INTERNAL PHOSPHORUS LOAD INACTIVATION IN MYSTIC LAKE, BARNSTABLE, MASSACHUSETTS, 2010

FINAL REPORT

BY WATER RESOURCE SERVICES, INC.

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IN COOPERATION WITH AQUATIC CONTROL TECHNOLOGY

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Executive Summary

Mystic Lake is part of the Indian Ponds, a set of three kettlehole ponds in Marstons Mills, MA, a village of Barnstable on Cape Cod in Massachusetts. It covers 149 acres (60 ha) to an average depth of 4.6 m (15 ft) with a maximum depth of 14.3 m (47 ft) and a volume of 3128 acre feet (3.86 million m^3). It supplies recreational opportunity (swimming, boating, fishing, nature watching), aquatic habitat (including an alewife nursery), and water for cranberry bog flooding. Long considered a natural resource jewel, Mystic Lake experienced noticeably increased algae and decreased water clarity over the last decade. Studies by the Cape Cod Commission and AECOM Corporation determined that increased phosphorus concentrations were the cause and that internal loading was the primary source of phosphorus. Plans to conduct a phosphorus inactivation treatment using aluminum compounds were delayed in the permitting process as a consequence of concern over potential impacts to one of the most diverse and abundant mussel communities in the Commonwealth, including species listed for protection under the Massachusetts Endangered Species Act. However, a major mussel die off in August of 2009 and additional die off in 2010 reduced that mussel community by about 90% and indicated that the current level of fertility was not beneficial to mussels. The inactivation project was then allowed to proceed and was conducted in September and early October of 2010.

Treatment planning by WRS included collection of additional sediment samples and lab assays to determine both the effectiveness of varying doses and the maximum aluminum concentration tolerable to fish and invertebrates during treatment. It also included additional pre-treatment water quality assessment and development of a monitoring plan for the period of treatment and a year afterward. Actual treatment was conducted by Aquatic Control Technology and involved the application of a total of 10,466 kg (23,025 lbs) of aluminum over 58.1 acres of the lake in six defined areas, mostly deeper than 30 ft, with doses varying by treatment area from 35 to 50 g/m^2. Two areas would have been treated at higher doses, but the permit specified a maximum dose of 50 g/m^2. The targeted ratio of aluminum sulfate to sodium aluminate was 2:1 by volume, intended to yield minimal change in pH. The applied volumes were 21,002 gallons of aluminum sulfate and 10,553 gallons of sodium aluminate. Treatment was conducted on six days over a 15 day period, beginning on September 21, 2010, with most areas receiving two treatments of half the intended dose in each treatment, several days apart. Adjacent areas were not treated consecutively to allow movement of mobile organisms to nearby untreated areas. One area was shallower than normally treated, but was treated to include mussel beds that were being monitored for possible impacts.

Monitoring during treatment revealed no pH values outside the targeted range of 6.0 to 8.0 SU and no depletion of alkalinity. Very few dead fish were encountered on daily surveys, and those found mostly showed obvious signs of mortality due to causes other than aluminum toxicity. In a detailed study by Biodrawversity, no mussel mortality or behavioral abnormalities were attributable to the aluminum application. It is likely that chironomid larvae and oligochaete worms living in anoxic sediments were smothered, but these were not considered to be resources to be protected, and rapid colonization was expected. The treatment was conducted safely from human health and ecological perspectives.

Based on substantial historical data from the Pond And Lake Steward program, water clarity is found to be linked to algal biomass, which is linked to phosphorus, which is linked to oxygen in Mystic Lake. When the lake stratifies at a shallower depth, more bottom area is subjected to low oxygen and possible phosphorus release, there is more volume in the lower water layer to absorb that phosphorus, more phosphorus is expected to reach the upper waters, more algae are produced and accumulate in the upper waters, and water clarity declines. The treatment bound up large amounts of phosphorus in surficial sediments, limiting the availability of phosphorus that could be released from the sediment when exposed to low oxygen. The treatment also removed a large portion of the phosphorus already released to deep water in the lake in 2010, but was less efficient at removing phosphorus in shallow waters where phosphorus concentrations, while high enough to support algal blooms, are low relative to reaction processes involving aluminum.

From 2001 through 2007, the mass of phosphorus in Mystic Lake averaged about 196 kg and was equivalent to one year of loading, as the flushing rate is conveniently close to once per year. Incomplete sampling in 2008 and 2009 prevent valid estimation of phosphorus mass in the lake, but in 2010, after the mussel die off but before treatment, the mass was measured at 245 kg in August and 214 kg in September. After treatment, phosphorus mass was as low as 86 kg, matching the target for achieving an average concentration of 10 ug/L and preventing algal blooms. Reduction in the internal load was >90%. In May of 2011, with relatively low phosphorus and a zooplankton community of substantial biomass with large bodied grazers dominating, water clarity reached 9 m (30 ft). However, phosphorus in shallow water was not sufficiently reduced by the treatment, and remained high enough to support algal blooms after the alewife run produced many young of the year that decimate the zooplankton community over the summer. Water clarity was higher in summer of 2011 than in recent years and was considerably more stable at between 2 and 3 m, but did not achieve the 6 m clarity associated with the inactivation treatment at Hamblin Pond in 1995, a clarity level that has been sustained for 16 years so far.

