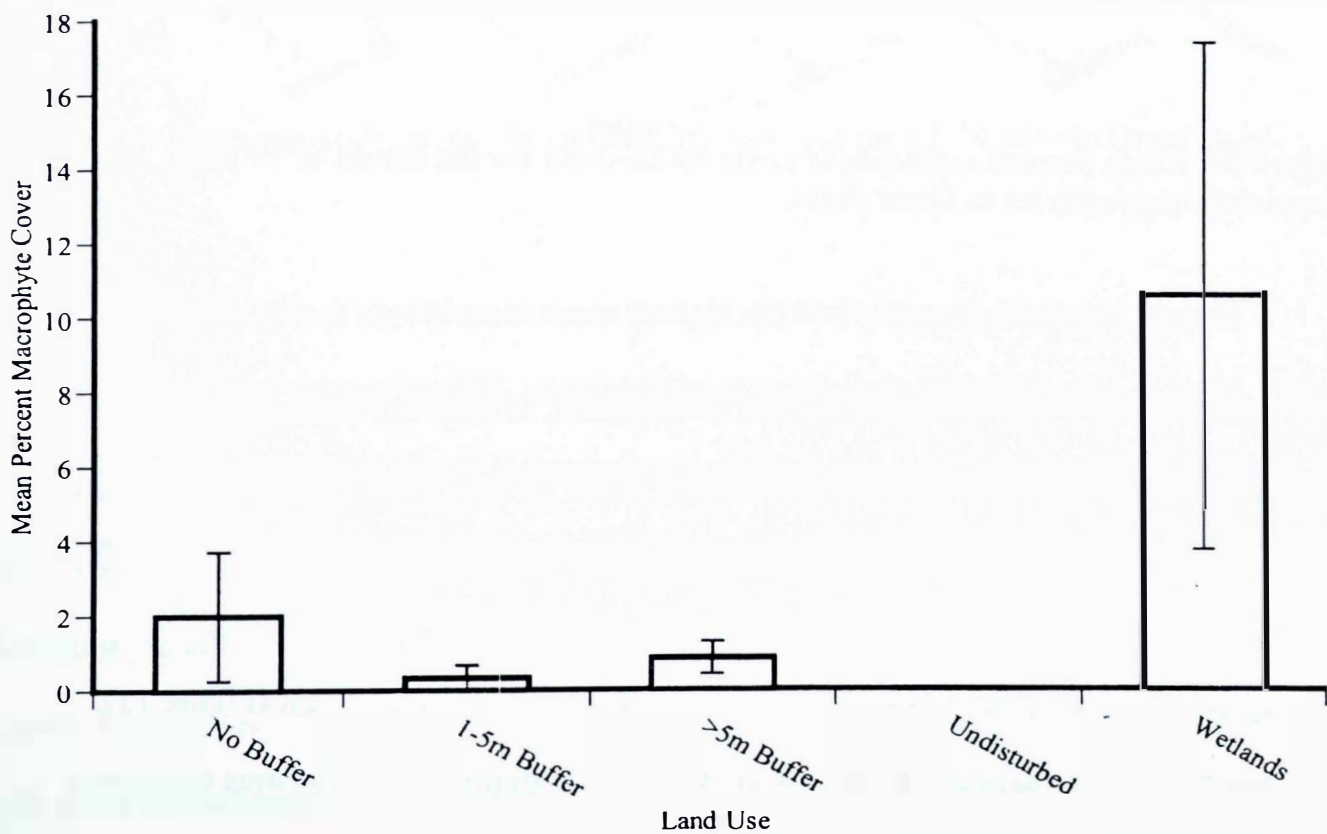


the site 3m away from shore. Researchers faced shore, held the densiometer level at elbow height approximately 30-45cm away from their body, and recorded the number of dots on the gridded mirror not covered by canopy. To calculate the percentage of canopy cover, the number of uncovered dots was subtracted from 96, the total number of dots possible. This number was divided by 96 to obtain the fraction of covered dots, and then multiplied by 100 to convert it into a percentage. The differences in canopy cover between the aquatic and terrestrial edges of the aquatic portion of the site were compared. The dominant vegetation types within the terrestrial portion were also recorded. CEAT used these data to determine whether different types of vegetation were more prevalent in developed areas and to investigate relationships between riparian community composition and littoral biological and chemical parameters. If terrestrial vegetation could not be identified in the field, samples were collected for later identification. These specimens were identified to the family level in the laboratory using several guides (Seiler 2010; Preston and Braham 2002; Samuelson 2006; and Kricher and Morrison 1998). CEAT calculated mean canopy cover and mean terrestrial family richness. Descriptive and comparative statistics were performed.

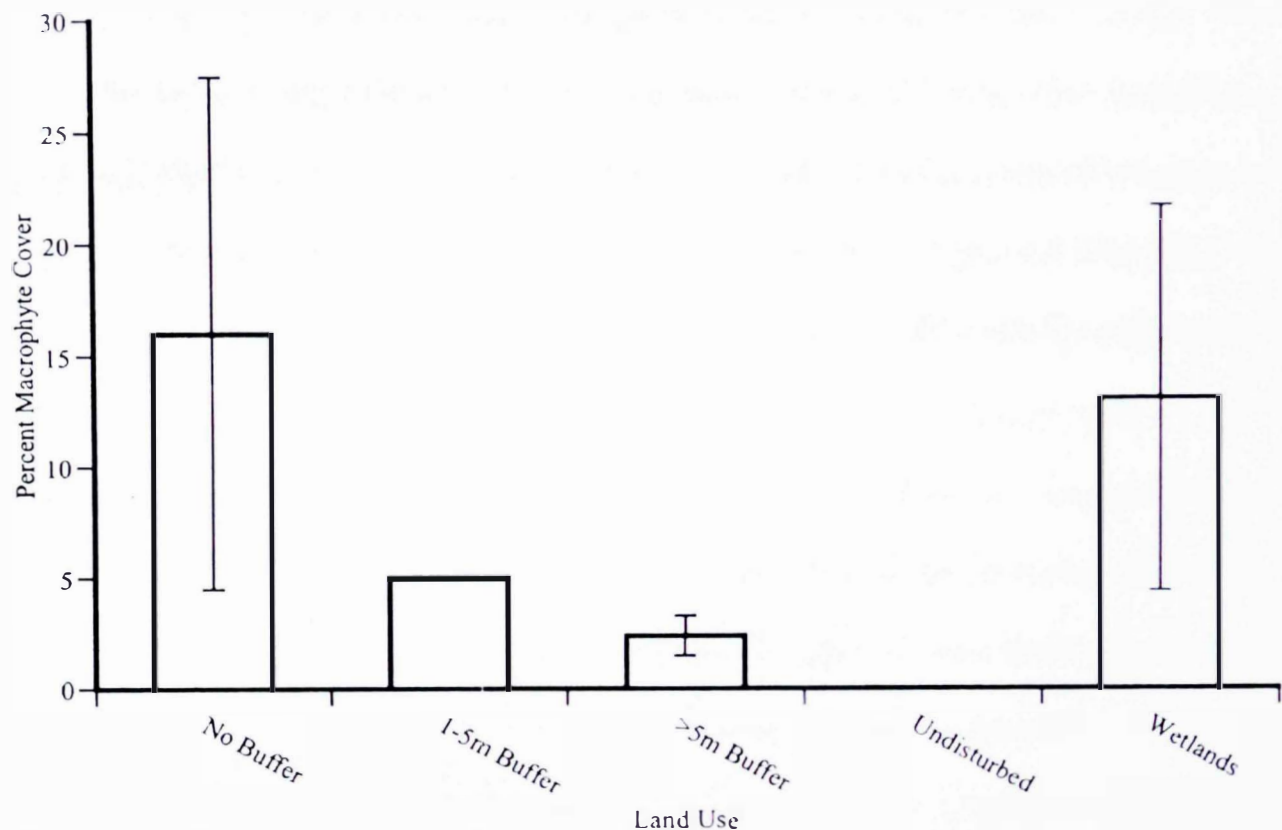
## **Habitat Results**

A one-way ANOVA found no significance between land use types and macrophyte cover ( $F_{4,69}=1.92$ ,  $p=0.1177$ ). Overall, wetland sites had the highest mean macrophyte cover ( $10.6\% \pm \text{SE } 6.84$ ,  $n=14$ ) of all land use categories ( $n=74$ ). Sites with no buffer had the second highest mean macrophyte cover ( $2\% \pm \text{SE } 1.72$ ,  $n=16$ ). No macrophytes were found at undisturbed sites ( $n=15$ ; Figure 9). However, when considering only those sites where macrophytes were found ( $n=19$ ), the general trend

differed. Again, a one-way ANOVA found no significance between land use types ( $F_{3,15}=.29$ ,  $p=0.8339$ ). For this subset of data, the greatest mean macrophyte cover was found at no buffer sites ( $16\% \pm \text{SE } 11.5$ ,  $n=2$ ). Wetland sites had the second highest mean cover ( $13\% \pm \text{SE } 8.63$ ,  $n=14$ ), followed by sites with 1-5m buffers ( $5\% \pm \text{SE } 0.00$ ,  $n=1$ ) and  $>5\text{m}$  buffers ( $2.4\% \pm \text{SE } 0.90$ ,  $n=5$ , Figure 10).

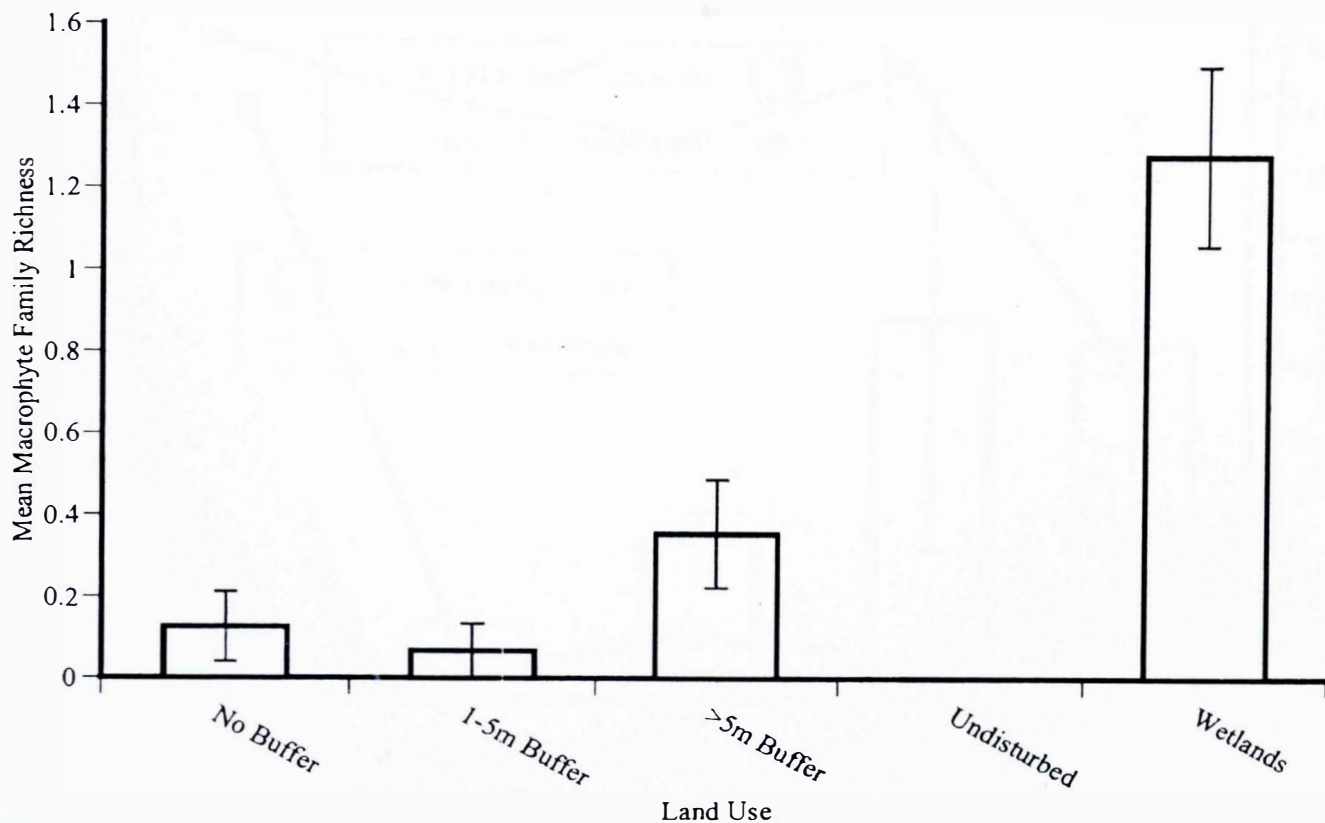


**Figure 9.** Mean percent macrophyte cover by land use type for all 74 sites in Great Pond.



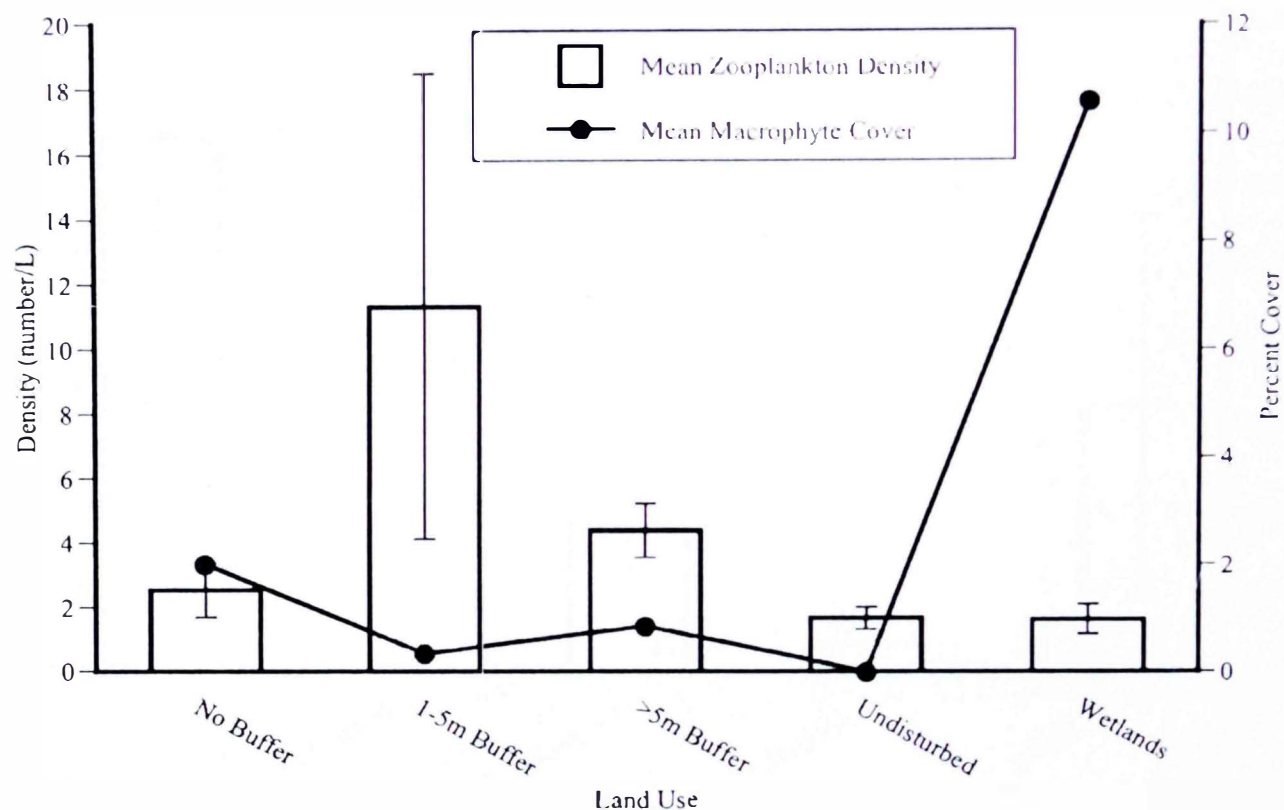
**Figure 10.** Mean percent macrophyte cover by land use for the subset of 19 sites containing macrophytes in Great Pond.

Overall, wetland sites also had the highest mean macrophyte family richness (1.28). The one-way ANOVA found that family richness differed among land use types ( $F_{4,67}=12.85$ ,  $p<0.0001$ ). A post-hoc Scheffe test found that family richness was significantly different between wetlands and all other land use types ( $p<0.001$ , except for between wetlands and  $>5m$  buffer sites, where  $p=0.001$ ). Differences in mean macrophyte family richness between other site types were not significant (Figure 11). Macrophytes tended to occur in shallower sites, although many shallow sites contained no macrophytes. The data showed no linear relationship between depth and macrophyte cover. This is true for the subset of data from sites with macrophytes ( $r^2=0.002$ ,  $df=1,16$   $p=0.9153$ ) as well as from all sites, including those without macrophytes ( $r^2=0.014$ ,  $df=1,71$   $p=0.3188$ ).



**Figure 11.** Mean macrophyte family richness by land use for all 74 sites in Great Pond.

The presence and richness of macrophytes and zooplankton were not related by site or by land use type. There was no linear relationship between mean zooplankton family richness and mean macrophyte family richness ( $r^2=0.0222$ ,  $df=1,72$ ,  $p=0.2049$ ). Macrophyte family richness was lower than zooplankton family richness for all land use types. Additionally, there was no linear relationship between mean zooplankton density and mean macrophyte cover ( $r^2=0.0002$ ,  $df=1,72$ ,  $p=0.9093$ ; Figure 12).

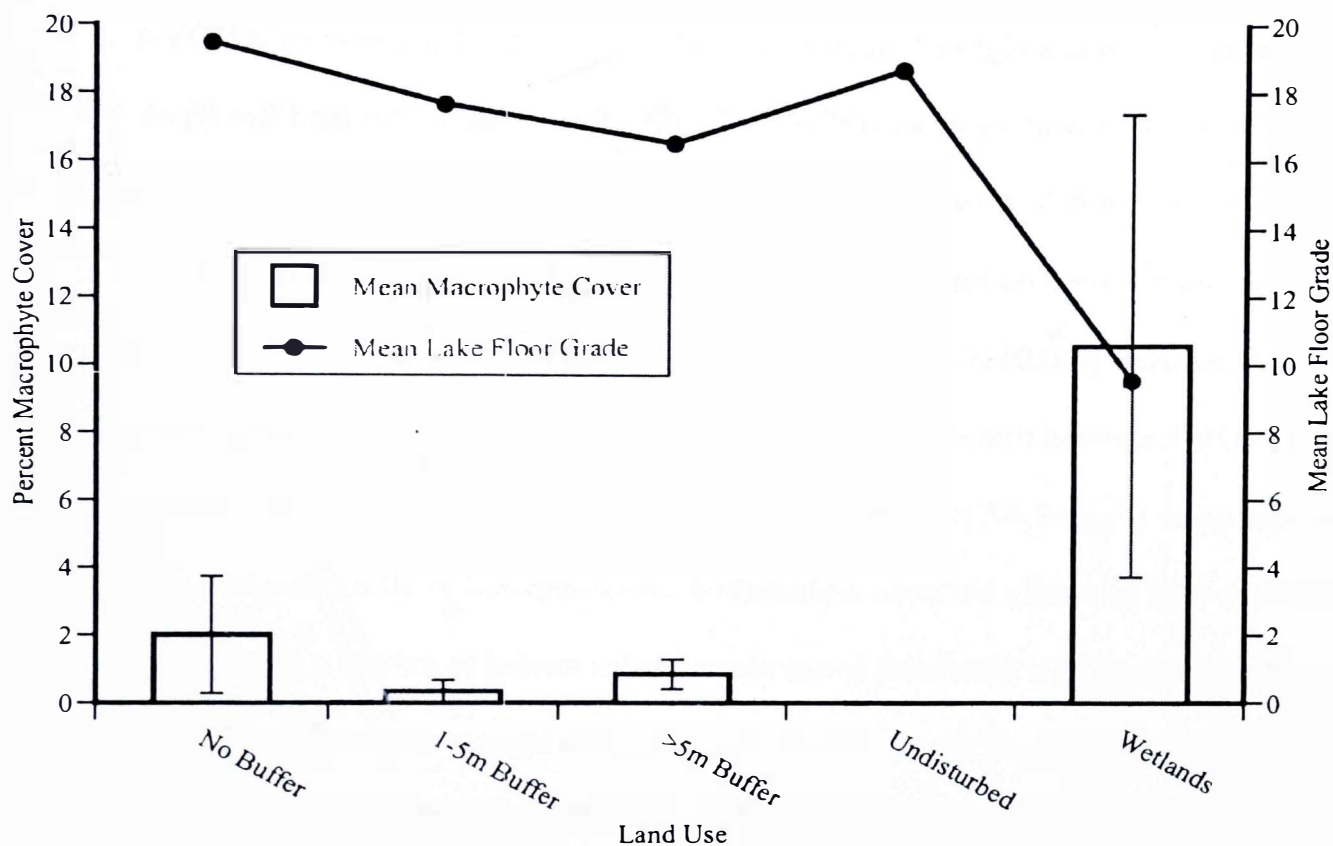


**Figure 12.** Mean zooplankton density in number of zooplankton per liter and mean percent cover of macrophytes by land use type for sites in Great Pond.

The presence of muck was associated with the presence of macrophytes. Out of nineteen sites with macrophytes, muck was present at fifteen sites. Rocky sites had less macrophyte growth. The results from the one-way ANOVA and a post-hoc Scheffe test show that percent cover of muck in wetland sites was significantly greater than the percent cover of muck found in all other sites (One-way ANOVA,  $F_{4,69}=126.63$ ,  $p<0.0001$ ,  $p<0.001$  for Scheffe).

There was some variation in shore grade and lake floor grade across land use categories, with more fine sediment at lower grades. Wetland sites had the lowest mean lake floor grade ( $10.9\% \pm \text{SE } 1.78$ ). The one-way ANOVA results indicated that lake floor grade was not significantly different for the five land use types ( $F_{4,68}=2.82$ ,  $p=0.0370$ ; Figure 13).

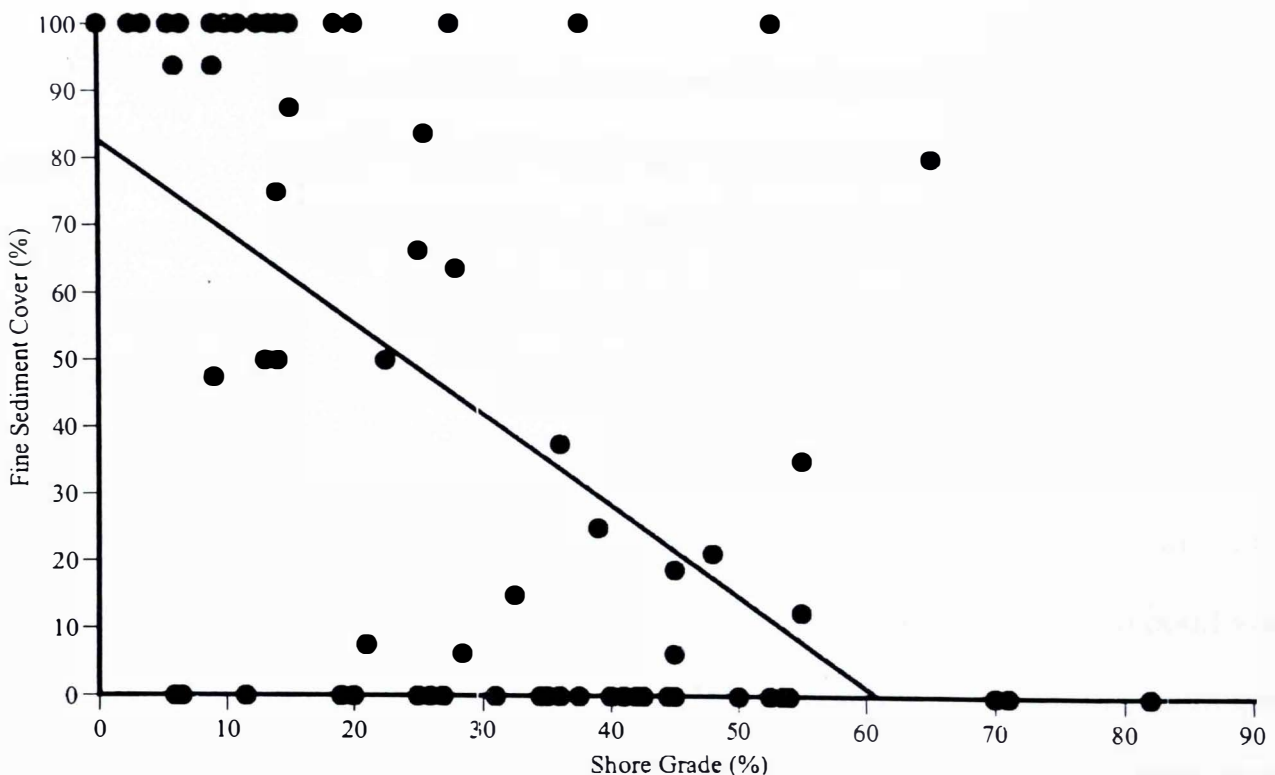




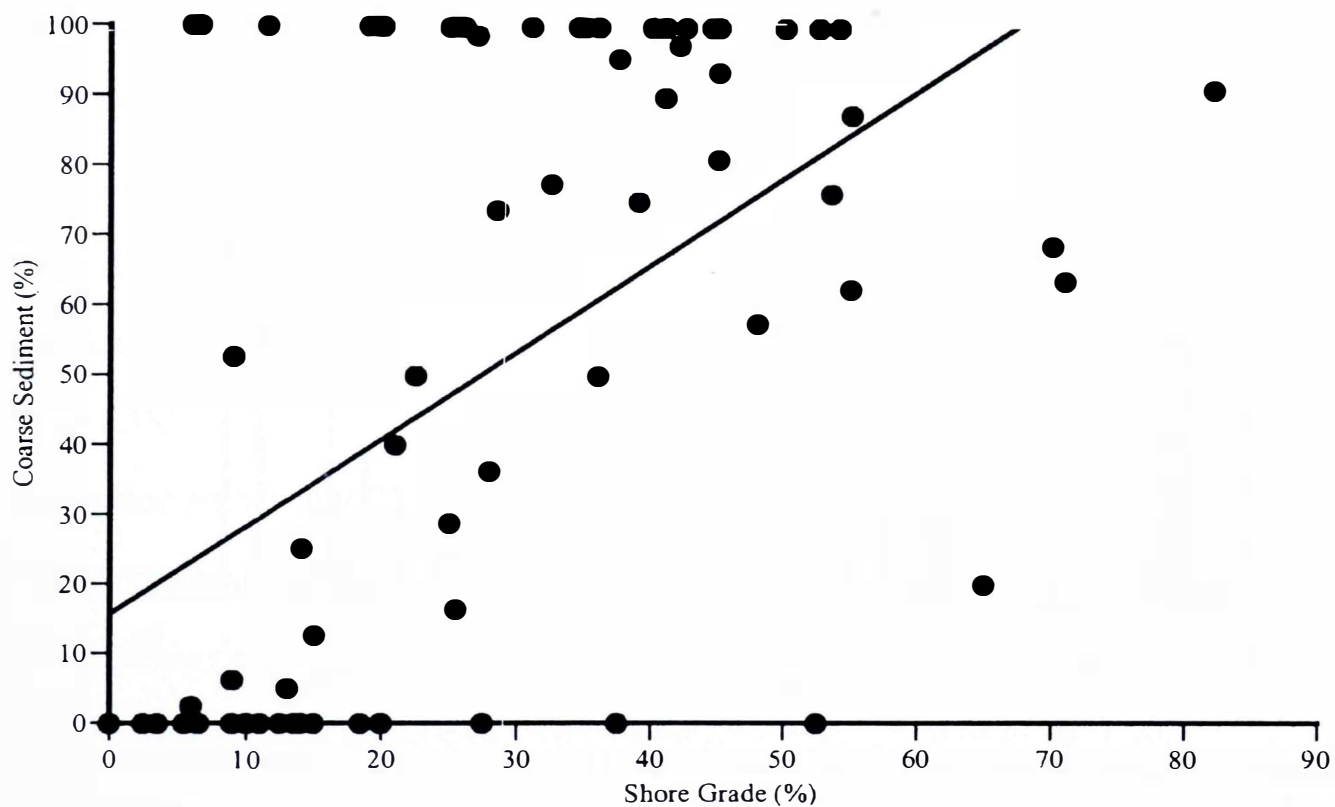
**Figure 13.** Mean macrophyte cover and mean lake floor grade by land use for all 74 sites in Great Pond.

Shore grade varied more than lake floor grade. Mean shore grade was lowest ( $15.6\% \pm \text{SE } 3.63$ ) for wetland sites, and second lowest ( $24.8\% \pm \text{SE } 3.72$ ) at sites with no buffers. Undisturbed sites and those with  $>5\text{m}$  buffers and  $1\text{-}5\text{m}$  buffers had consistently high mean shore grades ( $32\% \pm \text{SE } 4.63$ ,  $48.7\% \pm \text{SE } 12.8$  and  $37.5\% \pm \text{SE } 4.69$  respectively). The one-way ANOVA showed that shore grade varied significantly among land use types ( $F_{4,69}=3.44$ ,  $p=0.0128$ ), and was lower at wetland sites than at sites with  $>5\text{m}$  buffers (Scheffe post hoc test,  $p=0.0150$ ). Sites with low shore grades tended to have littoral zones consisting of high proportions of fine sediments (Figure 14). High proportions of coarse sediments were more often associated with sites that had steep shore grades (Figure 15). As level of disturbance increased, the percentage of finer sediments increased; sites with no buffers had a higher frequency of sand, while

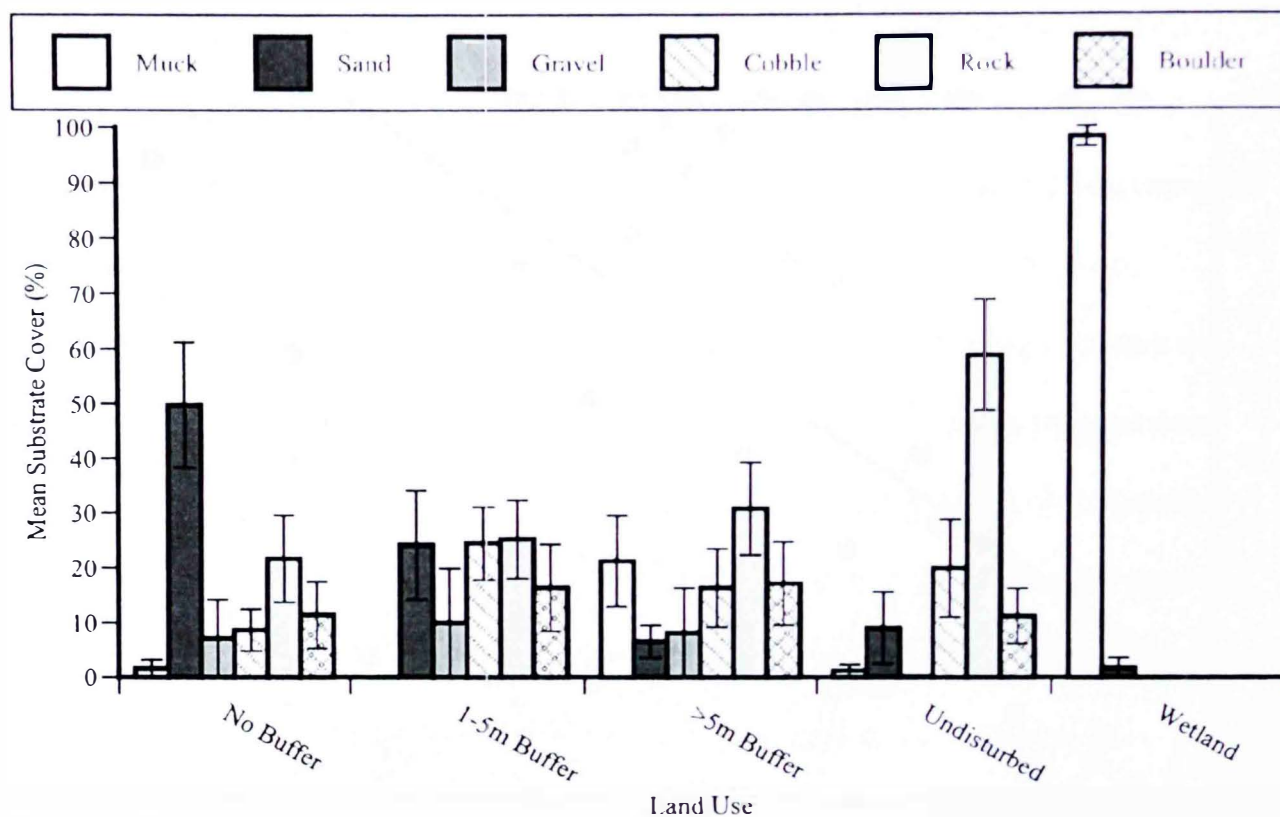
undisturbed sites had a higher frequency of rock (Figure 16). The one-way ANOVA showed that the percentage of sand was significantly different across land use types ( $F_{4,69}=6.92$ ,  $p=0.0002$ ). A subsequent Scheffé post hoc test showed that the percentage of sand differed between no buffer and >5m buffer sites ( $p=0.0050$ ), no buffer and undisturbed sites ( $p=0.0110$ ), and no buffer and wetland sites ( $p=0.0020$ ). Another one-way ANOVA showed that the percentage of rock cover differed significantly between land use types ( $F_{4,69}=7.47$ ,  $p<0.0001$ ); a post hoc Sheffe test showed that this statistical difference was primarily between undisturbed sites compared to all of the other land use categories (Figure 16). Sites with lower shore grades tended to exhibit a greater percent cover of coarse and small woody debris than sites with steeper shore grades. In comparison to the other land uses, wetland sites had the greatest percent cover of CWD, SWD, rootwads, and macrophytes (Figure 17).