The phosphorus mass did not increase significantly over the winter, but increased by 19 kg in the spring prior to stratification, and increased by another 18 kg over the summer of 2011, yielding a mass of 125 kg a year after treatment. This represents a substantial reduction from pre-treatment values, but is not as low as desired. There are multiple mechanisms by which phosphorus mass may have increased. Loading from the watershed in the rather rainy 2011 is possible, as is release of phosphorus from untreated or undertreated sediments. However, the most likely source of phosphorus in 2009 and 2010, with potential carry over into 2011, is from the decay of dead mussels. Up to 24 million mussels are estimated to have died in 2009 and 2010 in Mystic Lake, with each million mussels representing over 13 kg of phosphorus. The loss of over 90% of the mussel community presents a problem in terms of both increased phosphorus load and decreased filtration capacity, and could explain observed algal patterns and water clarity without consideration of other mechanisms.

The data show clearly that internal loading from sediments under >9 m (30 ft) of water depth has been markedly curtailed by the phosphorus inactivation treatment in fall of 2010. If the overall phosphorus load to Mystic Lake from untreated internal sources, most notably dead mussels, works its way through the system as would be expected over about 5 years, Mystic Lake may continue to improve. Monitoring is the most important management activity to be conducted over the next year or two.

Project Background and Need

Mystic Lake is a valued resource within the Town of Barnstable and the Village of Marstons Mills. It was long known as one of the aquatic gems of the town, along with Middle Pond, to which it is connected by a short, narrow channel. More recently, algal blooms have limited water clarity and raised concern. Hamblin Pond, to the south but connected to Middle Pond and Mystic Lake only by ground water flow, experienced major algal blooms linked to internal recycling of phosphorus throughout most of the second half of the 20th century, and was the subject of a restoration effort in 1995. Treatment with aluminum resulted in greatly reduced algal abundance and increased clarity. Evaluation of Mystic Lake was undertaken to determine the sources of overfertility and the potential for phosphorus inactivation to improve conditions as it did for Hamblin Pond. Much of this background information comes from the 2009 AECOM report on this effort, which in turn capitalized on considerable previous and ongoing efforts by other groups.

In 2007 and 2008, AECOM reviewed the existing data and information on Mystic Lake and conducted follow up investigations. Sources of existing information for Mystic Lake and its watershed were gathered from Town of Barnstable files and staff (data, GIS files. maps), regional agencies (Cape Cod Commission), state agencies (EOEA, MA DEP, MA DFW), federal agencies (USGS, NOAA), scientific literature, and the Indian Ponds Association (IPA). For Mystic Lake, a significant amount of useful water quality data were available from the Pond And Lake Steward (PALS) volunteer monitoring program and the recently completed Cape Cod Commission study (Eichner et al. 2006). Additional PALS data have been generated since the AECOM review.