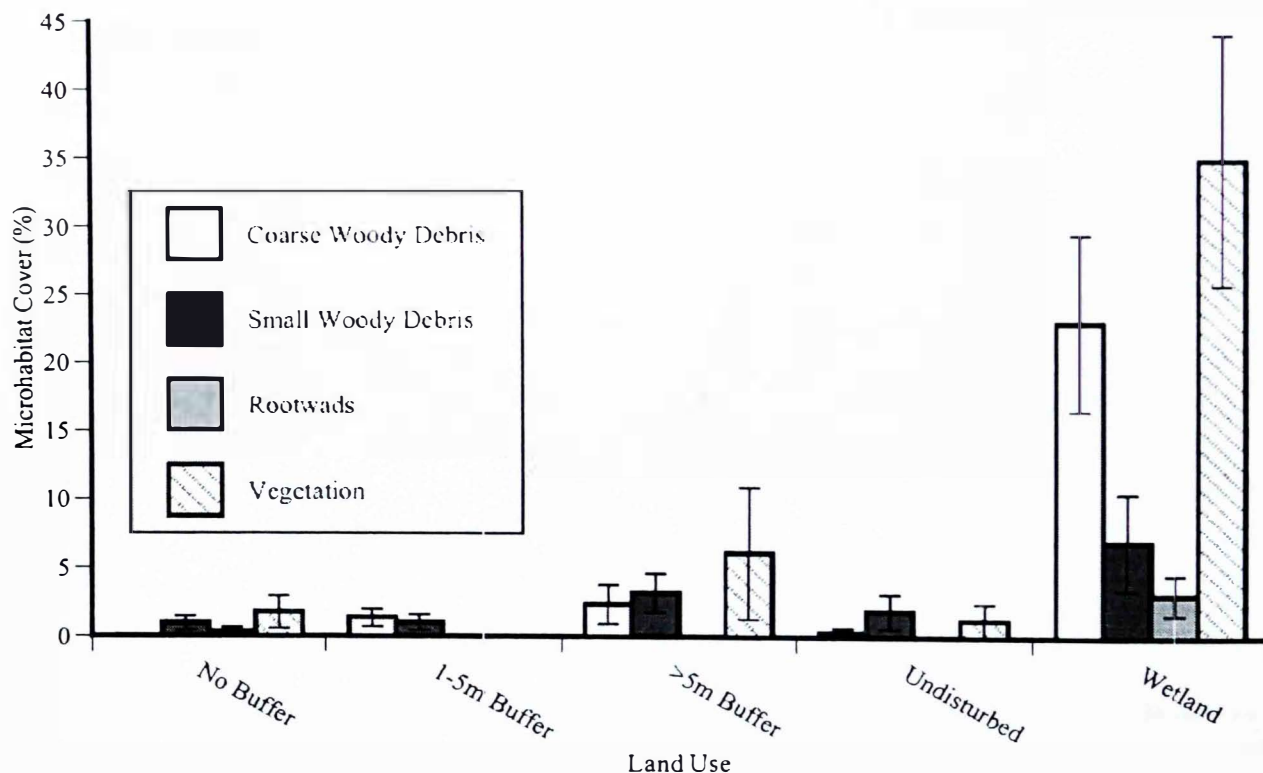
**Figure 14.** Mean cover of fine sediment (muck and sand) vs. shore grade of all 74 sites in Great Pond;  $r^2 = 3.199592E-1$ .



**Figure 15.** Mean cover of coarse sediment (boulder, rock, and cobble) versus shore grade of all 74 sites in Great Pond;  $r^2 = 2.839836E-1$ .

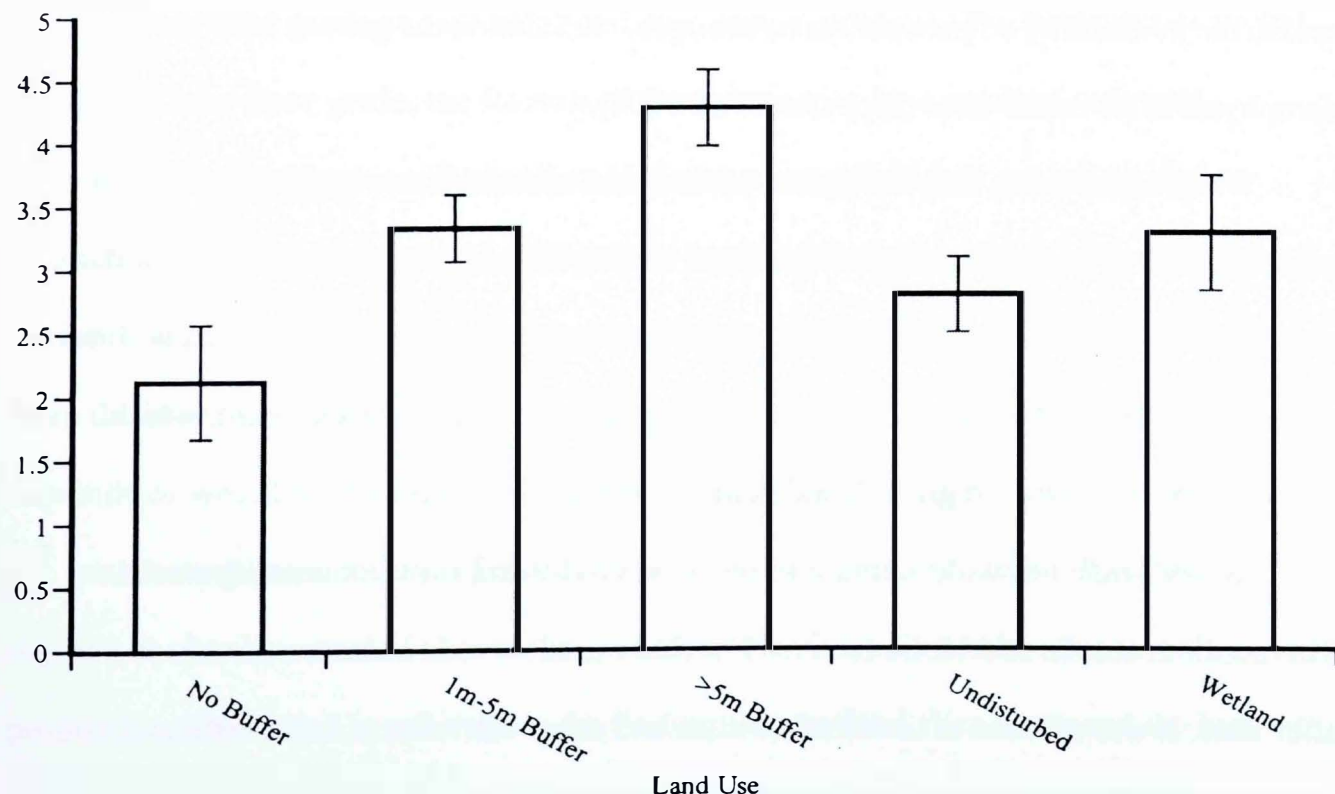


**Figure 16.** Mean percent cover of substrate by land use type for all 74 sites in Great Pond.



**Figure 17.** Mean percent cover of microhabitats by land use type for all 74 sites in Great Pond.

The mean family richness of terrestrial vegetation among land use types was not significantly different according to the one-way ANOVA ( $F_{4,69}=2.16$ ,  $p=0.0832$ ). However, mean terrestrial family richness was highest at >5m buffer sites ( $4.28 \pm \text{SE } 0.30$ ), second highest at 1-5m buffer sites ( $3.28 \pm \text{SE } 0.27$ ) and lowest at sites with no buffer ( $2.12 \pm \text{SE } 0.45$ ) (Figure 18). Mean terrestrial canopy cover was also not significantly different between land use types according to the one-way ANOVA ( $F_{4,69}=1.18$ ,  $p=0.3274$ ). However, the mean terrestrial canopy cover was higher for undisturbed sites than for developed sites (sites with no buffer, 1-5m buffer, and >5m buffer). There was a moderately linear relationship between terrestrial canopy cover and aquatic canopy cover ( $r^2=0.5599$ ,  $df=1,66$ ,  $p=0.0000$ ). Additionally, canopy cover did not appear to have a linear relationship with macrophyte cover ( $r^2=0.0440$ ,  $df=1,72$ ,  $p=0.0727$ ).



**Figure 18.** Mean family richness of riparian vegetation and mean shore slope by land use type for all 74 sites in Great Pond.



## Discussion

Differences in the growth of macrophytes in the littoral zone of Great Pond were found between land use types. Wetland sites exhibited the highest mean cover of macrophytes, followed by no buffer sites. Low shore grades, shallow depths, and small substrate particles characterize these two land use types. These conditions are favorable for macrophyte growth; low grades enable establishment on substrate, fine sediments hold more nutrients (Kalf 2002) and shallow water allows more sunlight to reach the substrate.

A potential explanation for high proportions of macrophyte cover in no buffer sites is that these sites have higher nutrient loading. For example, a shoreline without a buffer may have experienced deforestation, causing bank erosion and deposition of sediments that carry high concentrations of nutrients (Sand-Jensen 1998). This may increase the availability of phosphorus or nitrogen, stimulating the growth of macrophytes in no buffer sites (Jeppesen *et al.* 1998).

When considering only the subset of sites that contained macrophytes, the mean macrophyte cover was greatest at sites with no buffer. This may be an artifact of our data, as only two no buffer sites had macrophyte cover. It is possible that the two sites with no buffer and the wetland sites had an increased amount of nutrient rich sediment, which allow more macrophytes to grow (Sand-Jensen 1998).

Shore grade has wide-ranging implications in littoral zone community structure, involving both abiotic and biotic factors. The shore grade was lowest at wetlands and no buffer sites. It was greatest at undisturbed sites and sites with >5m or 1-5m buffers. Landowners may find it difficult to maintain cleared land on a steep grade, which

supports the finding that sites with greater shore grades tended to be undisturbed or have woody vegetation buffers.

CEAT examined shore grade in relation to sediment type in littoral areas. Sites with low shore grades tended to have relatively higher proportions of fine sediments, whereas sites with high shore grades showed higher proportions of coarse sediments. Sediment size in the littoral zone is influenced by many forces from within and outside of the lake. On the shore, humans may manipulate sediment through excavation, construction, and deforestation. This often leads to greater vulnerability to non-anthropogenic forces, such as wind, water, and ice, which can move and sort sediments. Within the lake, factors such as depth and the direction and strength of currents influence location and size of sediments. Fine sediments are more likely to erode than coarse sediments (Jennings *et al.* 1999).

Lake floor grade can affect littoral community structure. Wetland sites exhibited the lowest lake floor grade; the flatness of these sites may be a result of sediment deposition. Fine sediments often collect in flatter areas where they are not eroded as easily (Kalff 2002). However, sampling only occurred in shallow areas with relatively uniform lake floor grades due to convenience; it would have been impossible to sample from the shoreline in areas with steeper floor grades or deep water. Diving and boating capabilities would have enabled sampling of a more diverse range of floor grades.

Macrophytes tended to grow in areas with low lake floor grades. If sediment deposits were composed of nutrient-laden sediments, this may result in increased primary production. This trend is reflected in the finding that wetland sites contained the highest mean macrophyte cover. However, many of the wetland sites sampled by CEAT were in

close spatial proximity. A potential spatial bias, related to the bathymetry of Great Pond, may explain the similar lake floor profiles of wetland sites.

Decisions made by property-owners may also explain the relationships among land use, shore slope, and sediment size. People may be less likely to build a residence on a steep or rocky site, as it would require more manipulation before construction. Once a residence is constructed, homeowners may manipulate the shoreline to increase accessibility. Landowners may bring sediments to a site where they are not naturally found; for example, a shoreline resident may construct a beach out of sand in an area naturally dominated by rock and cobble substrates. Additionally the removal of a vegetative buffer may lead to an increase in erosion because vegetation and root systems no longer moderate runoff velocity and sediment-carrying capacity (Henderson-Sellers 1987).

CEAT found that sediment type dispersal was related to land use type. Muck was more frequently found in wetlands than in sites of any other land use type. Muck consisted of fine, organic sediments, such as those from the breakdown of vegetation, and is characterized by its texture and dark brown color (Brady and Weil 1996). Many wetlands on Great Pond are located in coves, which are more protected from wind and weather than exposed parts of the lakeshore. Macrophytes also collect this organic matter (Barko and Smart 1983). Fine sediment, which is eroded by wind and wave action, is eventually brought to the rear of coves where wetlands commonly exist, causing a buildup of muck. In this way, wetlands can accumulate nutrient rich sediment and act as a nutrient sink. More macrophytes are able to grow in these protected, sediment-rich areas,

and the decay of these organisms further contributes organic material to the substrate (Kalff 2002).

Muck was strongly correlated with macrophyte growth, shown by its presence in fifteen of the nineteen sites where macrophytes were found. The low depths and nutrients made available from decomposing organisms foster macrophyte growth (Barko and Smart 1983; Jeppesen *et al.* 1998). Macrophytes derive most of their nutrients from the substrate rather than the water column; nutrient rich, organic substrate is related to macrophyte success (Hynes 1960).

Sand was more frequently found at sites with no buffer than at sites with other land use types. There are several possible reasons for this trend. Sand is a fine sediment that erodes easily, especially in sites with no buffer, where there is no riparian vegetation to stabilize the shore. In some cases, landowners bring in sand from outside sources to construct beaches. Deposition of eroded sediments may cover the littoral area (Wetzel 2001), with potential impacts on littoral microhabitats.

Mean macrophyte family richness was calculated to assess the ability of the ecosystem to support a diversity of organisms. Mean macrophyte family richness was higher at wetland sites. This is likely due to the presence of suitable macrophyte habitat in wetlands, where organic substrate and protection from wind contribute to macrophyte survival (Kalff 2002). The higher diversity of wetland organisms provides a strong argument for wetland conservation.

Many of the variables previously discussed also correspond to the presence of woody debris microhabitats. High proportions of CWD and SWD were found in wetland sites. It is possible that CWD and SWD were also present in deeper areas, but this study



focused only on the littoral zone. The findings are consistent with other studies that found lower amounts of woody debris at developed sites (Jennings *et al.* 1999).

Mean terrestrial family richness indicates diversity in woody vegetation. The greatest mean terrestrial family richness was found at >5m buffer and 1-5m buffer sites and the lowest mean terrestrial vegetation family richness was found at sites with no buffer, although these differences are not significant. The lowest mean family richness was found at sites with no buffer, which was expected because woody vegetation has been cleared for lake access at this land use type. Sites with a higher shore grade also had greater mean terrestrial vegetation richness. This may be because some sites with higher grades are more difficult to maintain, resulting in greater woody vegetation growth. However, this may not be a general trend as differences between land use types were not significant.

CEAT did not find any significant differences in mean canopy cover between land use types. Mean terrestrial canopy cover was slightly higher for undisturbed sites, which may be due to larger, more abundant woody vegetation along undeveloped shoreline. CEAT expected to see a strong relationship between terrestrial and aquatic canopy cover. However, the relationship was not as strong as expected. It is likely that one reading in the center of the site was not representative of the site as a whole. The results may also be skewed, because of modified sampling procedure, seasonal changes in foliage, and experimenter bias.

CEAT expected to find one of two relationships between canopy cover and macrophyte presence. One possibility was a negative relationship, resulting from canopy cover shading the littoral zone and therefore reducing macrophyte growth. The other



possibility was that sites with canopy cover would have greater macrophyte growth because of increased organic debris deposition. However, no relationship was found between canopy cover and macrophyte presence in Great Pond. This may be due to the limited accuracy of a single canopy cover reading on the aquatic edge of each site. Sites that actually had little aquatic canopy cover could be falsely reported as having high amounts of cover. Taking readings facing each direction and averaging would have provided a more accurate reading.

Macrophytes can provide a refuge for zooplankton. However, CEAT found no correlations between zooplankton density or richness and macrophyte density or richness. It is unlikely that this is due to day-to-day variation, as all sample collection occurred at a consistent time. Seasonal variation could be at play, as sampling was conducted during fall mixing. Mean zooplankton density was highest in 1-5m buffer sites, which had low macrophyte cover. This suggests that zooplankton may prefer non-vegetated areas, as fish predation may be concentrated in macrophyte beds (Jeppesen *et al.* 1998). However, the lack of macrophyte cover in undisturbed sites was not accompanied by an increase in zooplankton density. Zooplankton density and richness must be more influenced by other factors.

## ***Fauna***

### **Introduction to Fauna**

Biological assessment and monitoring are important parts of evaluating the health of lake ecosystems. Surveying aquatic life can reveal relationships between many different water quality metrics, such as temperature and nutrient levels. This is possible because aquatic organisms reflect the cumulative effects of multiple environmental stressors (Barbour *et al.* 1999; EPA 2002). A further advantage of this type of surveying is that biological communities reflect long-term changes in water quality. Such patterns may be more difficult to detect using direct sampling methods of water chemistry, as water quality metrics indicate relatively short-term trends (Barbour *et al.* 1999). Aquatic communities can also reflect the state of parameters other than water quality, such as degradation of coarse woody debris (CWD) habitats and reduction in leaf litter. All of these components make bioassessment an effective way to assess lake health in relation to lakeshore development.

The current study considers three different biological assemblages in its survey of the fauna of Great Pond: zooplankton, fish, and macroinvertebrates. By examining each assemblage type, various parameters relating to lake health may be observed. For example, fish populations are useful to measure because they represent a variety of levels within the food chain and have relatively long life spans. This makes fish an informative component in a broad, long-term assessment of lake health. In contrast, certain sensitive life stages of macroinvertebrates will respond quickly to stressors). In particular, larval stages have low tolerance to stressors (Barbour *et al.* 1999). Zooplankton are also valuable

to lake assessment because zooplankton assemblages may be a good indicator of nutrient loading and the potential for algal blooms (Buskey 1997).

## **Zooplankton**

### *Introduction*

Zooplankton are linked to multiple factors that are indicative of the health of lake ecosystems (Barbiero *et al.* 2009). The mechanisms that drive zooplankton community structure include nutrient levels, phytoplankton abundance and composition, and predation pressure by fish and macroinvertebrates (Stemberger *et al.* 2001). For example, one study found that zooplankton in the order Cladocera, known as water fleas, and copepods in the order Cyclopoida were more abundant in eutrophic lakes. In oligotrophic lakes, copepods in the order Calanoida tended to be dominant (Gannon and Stemberger 1978). Furthermore, greater densities of zooplankton are seen in eutrophic lakes (Barbiero *et al.* 2009). A study of southern Wisconsin lakes found a correlation between zooplankton community richness and land use, showing that agricultural sites without a buffer had significantly lower zooplankton diversity than sites with buffer strips over 30m. Additionally, the presence of macrophytes was associated with increased zooplankton diversity (Dodson *et al.* 2005).

The relative abundance of large and small zooplankton families can indicate to what degree a lake is influenced by bottom-up or top-down interactions (Stemberger *et al.* 2001). In top-down interactions, organisms at the top of the food web control interspecific interactions. One example of top-down control is the trophic cascade, in which the abundance of organisms in one trophic level suppresses the abundance of the trophic level below, allowing the next lower trophic level to flourish (Stephen *et al.*

1998). In bottom-up interactions, changes in nutrient concentrations can result in increases in primary producers, affecting interactions of organisms higher in the food web. For instance, if the lake is run by bottom-up control, influxes in nutrient levels could alter concentrations of phytoplankton and algae and, consequently, densities of zooplankton that respond to the abundance or scarcity of their food source. An example of top down controls was studied by Buskey *et al.* (1997), who found in a Texas estuary that a crash in zooplankton grazer populations coincided with initiation of a long-term harmful algae bloom, referred to as a “brown tide”. Likewise, Lampert *et al.* (1986) implicated zooplankton grazing as the agent causing a “clear water period” of minimal chlorophyll and reduced algal biomass in German lakes. Larger zooplankton are considered to be better competitors for phytoplankton than the smaller zooplankton, making these larger individuals more important in top-down controls (Arnott and Vanni 1993; Xu *et al.* 2001). For example, Daphniidae are considered to have a superior exploitative ability than small Cladocera. The size efficiency hypothesis, which states that larger zooplankton are more efficient consumers, and Daphniidae’s resilience to starvation account for this competitive advantage (Gilbert 1988). Fish often selectively graze on larger zooplankton, sometimes resulting in a trophic cascade that can allow phytoplankton to flourish. The role of predation can be quantified by comparing the presence of larger zooplankton to the presence of smaller zooplankton (Galbraith 1967). Families including small zooplankton are Bosminidae, Chydoridae, and Centropagidae (Sommer *et al.* 1986), while most members of the family Daphniidae are characterized by large (Stemberger and Lazorchak 1994; Wissel *et al.* 2003).

It is possible that zooplankton densities and community structure can be influenced by factors other than lake health. Due to the seasonal nature of zooplankton life cycles, some families are naturally more abundant at certain times of the year (Allan 1976). Since this study was restricted to September and October, it is possible that species that reach a peak in early summer were not as abundant. For example, Daphniidae bloom in June in New England lakes (Allan 1976). The physical location of the sampling can also affect the composition of zooplankton. Smaller Cladocera and Cyclopoida copepods tend to be dominant in littoral regions, whereas Calanoid copepods tend to be dominant in the center of the lake (Jakobsen and Johnsen 1987). Wind can also effect the distribution of zooplankton. A study on Loch Ness in Scotland demonstrated that the gradient of plankton along the length of the lake was strongly influenced by recent wind history and readily changed with changes in wind direction (Jones *et al.* 1995). Another study on a lake in Michigan also found that wind influenced the distribution of zooplankton (Folt *et al.* 1993). Thus, it is possible that one side of the lake may have greater densities of zooplankton than other areas.

CEAT tested the hypotheses that residential sites would exhibit greater densities and lower diversity levels of zooplankton than undisturbed and wetland sites. Residential sites with wider buffers were expected to have a greater diversity and lower densities of zooplankton than residential sites with smaller buffers. CEAT expected that there would be a greater abundance of small Cladocera than large Cladocera and a greater abundance of Cyclopoid copepods than other copepods, due to the close proximity to the shoreline. CEAT also predicted that zooplankton community structure is controlled by a



combination of top-down and bottom-up processes through fish predation and nutrient cycling.

### *Methods*

Two separate zooplankton tows were taken parallel to shore 1.5m and 3m from the shoreline. Each tow zone was 10m long. The tow was taken with an 83-micron cone plankton net and was pulled in by hand. Samples were initially kept on ice before being stored in a refrigerator and then processed. Samples taken on 27-September-2010 were stored in a freezer. Beginning on the 30-September-2010, samples were preserved in the field with ethanol, in addition to being kept on ice.

In the laboratory, the two tows were combined using a 90-micron sieve and then rinsed into a labeled storage bottle with 50ml of 70% ethanol solution. Due to the prevalence of extraneous material in some samples, mostly organic debris from wetlands, these combined samples were diluted to a 100ml solution of 70% ethanol. PhloxineB dye was also added to these samples to enhance the visibility of the zooplankton from the debris, by staining the zooplankton bright pink.

Three Sedgwick-Rafter slides containing 1ml of the diluted sample were examined from each bottle under 5X magnification on a compact-light microscope. All zooplankton on each slide were identified to the family and the number of individuals was tallied. The data from the three slides were averaged. This data was then converted to density per liter of ambient water to represent the density of zooplankton in the lake. Since two net sizes, one 12cm in diameter and one 20cm in diameter, were used, the conversion to density per liter accounted for differences caused by net size. Diversity was calculated by the Shannon-Wiener Index, which is formulated as:

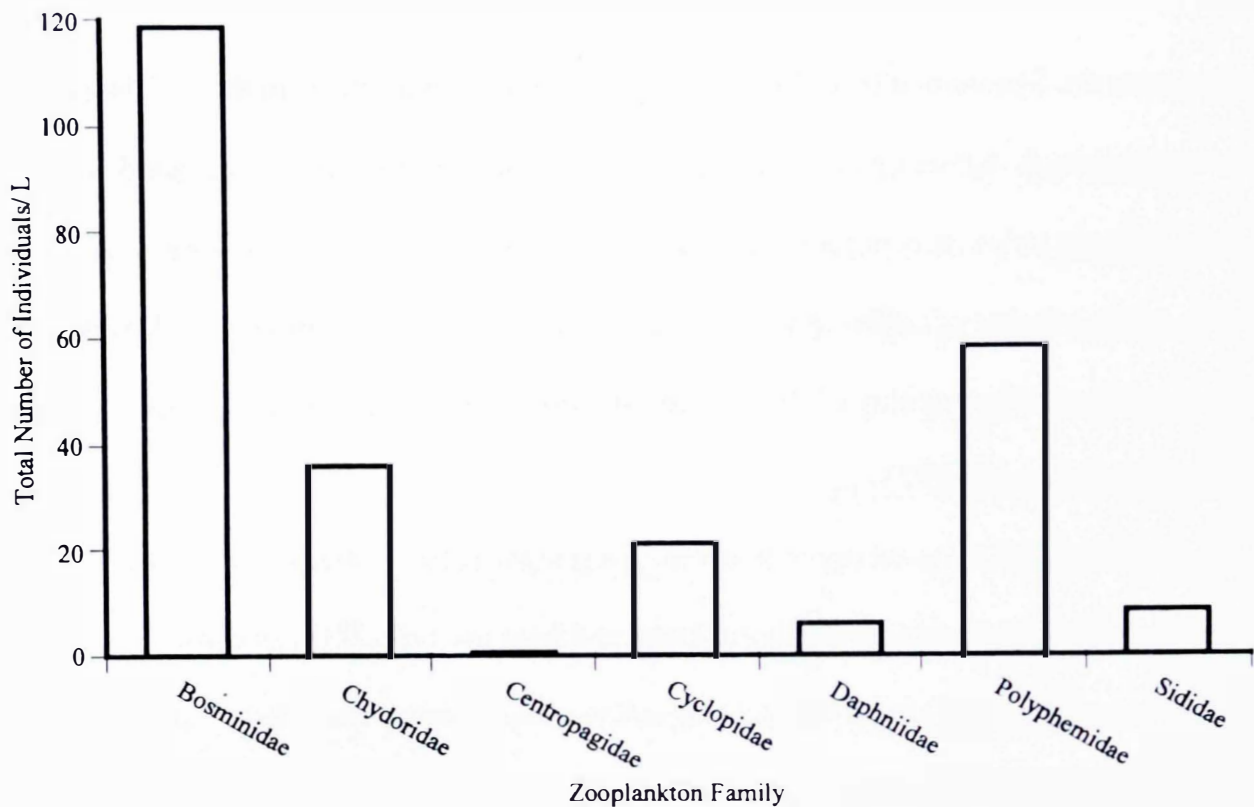
$$H' = - \sum_{i=1}^S (p_i \ln p_i)$$

H' represents the Shannon-Wiener Index. The proportion of individuals in the "i<sup>th</sup>" taxon of the community is denoted by  $p_i$ .  $S$  denotes the total number of species represented. The Shannon-Wiener Index incorporates both species richness and evenness. Richness is a measure of the number of different taxa of organisms present in a particular area, while evenness compares the equality of the number of individuals of each of the species present (Kwiatkowski 1973).

One-way ANOVAs were run to determine if significant relationships existed between zooplankton densities, family richness, and land use type. If the one-way ANOVA was significant, a post-hoc Scheffe test was run to determine where significant differences were (STATA 2009).

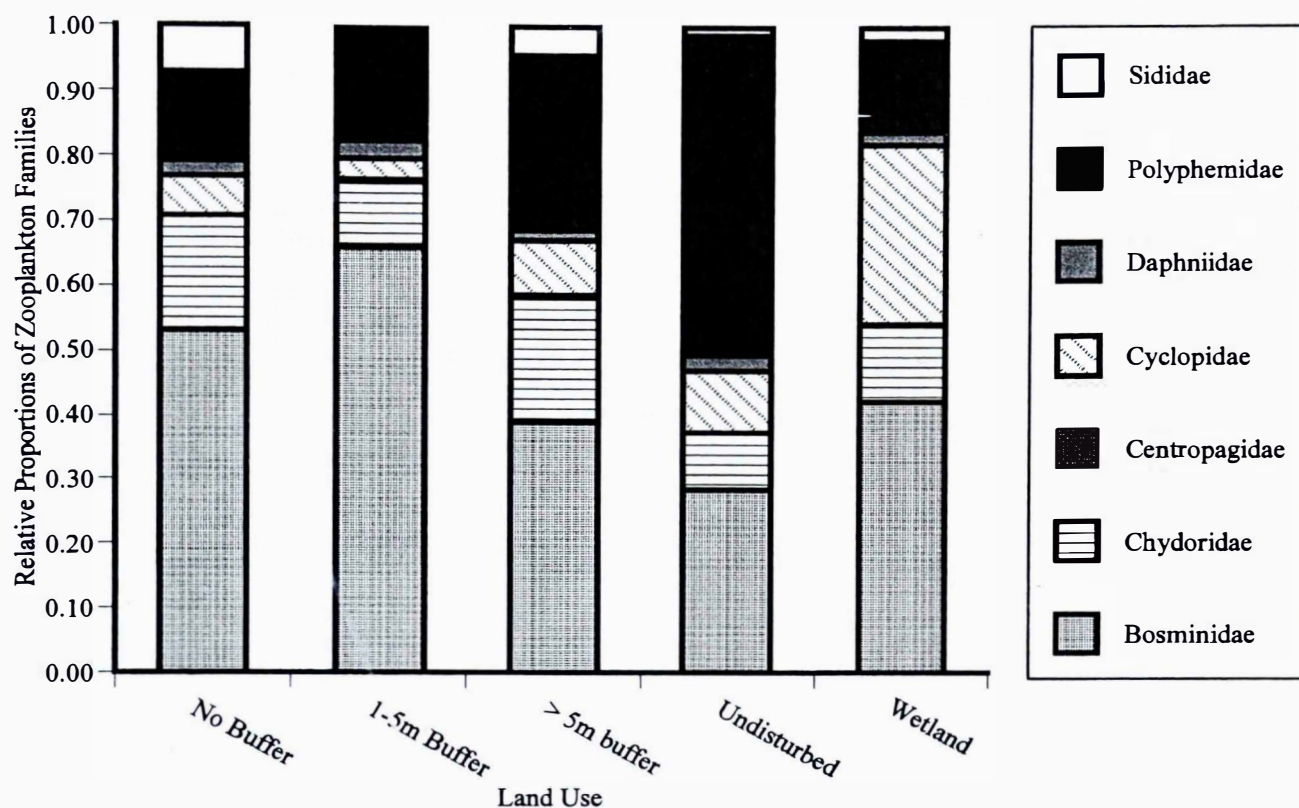
### *Results*

CEAT found ten families of zooplankton in this study. Bosminidae was the most abundant family followed by Polyphemidae and then Chydoridae (Figure 19). The least abundant family in which more than one specimen was observed was Centropagidae. Only one specimen each of Holopedidae, Lepotoridae, and Monidae was observed over the course of this study.



**Figure 19.** Total abundance within each represented zooplankton family.

There were no significant differences between the densities of each zooplankton family at each land use type (One-way ANOVAs,  $p > 0.03$ ). Bosiminidae was dominant at every land use type except for at undisturbed sites, where Polyphemidae was the most abundant family (Figure 20).



**Figure 20.** The relative proportions of zooplankton families by land use type for all 74 sites in Great Pond.

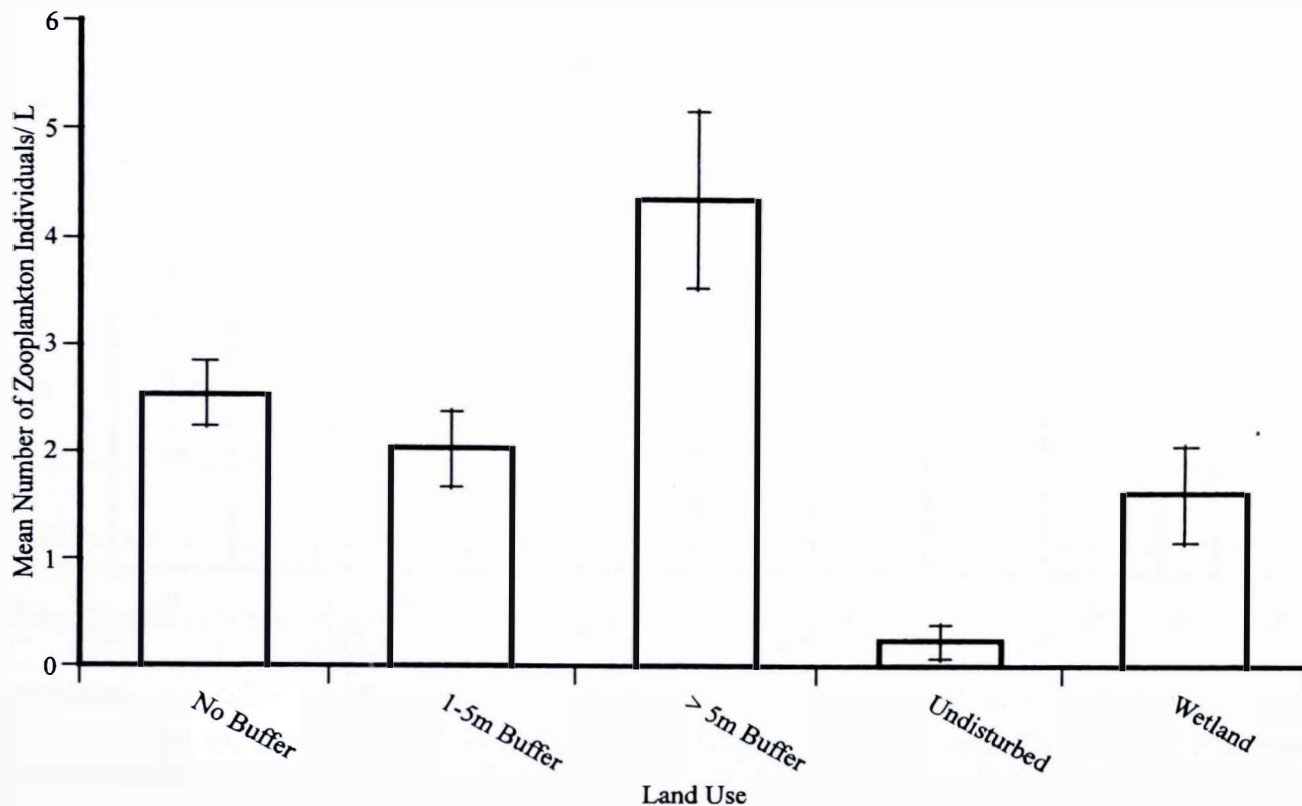
A one-way ANOVA test was run to determine if clustering of sampling sites resulted in spatial bias. Significant differences were found in zooplankton density between spatial clusters only for sites with no buffer (One-way ANOVA,  $F_{4,11}=7.02$ ,  $p=0.0046$ ). No significant differences in zooplankton density were observed between sites sampled on different dates (Table 2).

**Table 2.** Summary of p-values from one-way ANOVAs (STATA 2009) analyzing zooplankton data by date and by cluster across land use types.

Land Use	By Date	By Cluster
No Buffer	0.9399	0.0046
1-5m Buffer	0.9704	0.606
>5m Buffer	0.4289	0.1602
Undisturbed	0.722	0.4857
Wetlands	0.9826	0.0946

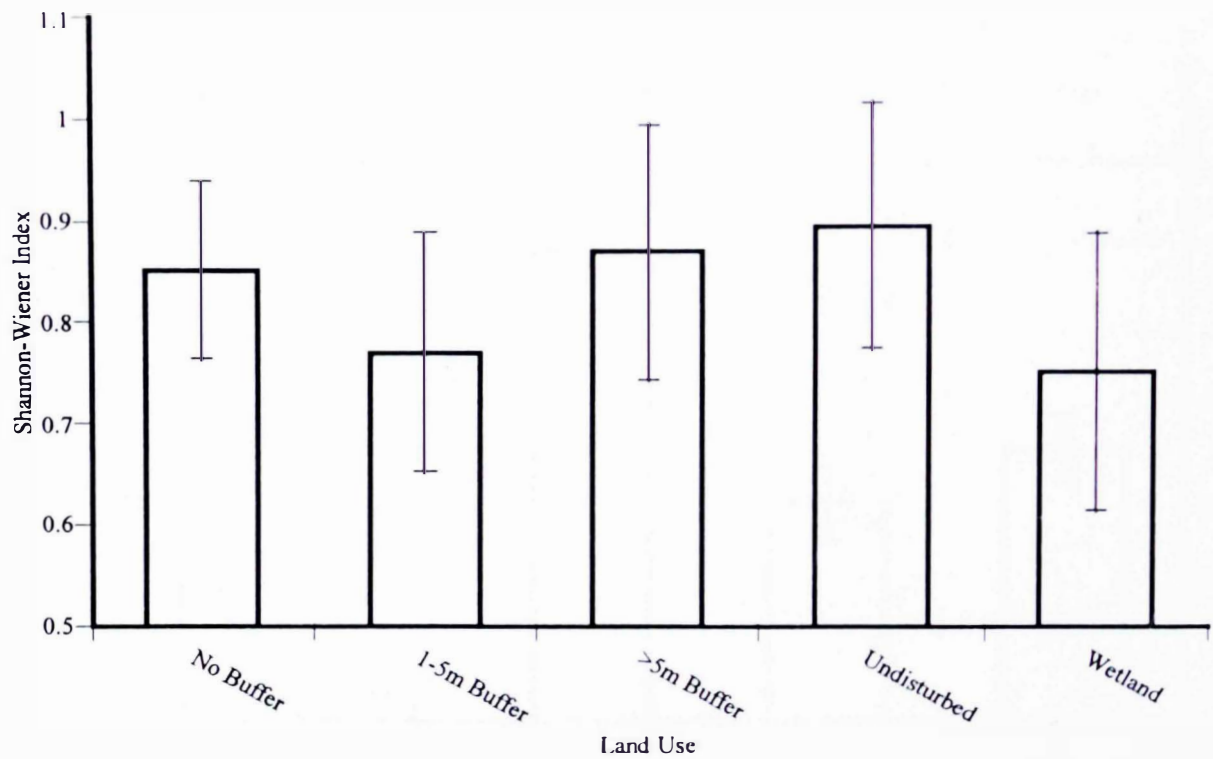
The total density of zooplankton per liter was calculated for each site. There were no significant differences in density between the different land use types (One-way ANOVA,  $F_{4,69}=1.60$ ,  $p=0.1849$ ). Sites with 1-5m buffers had the greatest average ( $\pm 1$ SE) density of zooplankton with  $4.46 \pm \text{SE } 1.39$  individuals per liter, while wetlands exhibited the lowest density with  $1.62 \pm \text{SE } 0.45$  individuals per liter (Figure 21). One 1-5m buffer site was excluded from this analysis because it was an outlier, with a density of 110 individuals per liter, compared to the mean of 3.34 individuals per liter in all other sites. Since the value of 110 is more than eight standard deviations away from the mean, by Chauvenet's criteria it is an outlier (Irwin 1925).



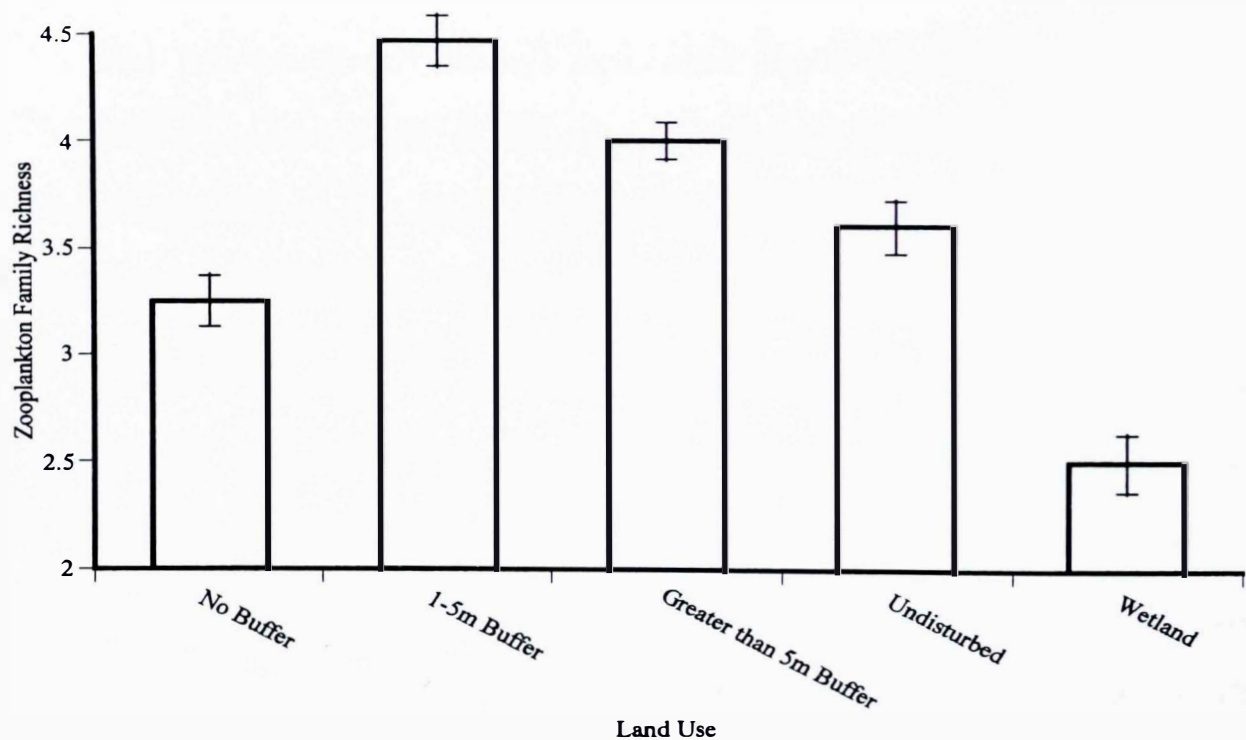


**Figure 21.** Mean total density of zooplankton by land use type for all 74 sites in Great Pond.

Richness and Shannon-Wiener Index values were auto correlated ( $r^2=0.71$ ), so only richness was analyzed for significance between land use types. However, graphical representation of Shannon-Wiener indices of the different land use types showed no trend (Figure 22). Sites with a 1-5m buffer had the highest family richness with  $4.47 \pm \text{SE } 0.36$  families observed per site, and wetlands had the lowest richness with  $2.50 \pm \text{SE } 0.48$  families observed per site (Figure 23). The difference between these land use types was the only significant difference in an one-way ANOVA comparing richness and land use ( $F_{4,69}=3.45$ ,  $p=0.0126$ ), as indicated by the Scheffe post hoc test ( $p=0.025$ ). Densities for each zooplankton family were not significantly different at each land use type (One-way ANOVAs,  $p>0.03$ ).



**Figure 22.** Shannon-Wiener Index of zooplankton families by land use type for all 74 sites in Great Pond.



**Figure 23.** Mean zooplankton family richness by land use for all 74 sites in Great Pond.

Densities for each zooplankton family were not significantly different at each land use type (One-way ANOVAs,  $p > 0.03$ ).

### *Discussion*

The prediction that residential sites would have a higher density and lower diversity of zooplankton than undisturbed and wetland sites was not supported. Undisturbed sites had lower zooplankton densities than residential sites, but this difference was not significant, nor was there a significant difference between zooplankton densities in residential and wetland sites. This finding may be due to the higher levels of nutrients generally found in wetlands during the fall. In autumn, there is a greater concentration of standing dead plant material, including leaf litter. Upon the decomposition of this organic material dead plant material, nutrients are introduced to the littoral watershed (Kadlec and Wallace 2009). If zooplankton populations in Great Pond are responding to bottom-up processes, this increase in nutrients could cause increases in zooplankton densities. The relatively higher densities of zooplankton at wetland sites in comparison to undisturbed sites may also be due to the higher presence of aquatic macrophytes at wetland sites. These macrophytes can act as a refuge for zooplankton (Jeppesen *et al.* 1997; Jeppesen *et al.* 1998; Perrow *et al.* 1999). Also, the reduced wave action at wetland sites could account for an increase in zooplankton densities.

Sites with 1-5m buffers and >5m buffers had greater family richness than sites with no buffer; however, sites with buffers also had greater family richness than undisturbed and wetland sites. The only significant relationship in the Scheffe analysis showed lower family richness in wetland sites than in sites with a 1-5m buffer. These findings are contrary to what was expected, so the initial hypothesis was rejected.

Perhaps insufficient sampling effort can account for these differences, since lower densities were observed at undeveloped and wetland sites than at residential sites. For example, the water column at many wetland sites was shallow. Consequently, samples gathered often contained sediment and lower volumes of water were towed in wetland sites.

Diversity was similar across all sites and no significant differences or general trends were observed among the different land use types. Since diversity takes both richness and evenness into account, this similarity may indicate that evenness is consistent across the different land use categories. Relatively similar proportions of families found across the different land use types support this hypothesis (Figure 20). Only undisturbed sites deviated from the general trend, since they were dominated by Polyphemidae, although Bosminidae dominated all other land use types. Since the densities of zooplankton were lower at undisturbed sites than at the other sites, fewer individuals were counted from each site; therefore, the samples assessed are less likely to be representative of the actual relative densities of families at undisturbed sites.

The prediction that residential sites with larger buffers will have a lower density but higher diversity and richness of zooplankton than residential sites with no buffer was not supported. While differences in zooplankton densities were not significant by land use type, densities were highest at sites with >5m buffers, which was unexpected. This may indicate that factors other than land use are important in determining the distribution of zooplankton within a lake. However, family richness was found to be highest in sites with 1-5m buffers, and sites with >5m buffers had higher family richness than sites with no buffer. In residential sites, the presence of a buffer yielded a greater family richness

than in sites with no buffer. This corresponds with our predictions, even though the difference was not statistically significant in Scheffe post hoc comparisons. However, sites with 1-5m buffers had greater richness than sites with >5m buffers. This was not expected, and could indicate that the presence of a buffer is more important than the size of the buffer.

Increased nutrient input from developed residential sites was expected to contribute to increases in zooplankton densities through bottom-up mechanisms (Stemberger *et al.* 2001). However, there were no significant differences or clear trends in the water quality data among the different land uses to support the hypothesis that increased nutrient inputs are responsible for the increase in zooplankton densities. The absence of a trend may be due to the low sensitivity of the chemical tests used, as discussed in Nitrate Section, but may also indicate that other factors are more important in determining the distribution of zooplankton density within Great Pond.

Water temperature, although not considered in this study, is an important factor influencing the distribution of zooplankton in freshwater lakes. Several studies have demonstrated that zooplankton migrate deeper within the water column in response to cooler temperatures (Allan 1976). Because the current study was conducted in the fall, average temperatures decreased over the course of sampling period. Temporal effects were controlled for by testing for influence of date in the MANOVA, and date of sampling was found to be non-significant for zooplankton density, richness, and diversity.

Wind and currents can also have an important influence on the distribution of zooplankton density within a lake, regardless of land use. The influence of wind and



currents is determined by the morphology of the lake basin and topography around the lake (Folt *et al.* 1993; Jones *et al.* 1995). These effects were partially controlled for in a MANOVA, which tested for differences among sites clustered into seven groups on a spatial basis. This test found no significant differences among sites by spatial location, except in no buffer sites. This may suggest that wind influence on the distribution of zooplankton is moderated by the presence of wind-breaking vegetation in the buffer, although further study is necessary to validate such a hypothesis.

Finally, the prediction that Calanoida copepods would be less abundant than Cyclopoida copepods and that small Cladocera, such as Bosminidae, would be more abundant than large Cladocera was supported. Individuals from the order Cyclopoida were observed more frequently than individuals from the order Calanoida, represented by individuals belonging to the family Centropagidae, at all land use types. Bosminidae was the most abundant family observed, especially in comparison to the relatively rare Daphniidae. The rarity of Daphniidae in this study may be a reflection of seasonality since Daphniidae blooms in the summer (Allan 1976). The dominance of small Cladocerans and Cyclopoida is likely reflective of the location of the sampling. These families are generally more abundant in the littoral zone, where as larger Cladocera, such as Daphniidae, and Calanoida families are more prevalent towards the center of lakes (Jakobsen and Johnson 1987).

## Fish

### *Introduction*

Local fish assemblages have a number of unique characteristics that make them a vital component of rapid bioassessments in aquatic systems. Fish play a unique role in habitat coupling, which refers to the connections that exist between different habitats. Because fish are mobile and represent a diversity of foraging types and trophic levels, they are instrumental in the coupling of benthic, pelagic, and riparian habitats in lake systems (Schindler and Scheuerell 2002; Lampert and Sommer 2007). As a result, many effects must be considered in the assessment of fish assemblages.

Fish are effective as a long-term indicator of habitat conditions because of their relatively long life spans compared to other assemblage types (Karr 1987). Therefore the status of fish communities may reflect many different aspects of the lake ecosystem (Schindler and Scheuerell 2002). Other advantages of sampling fish assemblages include non-destructive sampling, due to easy in-field identification and measurement and a comparatively strong information base regarding life histories (Barbour *et al.* 1999).

The potential impacts of human influence on fish populations are extensive and take many forms, and vary with intensity of anthropogenic activity. Eutrophication of lakes, resulting from increased runoff due to shoreline development, leads to initial increases in fish production (Hanson and Leggett 1982; Goldman and Horne 1983). This occurs in conjunction with an overall increase in biomass in the littoral zone due to increases in limiting nutrients (Wetzel 2001; Lampert and Sommer 2007). Despite increases in production, however, eutrophication also has negative implications for fish.

Increases in nutrient supply often lead to less desirable species composition of fish assemblages (Goldman and Horne 1983). A commonly observed progression is from species highly valued by anglers, such as Salmoniformes (salmon and trout), to Perciformes (perch-like fishes), and finally to Cypriniforms (roaches; Persson *et al.* 1991). Additionally, excessive eutrophication results in degradation of water quality, such as hypoxia or anoxia, which may lead to fish kills (Goldman and Horne 1983; Lampert and Sommer 2007).

Another implication of shoreline development is loss of fish habitat. Complex habitats, such as CWD and macrophytes, act as refuges for juvenile fishes, providing substrate, food, and escape from predators (Crowder and Cooper 1982; Everett and Ruiz 1993; Schindler and Scheuerell 2002). Removal of CWD has been correlated with increased human development (Marburg *et al.* 2006). Loss of these habitats has been linked to decreased fish growth and reduced abundance of smaller fish species. (Schindler and Scheuerell 2000; Sass *et al.* 2006). Destruction of complex habitats may also contribute to changes in species composition (Persson *et al.* 1991).

In addition to shoreline development, human introduction and stocking of non-native species can have a significant impact on fish communities and lakes in general. Threats to local fish populations include eradication of native species through competition and predation, introduced diseases, and dilution of genetic variation (Goldschmidt *et al.* 1993; Krebs 1994; Cowx 1998). Introduced fish also affect other organisms in lakes. For example, Parker *et al.* (2001) showed that stocked brook trout changed zooplankton species composition by feeding preferentially on larger species. Fish introductions have also been used to control algal blooms. In these instances,

piscivorous fish such as northern pike, which feed upon planktivorous fish, are added. Reduction of planktivorous fish populations increases populations of herbivorous zooplankton that reduce algae abundance (Carpenter *et al.* 1985; Berg *et al.* 1997).

The current study had two main objectives with respect to fish. The first was conducting an overall assessment of the littoral fish population of Great Pond. This included measures of species richness and abundance, as well as mean length of captured fish. Second, the impact of differences in land use type on fish assemblages was considered. CEAT expected higher species richness, abundance, and average total lengths in areas with fewer anthropogenic disturbances, such as undisturbed and wetland sites. As human influence increases, richness, abundance, and size were expected to decline. This is because human presence is related to loss of complex habitats such as CWD, which are vital to littoral fish assemblages (Schindler and Scheuerell 2000; Marburg *et al.* 2006; Sass *et al.* 2006). This prediction contrasts with previous studies that have found that biomass of fish increases with development due to higher nutrient availability from runoff (Hanson and Leggett 1982). Despite this, the importance of habitat loss was expected to outweigh the effects of nutrient loading, as has been found in other studies of temperate lakes (Schindler and Scheuerell 2000; Sass *et al.* 2006).

### *Methods*

Gear types and methods used for sampling fish assemblages have inherent strengths and weaknesses. As a result of these sampling biases, each gear type results in a slightly different representation of resident populations. Ideally, multiple gear types are employed in the hopes of canceling out discrepancies among the individual types (Johnson *et al.* 2005; MacRae and Jackson 2006; Ruetz *et al.* 2007). Seine nets, minnow

traps, and backpack electrofishers have all been shown to be effective gear types in littoral lake habitats (Pierce *et al.* 1994; MacRae and Jackson 2006; Ruetz *et al.* 2007). Electrofishing was not an option for CEAT at the time of this study. Minnow traps are able to sample a reasonable array of the fish population but are susceptible to local environmental factors such as changes in water temperature, which cause biases in species and size (Backiel and Welcomme 1980; Blaustein 1989). Seine nets are able to capture an accurate representation of population and are easy to use, but they are susceptible to frequent snags in complex habitats, thereby reducing efficiency. Seine nets and minnow traps were used initially because of availability. However, minnow trap fishing was discontinued after one week of sampling because catch rates were too low for beneficial analysis of the fish present in the littoral zone.

Fish in the littoral zone were sampled in September and October 2010 for each of the 74 different sites on Great Pond. Six-meter beach seines with 0.64cm mesh were used with methods modified from Pierce *et al.* (1990) and Pierce *et al.* (1994). Sites were seined with three successive hauls, each covering one-third of the sample area. All individuals caught were photographed, identified to species level, and measured for total length (mm). Due to low catch rates, data on littoral fish assemblages were excluded from statistical analysis. Fish size, abundance, species richness, and distribution were treated as qualitative data.



## *Results*

Due to the inefficiency of seine nets in complex habitats, fish samples were sparse across all land use types. This precluded the use of inferential statistics, and instead only descriptive statistics are employed.

Out of 74 sites, fish were caught in 14 sites. In total, 116 individual fish representing eight different species were caught. These included smallmouth bass, largemouth bass, brown bullhead, golden shiner, emerald shiner, black crappie, nine-spine stickleback, and banded killifish. Of these, only the smallmouth bass, largemouth bass, black crappie, and the brown bullhead have been reported previously in Great Pond (MDIFW 2007). Banded killifish were most abundant, representing 59% of all fish caught, followed by emerald shiner (27%) and brown bullhead (9%). Per-site abundance varied between 0 and 52 fish, though the majority of sites had 0 individuals. In terms of fish captured, the most productive associated land use type was no buffer, where 69% of all individuals were caught. In contrast, no fish were caught at 1-5m buffer sites. Other land use types had similarly low catch rates (Table 3). Catch rate varied with the presence of different microhabitats. Average fish caught per site was the highest in sandy habitats ( $4.5 \pm \text{SE } 2.8$ ), while all other microhabitats had an average of less than one fish per site (Table 4).

**Table 3.** The abundance of fish by species and land use types in the littoral zone in Great Pond.

Species	No Buffer	1-5m Buffer	>5m Buffer	Undisturbed	Wetland	All Land Use
Black Crappie	0	0	0	0	1	1
Nine-spine Stickleback	0	0	1	0	0	1
Largemouth Bass	1	0	0	0	2	3
Smallmouth Bass	1	0	0	0	0	1
Brown Bullhead	0	0	0	0	10	10
Emerald Shiner	22	0	0	9	0	31
Banded Killifish	56	0	6	6	0	68
Golden Shiner	0	0	1	0	0	1
Total Catch	80	0	8	15	13	116

**Table 4.** The average abundance of fish per site by microhabitat type in the littoral zone in Great Pond.

Microhabitat Type	Average Fish per Site	Standard Error
Rocks	0.2	0.2
Muck	0.3	0.1
Coarse Woody Debris	0.3	0.2
Macrophytes	0.3	0.2
Small Woody Debris	0.3	0.3
Sand	4.5	2.8

Total length of individuals varied from 10mm to 82mm, with a mean of  $47.6 \pm \text{SE } 1.5\text{mm}$  for all species. Mean total length varied by land use type, with the greatest value,  $64.0 \pm \text{SE } 2.3\text{mm}$ , found at wetlands, followed by no buffer ( $48.3 \pm \text{SE } 1.7\text{mm}$ ), >5m buffer ( $42.8 \pm \text{SE } 3.8\text{mm}$ ), and undisturbed sites ( $33.1 \pm \text{SE } 2.3\text{mm}$ ; Table 5).

**Table 5.** The average total length of fish found in different land use types in the littoral zone in Great Pond.

	Average Length (mm)	Standard Error (mm)
No Buffer	48.3	1.7
1-5m Buffer	0	0
>5m Buffer	42.8	3.8
Undisturbed	33.1	2.3
Wetland	64.0	2.3
All Land Use	47.6	1.5

### *Discussion*

Samples of the littoral fish community were determined to be inaccurate due to extremely low sample size and considerable difficulties encountered when sampling complex habitats. Although seining is an effective technique for collecting fish in smooth benthic habitats, efficiency decreases with increased snagging due to obstructions such as rocks, roots, and macrophytes (Pierce *et al.* 1990). Snagging causes the bottom lead line of the net to lift from the substrate, providing a means of escape for captured fish (Pierce *et al.* 1990; Murphy and Willis 1996). Furthermore, juvenile fish in the littoral zone are known to seek protection in more complex habitats such as coarse woody structures and macrophytes (Lewin *et al.* 2004). Such complications make seining even more inefficient because it is least effective at sampling the microhabitats where a large portion of littoral fish assemblages is expected to be found. These findings are consistent with this study. Nearly all Great Pond sample sites had some complex obstruction, such as rocks and coarse woody debris, and the majority of these yielded no fish. Few sites consisted of solely smooth substrate, and it was at these sites that the largest catches were recorded. Smooth substrates tended to be found at no buffer sites, and thus the no buffer land use had the highest catch-rates. Due to the sampling bias, the data are considered to be not

only inaccurate and insufficient, but also incomparable, as microhabitat occurrence varied between land use types. As a result, conclusions regarding the influence of land use type on species richness and relative abundance cannot be made.

Variation in the average total length of fish between land use types can be attributed to differences in species composition. Wetland sites, which had the highest mean total length, also had the larger fish species represented, such as brown bullhead and largemouth bass. This contrasts with undisturbed, >5m buffer, and no buffer sites, which recorded smaller mean lengths and harbored mostly smaller species: banded killifish and emerald shiners. Again, conclusions about the influence of land use type on total length and species composition cannot be made.

Three of the eight species found have not been previously recorded in Great Pond. These include the cyprinids emerald shiner, banded killifish, and nine-spine stickleback, all of which appear elsewhere in Maine (MDIFW 2007). These species may have been introduced to Great Pond since the time of the most recent fish survey in 2007 (Knowledge Base 2007). Species introductions by humans have been common in Maine's history and many non-native species have been introduced since the late 18<sup>th</sup> century (MDIFW 2007) For example, anglers may have introduced baitfishes such as emerald shiners into the lake. Alternatively, these non-game fishes may have been overlooked in past surveys that were focused on more highly valued species. Furthermore, the populations of these fishes may have increased in recent years as part of succession due to shoreline development. Fish assemblages have been shown to transition from salmonid to cyprinid dominance due to nutrient loading and loss of habitat (Persson *et al.* 1991).

This sort of shift may account for why the observed species have become more common. More complete sampling is required to confirm this trend in Great Pond.

The implications of shoreline development for the littoral fish community of Great Pond cannot be assessed given the limitations of the current dataset. Future study should aim to more efficiently sample all microhabitat types in order to accurately describe local assemblages and assess the strength of the community. Multiple gear types should be employed, especially electroshocking, as this will help account for weaknesses of any individual type.

## **Macroinvertebrates**

### *Introduction*

Aquatic macroinvertebrates are commonly used in bioassessment because their ubiquitous nature allows for different habitats to be compared. Furthermore, macroinvertebrates exhibit a range of responses to environmental stressors, their sedentary nature enables them to show localized effects of perturbations, and their life cycles allow temporal changes to be examined (Merritt *et al.* 2008). Therefore, aquatic macroinvertebrates accumulate the effects of both episodic and chronic disturbances, making them especially useful to assess the overall impacts of stressors (Barbour *et al.* 1999).

Macroinvertebrate habits and functional feeding groups are good indicators of community structure and organization (Merritt *et al.* 2008). “Clingers” is a habit description of organisms that can attach to the lake floor even in turbulent waters, and the percent of clingers present in a community is expected to decrease in response to environmental perturbation (EPA 2010a). Scrapers and filterers are descriptions of the



functional feeding groups of invertebrates. Scrapers graze on algae and associated materials that are attached to mineral and organic surfaces; filterers are suspension-feeders that consume fine particulate organic matter (Merritt *et al.* 2008). Scrapers and filterers are affected by organic toxicants, and filterers are the first functional feeding group expected to decrease in the presence of disturbance (Burton and Pitt 2001).

Shoreline development has been shown to significantly impact macroinvertebrate assemblages in German lowland lakes. Densities of pollution-tolerant macroinvertebrates increased with enhanced human development (Brauns *et al.* 2007). CEAT aimed to determine the impact of smaller-scale developments, such as private residences, on immediately adjacent macroinvertebrate communities, and to make inferences about overall lake health. Differences were expected in macroinvertebrate community assemblages between undeveloped and residential sites. A decrease is expected in taxa richness, evenness (which is combined with richness in diversity metrics), and composition measures as a response to increasing perturbation from residential development. CEAT expected to find high numbers of pollution-intolerant taxa at undisturbed and wetland sites. High numbers of pollution-tolerant taxa was expected at residential sites, since pollution-intolerant species would give way to more pollution-tolerant species as perturbation increased.

### *Methods*

Each aquatic site within the littoral zone was surveyed for seven microhabitats, modified from Brauns *et al.* (2007) and Strakosh (2009): roots, CWD, SWD, macrophytes, sand/silt, stones, and muck. CEAT sampled each habitat type for macroinvertebrates with techniques adapted from Brauns *et al.* (2007) and Kuhlmann *et*

*al.* (2005). Macroinvertebrates from the root habitat were collected with a D-net (12"x14", 500  $\mu$ m) in a vertical sweep. Similarly, macrophyte microhabitats were sampled in 1m horizontal sweeps with the D-net. Macroinvertebrates were brushed off of three pieces of CWD and/or SWD with similar states of decay. To sample sand/silt and muck microhabitats, a 30x30 cm<sup>2</sup> quadrat was randomly placed within the site and substrate was collected using a trowel. Rock microhabitats were also sampled by picking up each stone within a 30x30 cm<sup>2</sup> quadrat and brushing off macroinvertebrates. Each technique described above was repeated three times per site.

Beginning with 30-September-2010, macroinvertebrates were placed in a *Nasco* Whirl-Pak ®, preserved in 95% ethanol, and brought back to the laboratory for subsequent identification. Samples collected before 30-September-2010 were refrigerated. At the laboratory, silt/sand and muck samples were passed through a sieve tower (smallest mesh size at 125 $\mu$ m) in order to separate the invertebrates from the substrate. Macroinvertebrates found in roots, macrophytes, CWD, SWD, and rocks samples were either sieved or picked out by hand. All specimens were identified to the family level or lowest possible taxonomic level using identification keys in Merritt *et al.* (2008), Pennak (1953), Edmondson (1959), Needham and Needham (1962), Voshell (2002), and the MDNR (2003).

In order to determine the effect of shoreline development on the littoral zone, community structure was established by site and microhabitat. Total abundance is the total number of individuals in each sample or sub-sample (Merritt *et al.* 2008). The data were grouped and analyzed to determine basic metrics about the communities sampled and examine trends among land use types and microhabitats. Taxa richness, Simpson's

Diversity Index, and total abundance were calculated. Taxa richness was used instead of species richness, because identifying individuals to a higher taxon is simpler and less time-consuming than identifying individuals to species level. Taxa richness is closely related to species richness, allowing this modification to produce usable results (Balmford *et al.* 1996). Taxa richness is a measure of the total number of taxa in a community, and is expected to decrease as water quality declines (Merritt *et al.* 2008). The Simpson's Diversity index gives the probability that two consecutive samples will have invertebrates from the same family, thus representing richness and evenness of the macroinvertebrates living in the littoral benthos (Hunter and Gaston 1988; Mandaville 2002). It was calculated as:

$$D = 1 - \sum (p_i)^2$$

where  $p_i$  is the proportion of individuals in the "i<sup>th</sup>" taxon of the community and the  $s$  is the total number of taxa in the community (Krebs 1994). The index ranges from 0 to 1, with values closer to 1 representing a more diverse community. Total abundance is the total number of individuals in each sample or sub-sample (Merritt *et al.* 2008).

Family Biotic Index (FBI) was also calculated. Developed from Hilsenhoff's (1988) Biotic Index, the FBI provides a single pollution-tolerance value that is the average of the tolerance values of all species within the benthic community. Tolerance values range from 0 (very intolerant) to 10 (highly tolerant) based on a family's tolerance to pollution (Mandaville 2002). The FBI was calculated as:

$$FBI = \sum_{i=0}^{10} \frac{x_i * t_i}{n}$$

where  $x_i$  is the number of individuals in the " $i^{\text{th}}$ " taxon,  $t_i$  is the tolerance value of " $i^{\text{th}}$ " taxon, and  $n$  is the total number of organisms in the sample (PDEP 2009; Krebs 1994).

Additionally, for each land use type EPT was calculated. EPT refers to three orders: Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). The EPT index equals the total number of families represented within the three orders Ephemeroptera, Plecoptera, and Trichoptera (Mandaville 2002). However, this index was modified to indicate the percentage of families represented by the three orders, with the aim of providing more comparable results among sites. A high percentage of these taxa in a sample suggests that the water quality is pristine enough to support a large number of these intolerant species. An even distribution of EPT across the lake is a measure of community balance and uncontaminated biotic conditions (Burton and Pitt 2001).

Functional feeding groups were also analyzed in order to determine if there were differences in the dominance of particular feeding groups in relation to development (Mandaville 2002). The percentages of scrapers and filterers were calculated across land use types. In addition, macroinvertebrate habits were analyzed and the percentage of clingers at each land use type was calculated. These metrics allowed CEAT to glean more information about the macroinvertebrate community structure and organization of the littoral zone across land use types.

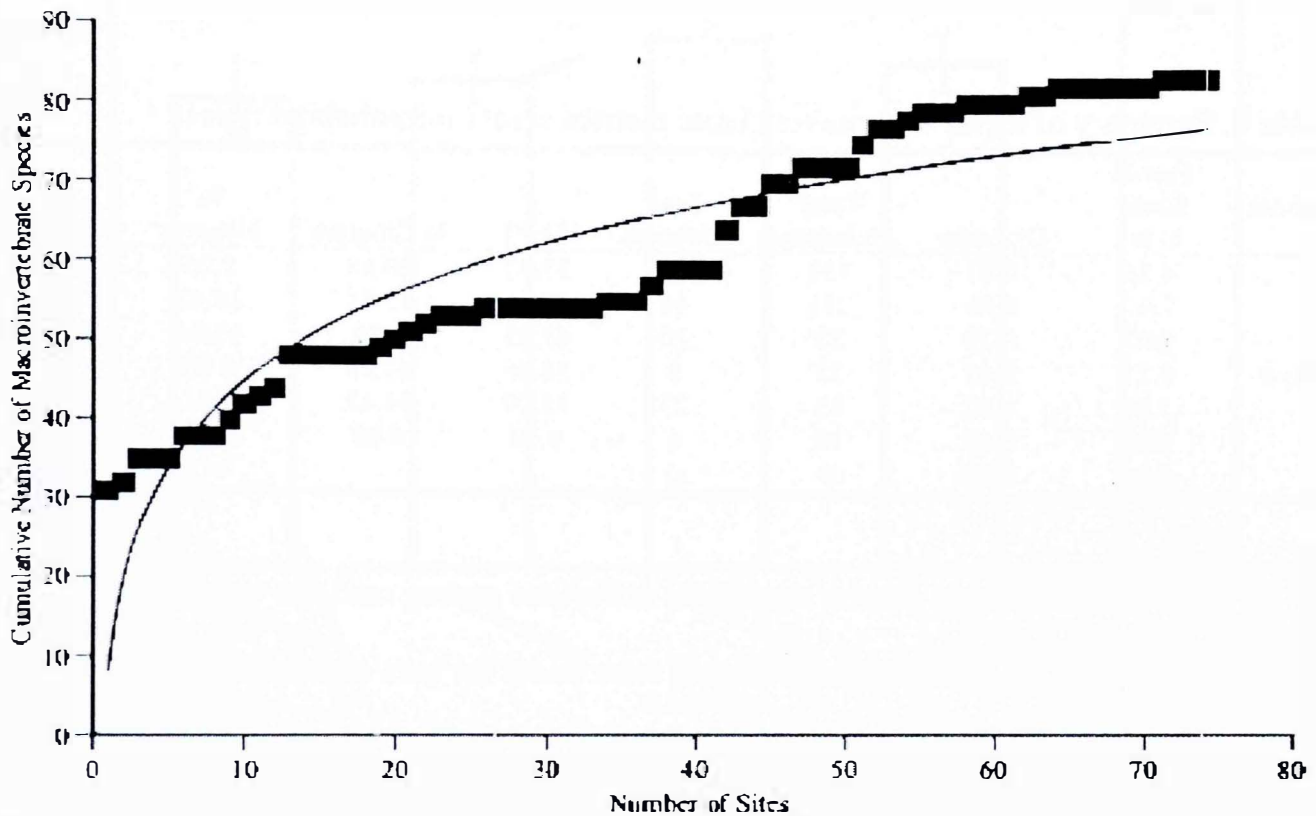
We conducted one-way ANOVAs to determine if there were any significant differences in the macroinvertebrate metrics between land use types. Scheffe post hoc tests were also performed to establish which individual variables exhibited significant differences. Furthermore, a linear regression was performed to determine the relationship between macrophyte taxa richness and macroinvertebrate taxa richness.

## *Results*

All seven microhabitats sampled for macroinvertebrates were found in at least one of the 74 sites. Rock microhabitats were the most frequently found at all land use types except for wetlands, where muck was predominant. Not all samples collected from microhabitats contained macroinvertebrates; a total of 13 out of 99 microhabitat samples contained no macroinvertebrates (two rock, nine sand, one CWD, and one muck).

CEAT identified specimens from 81 macroinvertebrate taxa, most of which were identified to the family level. Only 9 specimens could not be identified to family level. One specimen was identified to the species level, although it was caught in a minnow trap: a rusty crayfish (*Orconectes rusticus*). Sampling efficiency was plotted with the number of sites on the x-axis and the cumulative number of species on the y-axis. Additionally, a logarithmic curve of best fit was added ( $r^2=0.8201047$ ). This was done to assess whether our samples were representative of the actual variation of species present. (Figure 24).





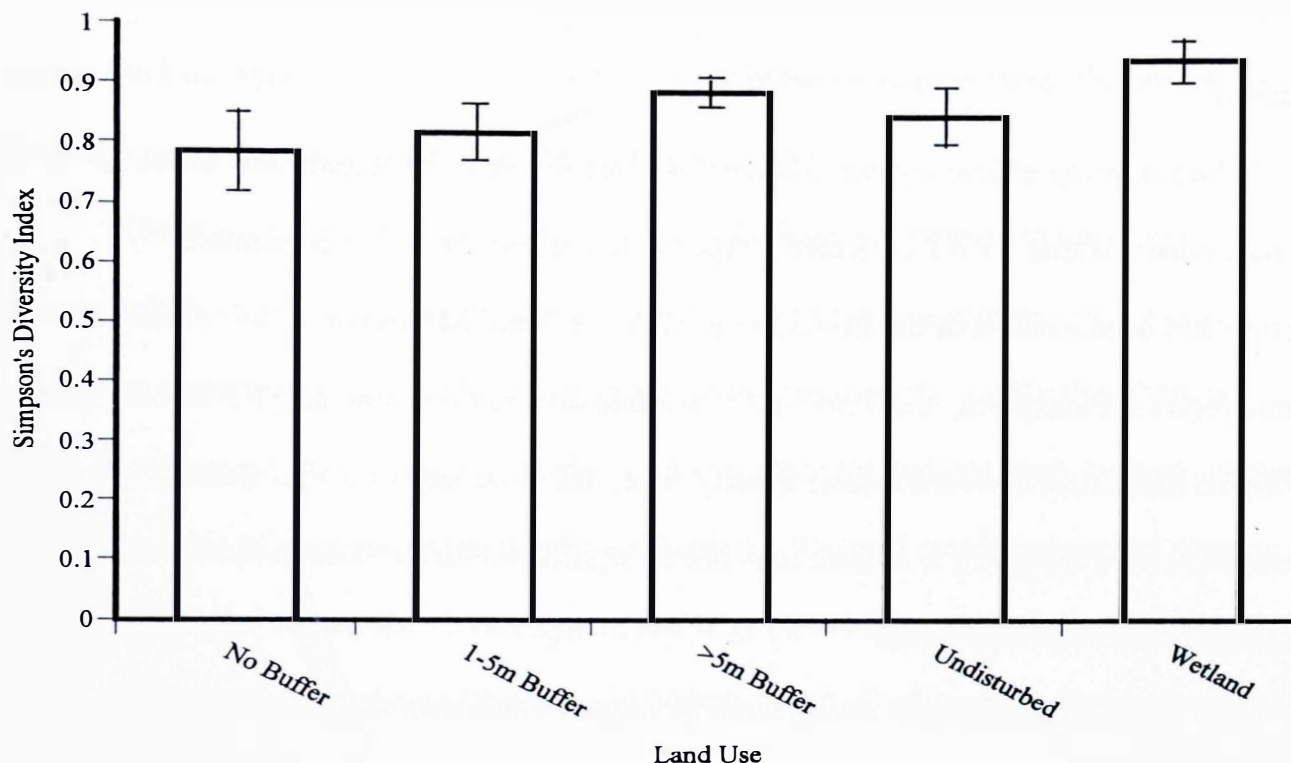
**Figure 24.** Sampling efficiency of macroinvertebrates for all 74 sites in Great Pond.

Although there were no significant differences in macroinvertebrate metrics between microhabitat types, rock microhabitats had the lowest FBI and the highest total abundance and taxa richness, whereas muck had the highest FBI and the greatest number of filterers. Sand microhabitats had the highest EPT. Macrophyte microhabitats had the greatest number of clingers. The microhabitat with the highest diversity was CWD (Table 6). A linear regression also showed that there is a weak but significant correlation between macroinvertebrate and macrophyte richness, ( $r^2=0.1991$ ,  $p=0.0001$ ).

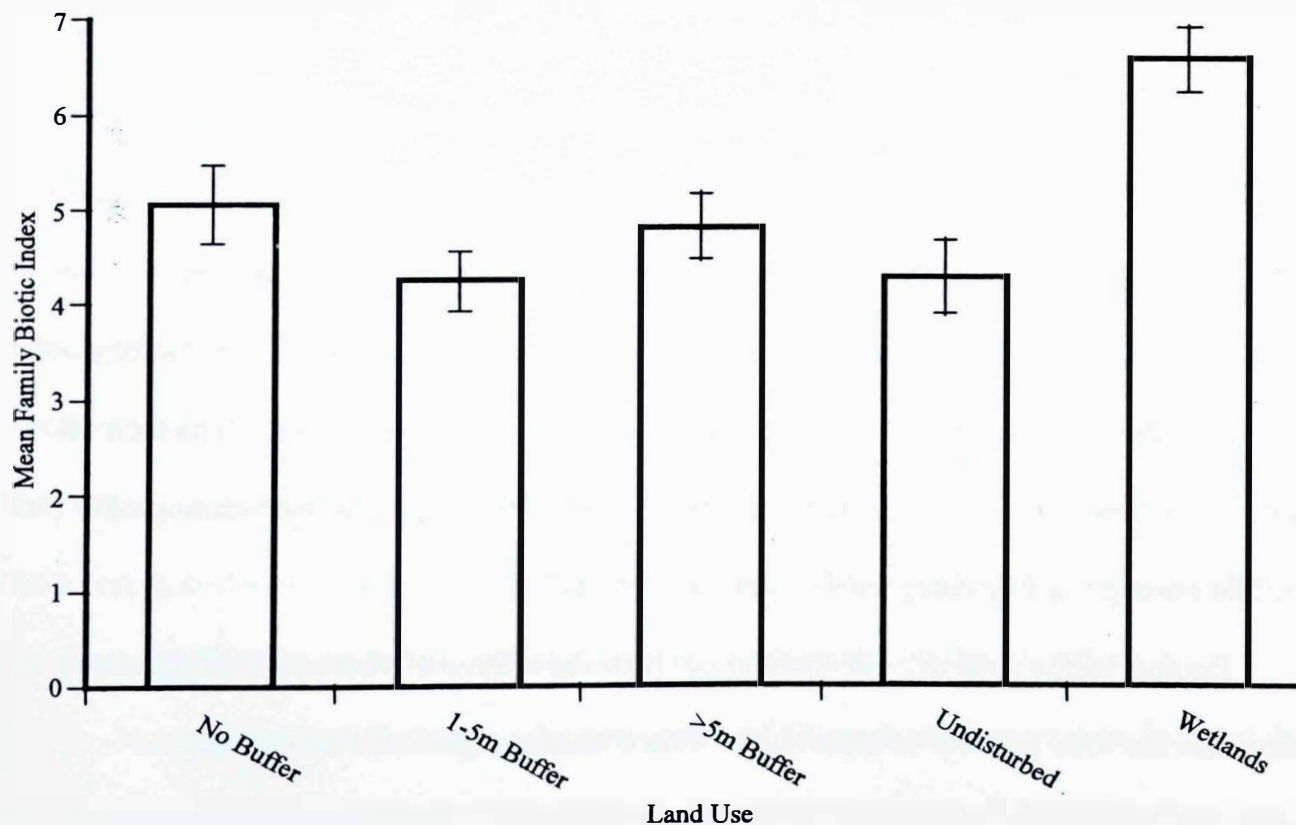
**Table 6.** Summary of mean macroinvertebrate metrics across microhabitat types.

Microhabitat Type	Family Biotic Index	Diversity	Total Abundance	Taxa Richness	%EPT	% Clingers	% Filterers	% Scrapers
Rock	4.36	0.81	753	60	33.73	20.58	2.25	64.04
Muck	7.07	0.06	211	44	18.01	15.31	28.42	25.14
Sand	6.07	0.79	23	10	47.83	8.70	13.04	0.00
Macrophyte	6.21	0.63	52	9	38.46	44.23	0.00	0.00
CWD	5.86	0.92	85	23	15.29	34.12	9.41	34.12
SWD	6.83	0.63	18	6	0.00	0.00	0.00	0.00
Roots	0.00	0.00	0	0	-	-	-	-

Most metrics did not show a significant difference among land use types using one-way ANOVAs. No significant differences were found for taxa abundance ( $F_{4,69}=0.22$ ,  $p=0.9238$ ), taxa richness ( $F_{4,69}=0.97$ ,  $p=0.4313$ ), EPT ( $F_{4,57}=2.06$ ,  $p=0.0985$ ), filterers ( $F_{4,57}=1.37$ ,  $p=0.2561$ ), scrapers ( $F_{4,57}=1.02$ ,  $p=0.4004$ ), and Simpson's Diversity index ( $F_{4,57}=0.65$ ,  $p=0.6306$ ) among land use types (Figure 25). The one-way ANOVA found a significant difference in the percent of clingers between land use types ( $F_{4,57}=5.62$ ,  $p=0.0007$ ). A Scheffe test showed that significant differences in clingers were between wetlands and no buffer sites ( $p=0.025$ ) and between wetlands and undisturbed sites ( $p=0.002$ ). A significant difference was also found in the FBI among land use types (One-way ANOVA,  $F_{4,57}=6.05$ ,  $p=0.0004$ ; Figure 26). A Scheffe test showed that the only significant differences in FBI were between wetlands and 1-5m buffer sites ( $p=0.005$ ), between wetlands and >5m buffer sites ( $p=0.04$ ), and between wetlands and undisturbed sites ( $p=0.003$ ).



**Figure 25.** Simpson's Diversity Index for macroinvertebrates by land use type for all 74 sites in Great Pond.



**Figure 26.** Mean Family Biotic Index of macroinvertebrates by land use type for all 74 sites in Great Pond.

## *Discussion*

The sampling efficiency curve did not deviate far from the logarithmic curve of best fit, indicating that CEAT collected a representative sample of the community structure and organization of the littoral zone of Great Pond. Most taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera are sensitive to pollution, so EPT values can provide an indication of overall water quality. Therefore, the lack of a significant difference in EPT indicates that there may not be significant differences in biotic conditions across land use types.

As scrapers and filterers are affected by organic toxicants, the lack of significant differences between the two feeding groups suggests that there may be no difference in the concentration of organic toxicants across land use types.

Simpson's Diversity Index is a measure of community structure since it takes into consideration taxa richness and evenness as opposed to the varying insensitivities of taxa involved (EPA 2004). Simpson's Diversity values closer to 1 indicate higher richness and evenness. Since our mean Simpson's Diversity values for each land use type were between 0.79 and 0.94, and there is no significant difference among them, our findings reflect equality in community diversity across land use types on Great Pond. The lack of significant differences in taxa richness among land use types suggests that water quality is similar among the five categories designated by CEAT.

Enumerations such as taxa abundance show the effects of stressors that increase or decrease the total number of individuals. There was no significant difference in abundance, suggesting that there is no significant difference in environmental stressors

across land use types. However, due to the variety between sample sizes, this metric can be problematic for detecting trends (Merritt *et al.* 2008).

The significant difference in the number of clingers between land use types, particularly between wetlands and the four other land use types, suggests that wetland sites have more tolerant taxa. However, this does not necessarily signify that there is more development at wetland sites than elsewhere, since the indices used were developed for the analysis of streams and rocky littoral habitats. Wetland areas accumulate organic carbon, may have lower dissolved oxygen, and may have higher temperatures than streams and typical lakeshores (Speiles and Mitsch 2000). Furthermore, the burrowing organisms often found in wetlands, such as oligochaetes, polychaetes, insect larvae, and shrimp can increase oxygen concentrations and facilitate the growth of plant species (Mermillod-Blondin and Lemoine 2010). Since clingers are adapted to live in streams and wave-swept rocky littoral lake zones, they are not expected to be found in wetlands, because the required habitat would not be as readily available. Since many wetlands have abundant detritus, the most common functional feeding group is likely to be collectors, which consume detritus (Speiles and Mitsch 2000; Merritt *et al.* 2008;). However, because scrapers and filterers are more intolerant to anthropogenic sources of pollution, they were considered to be the most appropriate bioassessment tool (Burton and Pitt 2001). As a result, collectors were not analyzed since their presence or absence would provide minimal information on the impact of shoreline development on the littoral zone.

No significant difference in the percentage of filterers or scrapers was found at the different land use types. This is likely because the communities at all land use types are composed of a variety of functional feeding groups, such as predators, grazers, piercers,



and shredders, and one group does not dominate others. The diversity of feeding groups contrasts with that of habitat types, such as clingers, that are dependent upon the physical and chemical conditions of the habitat (Merritt *et al.* 2008).

The significance between FBI and land use types may also be attributed to the inherent habitat difference of wetlands compared to other land use types. FBI was developed to assess the impact of anthropogenic sources of pollution, such as sewage effluent and pasturing of cattle (Hilsenhoff 1988). Unfortunately, FBI does not differentiate between natural and artificial sources of pollution. Wetlands are often nutrient sinks (Spieles and Mitsch 2000), which may produce misleading results. FBI was significantly higher at wetland sites. However, this is likely not due to shoreline development, but rather because of the inherently different characteristics of wetland habitats.

To assess the presence of a relationship between macroinvertebrates and their physical environment, a correlation analysis was conducted between macroinvertebrate and macrophyte richness. There was no significant relationship, indicating that presence or absence of macrophytes does not exclusively define macroinvertebrate distribution. In conclusion, other environmental factors, such as chemical or biological perturbations, may be more important in impacting macroinvertebrate diversity and distribution.

Rusty crayfish (*Orconectes rusticus*) are an invasive species in Maine and throughout the northeast and Midwestern U.S. (Phillips 2010, USGS 2010b). The species is typically brought into a watershed through live bait fisheries, and then quickly moves throughout the system. It eliminates macrophytes (Phillips 2010) and causes a decline in many other species, including fish competing for the same food resources and other

macroinvertebrates such as gastropods, trichopterans, and odonates (Hobbs *et al.* 1989; Wilson *et al.* 2004). Native crayfish species were nearly eradicated from a lake in Wisconsin after the introduction of *O. rusticus* (Wilson *et al.* 2004). The presence of this invasive species in Great Pond merits more investigation so that preventative measures can halt the progress of this species throughout the Belgrade Lakes region.

## Summary of Findings

Analysis of the collected data revealed several trends. First, water quality data indicated that there may be a significant relationship between nutrient loading and land use types. Nitrate levels varied significantly by land use types. Nitrate levels at no buffer sites were found to be above naturally-occurring levels when data from all dates were considered. However, excluding data from 14-October-2010, nitrates analyses indicated that nitrate levels at no buffer land use types were at naturally-occurring levels, below 0.30ppm (WLPR 2010), indicating that elevated nitrate levels may be due to localized effects. Total dissolved phosphorus testing yielded no statistical significant difference between land use types; however, all mean total dissolved phosphorus levels were above the naturally-occurring levels of 0.01ppm to 0.03ppm (HACH 1983). Therefore, Great Pond is at risk for algal blooms. Testing for water hardness found no significant differences between land use types. The highest mean levels were observed at wetland sites, though the overall hardness of the lake is considered soft. Soil phosphate levels showed no significant difference and no correlation between soil and water phosphorus levels; additionally, statistical analysis concluded that mean soil phosphate levels did not vary by land use.

Significant habitat findings were noted in variables ranging from microhabitat cover, substrate cover, grade, and terrestrial vegetation. Wetlands were assessed to have the most overall differences from other land use types. No-buffer sites were found to be similar to wetland sites in terms of shore grade and lake floor grade, macrophyte presence, and terrestrial vegetation composition. However, no-buffer sites did not contain CWD, in contrast to the abundance of CWD in wetland sites. Mean coverage by rocky

substrates (cobble, rock, boulder) was found to be higher at >5m buffer, 1-5m buffer, and undisturbed sites than at wetland and no-buffer sites. Sites with higher levels of muck and sand had higher concentrations of macrophytes than sites with primarily rocky substrates. Out of 19 sites with macrophytes, 15 sites had muck as a representative substrate. There was a higher concentration of macrophytes in sites with shallower lake floor grades, although there was not a strict correlation between the two variables. However, there was significant variance of shore grade among land use types.

Faunal analysis was separated into three major groups: zooplankton, fish and macroinvertebrates. Zooplankton family richness was highest in sites with buffers, with lowest family richness at wetland sites. Fish sampling was deemed inaccurate due to small sample size, methodological difficulties, and lack of differentiated gear type use. For the samples that were collected, wetland sites tended to harbor larger species, such as brown bullhead largemouth bass, whereas undisturbed and no buffer sites tended to contain mostly smaller species such as banded killifish and emerald shiners. Three previously unrecorded species were captured during sampling: emerald shiner, banded killifish, and nine-spine stickleback. Additionally, an invasive decapod species, the rusty crayfish (*Orconectes rusticus*), was captured in a minnow trap at a site with a 1-5m buffer. Analysis of macroinvertebrate diversity indicated that pollution-intolerant species were present, particularly at rocky sites. Any significant differences found in macroinvertebrates between land use types can be attributed to natural habitat variation rather than human impacts.

## Recommendations

The Colby Environmental Assessment Team has developed a list of suggestions for landowners around Great Pond in order to maintain or improve the state of the lake. While the littoral zone of Great Pond is currently in good condition, increased development may reduce water quality, damage habitats, and negatively impact faunal and macrophyte populations. The following recommendations provide guidelines to prevent future degradation.

### *1) Maintain vegetative buffers to protect the shoreline from erosion.*

Buffers act as natural barriers to erosion as a result of their root structure, which holds the soil in place. Erosion can be damaging in multiple ways. It deposits sediments in the littoral zone, changing the lake floor slope and altering the microhabitats and substrate (Alin *et al.* 1999). Additionally, contaminated or nutrient-rich sediments resulting from fertilizer use, sewage leaks, and storm drainage can enter the lake, altering nutrient levels and distribution (S.C.O.A.L.E. 1989). Increased levels of nutrients can lead to eutrophication. Maintaining vegetative buffers can reduce the likelihood of a lake becoming eutrophic.

Residents can become recognized as LakeSmart by a DEP certified Soil and Water Conservation District employee. The LakeSmart program emphasizes the importance of planting native trees and shrubs along the water's edge to help repair eroding areas and divert rainwater into vegetated areas (MDEP 2010a).

Changes in terrestrial vegetation on Great Pond should be monitored in order to see patterns over time. These patterns can provide a better understanding of the effects of increased development and the positive influence of buffers on lake health.



## ***2) Emphasize wetland protection.***

Wetlands offer important services to lakes and ponds, and their protection is necessary to ensure future lake health. Wetlands harbor a diversity of microhabitats, and the highly organic, nutrient-rich substrate supports a variety of macrophyte types. Altering wetlands can cause the soil to release nitrates and phosphates into the lake (Richardson 1985).

The Natural Resources Protection Act protects wetlands across the United States. This legislation regulates the alteration of wetlands by regulating removal or displacement of soil, sand, vegetation or other materials. It also regulates draining, filling, and construction or repair of permanent structures located in wetlands (DEP 2009). Maine residents should continue to protect wetlands surrounding Great Pond by avoiding development of wetlands and adjacent areas, as this could affect wetland function.

## ***3) Monitor non-native macrophyte, macroinvertebrate, and fish species.***

Non-native species can be faunal or vegetative, and both types have potentially devastating effects when introduced into native Maine ecosystems. In some cases, introduced species outcompete their native counterparts, which can change the elaborate food web of the ecosystem (Goldschmidt *et al.* 1993; Krebs 1994; Cowx 1998). Information on how to identify and report invasive aquatic plants can be found on the Maine DEP website (MDEP 2010b).

## ***4) Monitor phosphate and nitrate levels and sources spatially and seasonally.***

CEAT and the 2010 Maine Integrated Water Quality Report determined that phosphorus is a pollutant of Great Pond (MDEP 2010c). In 2010, the report lists Great

Pond in the Summary of Impaired Waters due to the high Total Maximum Daily Load of phosphorus.

Since elevated phosphate and nitrate levels can indicate increased nutrient loading due to human action, consistent monitoring can help to identify problem areas before severe water quality degradation occurs. Nitrates and phosphates should be measured at several locations throughout the lake to determine spatial differences in nutrient loading. They should also be measured in each of the four seasons, due to changes resulting from differential mixing of temperature layers that occur in the spring and the fall. It would be preferable if measurements in each season were taken at all of the sampling locations on the same day to minimize temporal variation.

The data collected could be used to determine if specific regions of Great Pond are problem areas in terms of nutrient loading. If so, subsequent regulation of point and non-point sources of phosphates and nitrates can be developed to protect water quality. Current Maine law prohibits the use of cleaning agents containing phosphates and the sale of phosphate-containing fertilizers (MDEP 2010d). Because a law is already in effect, monitoring fertilizer and other phosphate-containing household chemical use can be a relatively easy and effective endeavor. A survey-based study coupled with monitoring, potentially through a program like LakeSmart, could determine if phosphate-containing materials are used in the watershed or near the shore of Great Pond. The study could also assess what types of products phosphates are found in, and where these products are used (e.g., residences or agricultural lands).

***5) Avoid further alteration of the natural shoreline.***

In general, development should be regulated and monitored on the shores of Great Pond. By protecting natural ecosystems and monitoring the effects of protection and development, the health of Great Pond can be better understood and addressed when issues like excessive phosphorus arise.

***6) Promote education about lake health.***

One of the best ways to improve lake health is through education. Information on lake health in Maine and how to maintain it is available on the Belgrade Lakes Association webpage (BLA 2010).

**Buffer Resources for Residents:**

Buffer Handbook, Buffer Plant List, Information for Camp Owners, Conservation Practices for Homeowners Fact Sheets, LakeSmart information, and Phosphorus-Free Fertilizer information can be found on the Maine Department of Environmental Protection LakeSmart resources website:

<http://www.maine.gov/dep/blwq/doclake/lakesmart/resources.htm>

**Invasive Species Reporting Information:**

<http://www.state.me.us/dep/blwq/topic/invasives/>

**Maine Lake Health Education:**

<http://www.belgradelakesassociation.com/Resources/WaterQuality/tabid/83/Default.asp>

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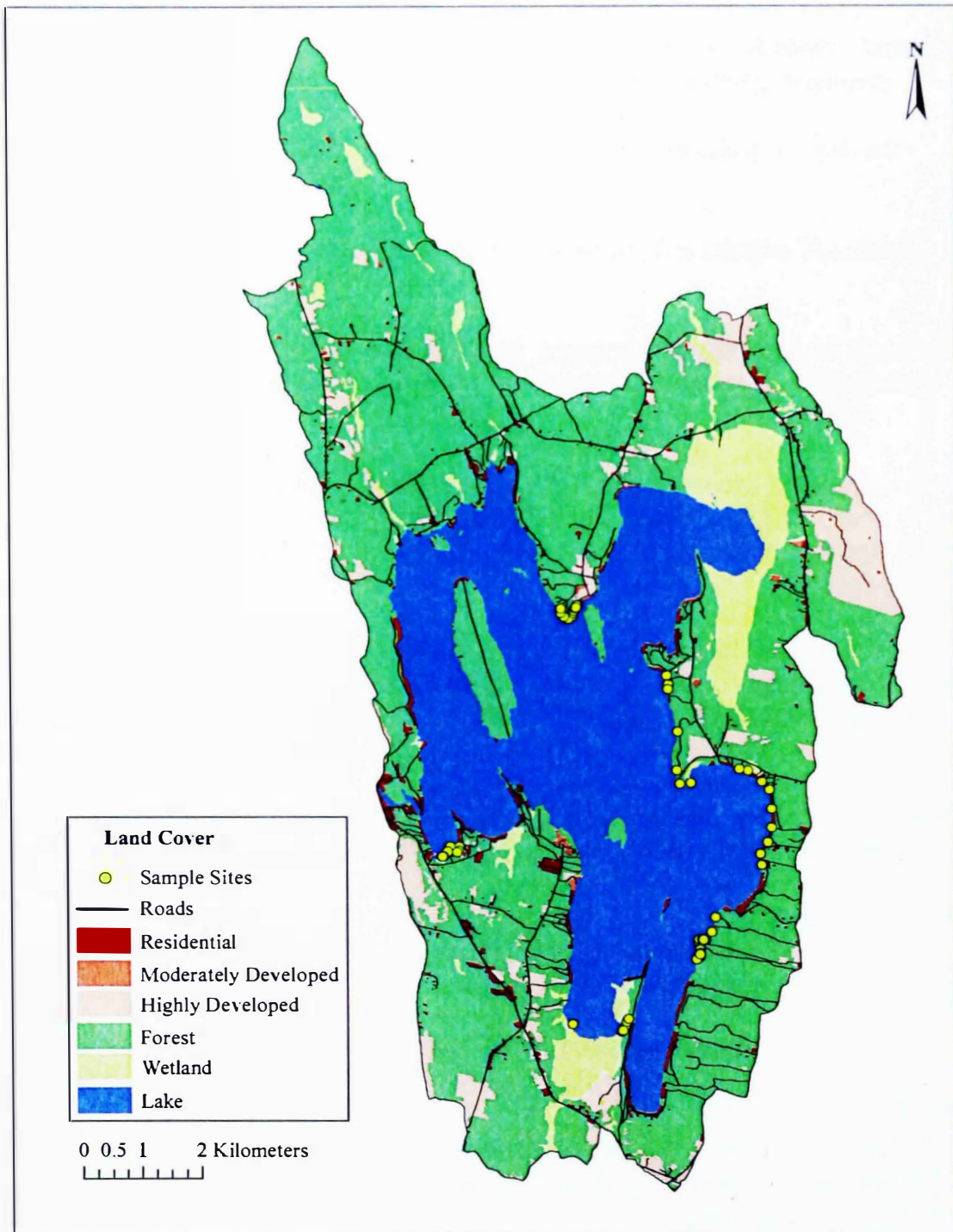
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### III. Appendices

#### *Appendix A. Map of Land Uses Around Great Pond*





## *Appendix A. Map of Land Uses Around Great Pond*

This map of land cover data, was obtained from the Maine Office of GIS, and was reclassified by increments of development in the Great Pond Watershed. Original land cover categories were derived from 1999-2001 Landsat imagery and refined with 2004 SPOT imagery by the Maine Office of GIS. These categories were grouped, by levels of development, into five land covers: residential, highly disturbed, forest, moderately disturbed, and wetland.

**Disturbed:** Bare Ground, Cleared Land, Commercial/Municipal, Cropland/Farm, Paved Road, Dirt Road, Golf Courses, Gravel Pit, Open Field, State Road

**Forest:** Coniferous Forest, Deciduous Forest, Mixed Forest, Transitional Forest and Tree Farm.

**Moderately Disturbed:** Logging Area, Park, Regenerating Land, Reverting Land

**Residential:** Camp, shoreline residential and non-shoreline residential.

**Wetland:** Wetland

## *Appendix B. Field Data Collection Sheets*

### **DATA SHEET**

DATE:

GPS NUMBER:

COORDINATES:

LAND USE

CLASSIFICATION:

COLLECTION TEAM

NAMES:

CAMERA

NUMBER:

SITE PHOTO NUMBERS:

### ***Habitat classification***

Use visual inspection to estimate percent coverage of the entire site for each habitat classification.

	< 1%	1% -- 10%	11% -- 25%	26% -- 50%	51% -- 75%	76% -- 100%
Large woody debris						
Small woody debris						
Rootwads						
Vegetation						

### ***Water chemistry***

pH:	
temperature:	

### ***Shore slope***

Record vertical height of shore above the water level 1m and 2m from water's edge. Use level to ensure tape measure is flat. These measurements should be taken in the center of the site.

1 m:	
2 m:	

### ***Dominant shoreline vegetation***

Record the dominant terrestrial vegetation type(s) at the site. Be as specific as possible. Also take a picture of the area.

Photo numbers:						
Vegetation types:						

### **Quadrat placement**

Record random number (1-3) used to place quadrat within subsite. Subsites are numbered from left- right, facing water.

Subsite 1:	
Subsite 2:	
Subsite 3:	
Subsite 4:	

### **Macrophyte identification:**

Identify all macrophytes present in the quadrat to the family level. If possible, identify to the species level. See guide for identification. Take a sample and a photo of each new macrophyte encountered. Also take detailed notes for identification purposes in field notebooks. For each species, also estimate the percent area covered by that species.

<b>Macrophyte type</b>	<b>Photo #</b>	<b>&lt;1%</b>	<b>1% - 10%</b>	<b>11% - 25%</b>	<b>26% - 50%</b>	<b>51% - 75%</b>	<b>76% - 100%</b>

### **Substrate**

Identify the dominant sediment type within the quadrat. Muck refers to organic material.

	<i>Muck</i>	<i>Mud</i>	<i>Sand</i>	<i>Gravel</i>	<i>Cobble</i>	<i>Rock</i>	<i>Boulder</i>
Quadrat 1:							
Quadrat 2:							
Quadrat 3:							
Quadrat 4:							

### **Canopy cover**

Follow the instructions on the inside lid of the spherical densiometer. Face the shoreline and record the total dots **unoccupied**. Do this along the shoreline and 3m out on the aquatic edge, 5m from the side of the site (2 readings total).

shoreline:	
aquatic:	

### **Lake floor slope**

Measure the water depth 1.5m and 3m from shoreline. This measurement should be taken at the center of the site.

1.5m:	
3m:	

### ***Appendix C. Sediment Used in Substrate Classification***

#### **Description of sediments used in substrate classification**

Substrate	Description
Muck	Decomposed organic material
Sand	0.063mm - 2mm
Gravel	2mm - 64mm
Cobble	64mm - 128mm
Rock	128mm - 512mm
Boulder	>512mm

### ***Appendix D. List of Macrophytes Found***

List of Macrophytes Found in the Littoral Zone of Great Pond.

Family	Number of Sites
Nymphaeaceae	5
Hydrocharitaceae	10
Cyperaceae	5
Eriocaulaceae	6



### *Appendix E. Terrestrial Vegetation*

List of Terrestrial Vegetation Found in the Littoral Zone of Great Pond.

Family	Number of Sites
Aceraceae	31
Aquifoliaceae	3
Asteraceae	6
Betulaceae	35
Caprifoliaceae	3
Cornaceae	1
Cupressaceae	28
Dryopteridaceae	2
Ericaceae	24
Fagaceae	9
Hamamelidaceae	4
Myricaceae	15
Osmundaceae	1
Pinaceae	43
Poaceae	9
Rhamnaceae	1
Rosaceae	7
Rosaceae	7
Rubiaceae	3
Salicaceae	2
Ulmaceae	1

### *Appendix F. Fish Species*

Fish species previously observed in Great Pond as of a 2007 survey.

Common Name	Scientific Name
American Eel	<i>Anguilla rostrata</i>
Black Crappie	<i>Pomoxis nigromaculatus</i>
Brown Bullhead	<i>Ameiurus nebulosus</i>
Brown Trout	<i>Salmo trutta</i>
Chain Pickerel	<i>Esox niger</i>
Fallfish	<i>Semotilus corporalis</i>
Golden Shiner	<i>Notemigonus crysoleucas</i>
Landlocked Alewife	<i>Alosa pseudoharengus</i>
Landlocked Salmon	<i>Salmo salar</i>
Largemouth Bass	<i>Micropterus salmoides</i>
Northern Pike	<i>Esox lucius</i>
Pumpkinseed	<i>Lepomis gibbosus</i>
Rainbow Smelt	<i>Osmerus mordax</i>
Redbreast Sunfish	<i>Lepomis auritus</i>
Smallmouth Bass	<i>Micropterus dolomieu</i>
White Perch	<i>Morone americana</i>
White Sucker	<i>Catostomus commersoni</i>
Yellow Perch	<i>Perca flavescens</i>

## *Appendix G. List of Macroinvertebrates*

### List of Macroinvertebrates Found in Each Land Use Type

Order	Family	No Buffer	1-5m Buffer	>5m Buffer	Undisturbed	Wetlands
Actiniedida	Eylaidae					X
Amphipoda	Gammaridae	X	X	X	X	X
	Talitridae	X	X	X	X	X
Annelida	Haplotaxidae					X
Annelida	Oligochaeta					X
Bivalvia	Pelecypoda					X
	Sphaeriidae					X
	Unionidae					X
Coleoptera	Amphizoidae		X			
	Chrysomelidae				X	
	Dryopidae	X	X		X	
	Elmidae		X	X	X	
	Halplidae			X	X	
	Hydrophilidae		X	X	X	X
	Limnichidae/Lutrochidae			X		
	Noteridae			X		
	Psephenidae	X	X	X	X	X
	Staphylinidae	X				
Collembola		X				
Decapoda	Astacidae				X	
	Cambaridae			X		
Diptera	Ceratopogonidae	X		X		X
	Chaoboridae	X				
	Chironomidae	X	X	X		X
	Dixidae					X
	Psychodidae		X			
	Simuliidae					X
	Stratiomyidae					X
	Tabanidae					X
	Tipulidae	X				X
Ephemeroptera	Baetidae		X	X		X
	Caenidae		X			
	Ephemerellidae	X		X	X	
	Heptageniidae	X	X	X	X	X
	Leptohyphidae	X				
	Leptophlebiidae		X			
	Potamanthidae			X		
	Tricorythidae		X	X	X	
Gastropoda	Amnicolidae	X			X	X
	Ancylidae					X
	Balvatidae					X
	Bithyniidae	X		X		

	gastropoda		X		X	X
	Helisoma			X		
	Hydrobiidae	X	X	X		X
	Lymnaeidae		X			X
	Physidae		X			X
	Planorbidae		X	X	X	X
	Valvatidae	X	X			X
	Viviparidae	X	X	X	X	X
Hemiptera	Macroveliidae					X
	Notonectidae					X
	Pleidae					X
Hirudinea		X			X	X
Isopoda	Asellidae	X	X	X		X
Lepidoptera	Crambidae			X		
Megaloptera	Sialidae	X			X	
Odonata	Aeshnidae			X		
	Coenagrionidae	X	X	X	X	X
	Lestidae					X
	Libellulidae			X		X
	Macromiidae				X	
	Zygoptera				X	
Oligochaeta	Branchibdellidae				X	X
Platyhelminthes	Turbellaria	X	X	X	X	
Plecoptera	Perlidae	X	X	X	X	X
	Perlodidae	X				
	Pteronarcyidae				X	X
Rhynchobdellida	Glossiphoniidae		X			X
Trichoptera	Brachycentridae				X	
	Glossomatidae					X
	Hydropsychidae	X				
	Hydroptilidae		X	X	X	X
	Lepidostomatidae				X	
	Leptoceridae			X	X	X
	Odontoceridae		X		X	
	Philopotamidae		X	X		X
	Phryganeidae					X
	Polycentropodidae	X	X	X	X	X
	Psychomyiidae	X				
	Rhyacophilidae					X
Tricladia	Dendrocoelidae	X				
Trombidiform	Hydrachna		X			
Turbellaria	Planariidae		X		X	
	Turbellaria			X		