Mystic Lake Features

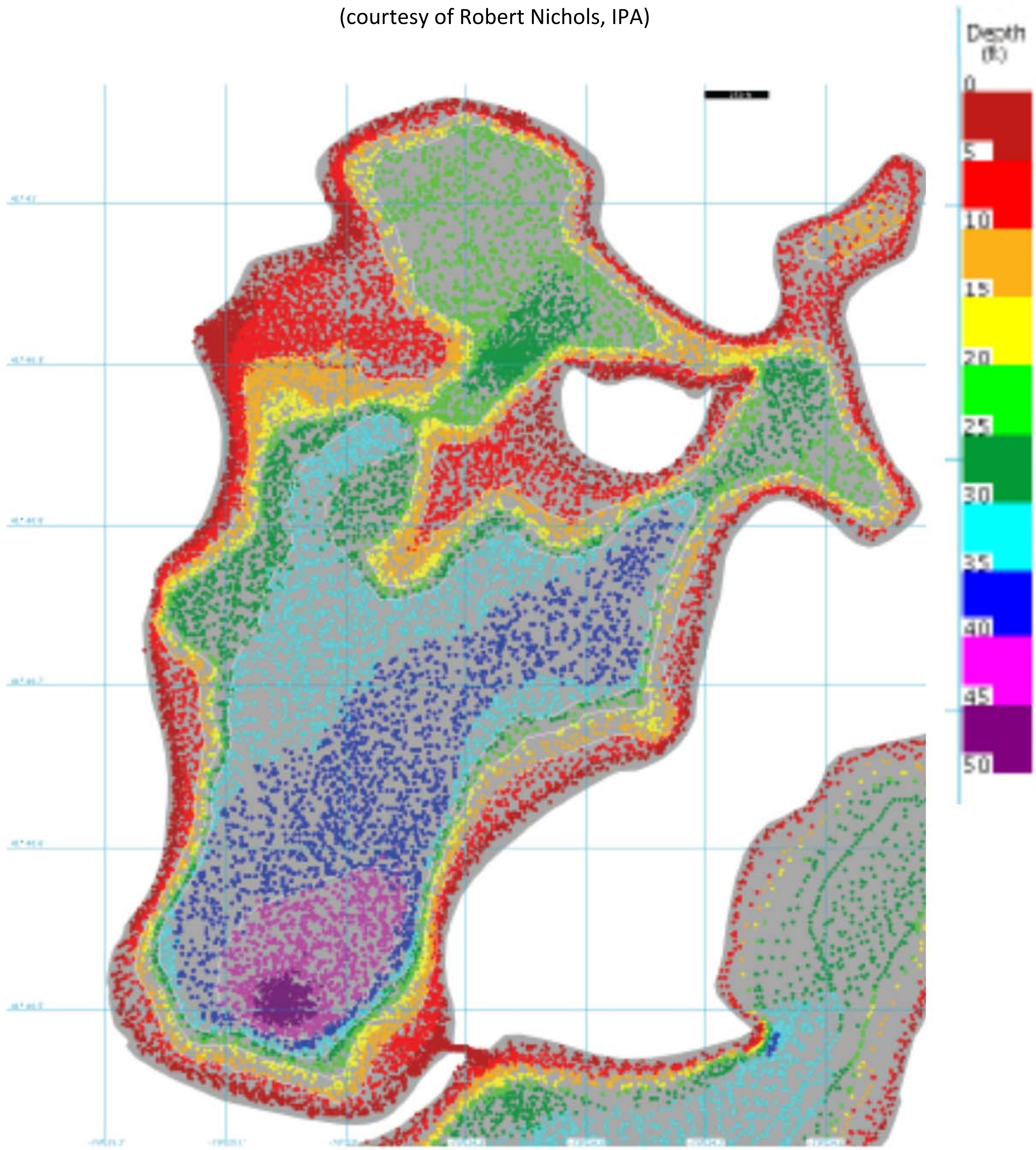
Mystic Lake (60 ha or 149 acres) is one of three hydrologically connected kettlehole ponds collectively known as the Indian Ponds located in Barnstable, MA (Figure 1). Estimates of depth and volume vary somewhat among reports. Maximum depth is 14.3 m (47 ft) and an average depth is about 4.6 m (15 ft). Mystic Lake has a deep basin in the southern part of the lake with shallower embayments to the north and northwest (Figure 2). Water volume for Mystic Lake is about 3.86 million m^3, or about 3128 acre-feet. Detention time is very close to one year, which equates to one flushing per year.

Mystic Lake is classified as Class B waters under the Massachusetts surface water quality standards (314 CMR 4.00). Mystic Lake is also designated as a Great Pond (MA DEP 2007), one of eleven recognized Great Ponds in Barnstable, covering at least 10 acres in its natural state. Mystic Lake is used for recreational fishing, swimming, and boating (CCC 2003), and is also the source for irrigation and harvest flood water for cranberry bogs. With a strong connection to ground water, Mystic Lake is also linked to water supplies on the Cape via wells in this sole source aquifer. A small undeveloped stretch of shoreline off Race Lane at the north end of the lake offers public access to Mystic Lake, including the potential for small boat launching, and there are several community associations with beach and boat launch facilities.

Figure 1. Mystic Lake and the other two Indian Ponds.

Figure 2. Bathymetry of Mystic Lake.

(courtesy of Robert Nichols, IPA)



Mystic Lake Bathymetry
Depth Points Color Coded Every 5 ft
Contour Interval 10 ft
Maximum Depth 47 ft

Mystic Lake supports a warmwater fish community and seasonal runs of anadromous alewife. The herring run is through outlet structure on Middle Pond with access to Mystic Pond via the small surface water connection at the southeastern end of the lake. Herring runs vary widely among years, but there are almost always large numbers of young of the year alewife in Mystic Lake for the summer, and these fish decimate the zooplankton community. Suitable summer habitat for trout is very limited, as the surface waters are too warm and the bottom waters too low in oxygen. Largemouth bass, smallmouth bass, yellow perch and white suckers are abundant, with many large specimens observed.

Mystic Lake and its riparian and littoral zones have been host to a number of state-protected species. The mussel community is unusually rich and diverse for a Massachusetts pond and includes seven species, three of which are protected by the state (Table 1). The presence of protected mussel species was a major factor in project planning and permitting. However, a large die-off occurred in summer of 2009, with additional mortality in 2010, greatly reducing mussel populations. This will be addressed in more detail separately in this report. There are also three species of protected odonates (dragonflies and damselflies) which inhabit emergent and submerged aquatic vegetation in the riparian and littoral zones. These include the Comet Darner (*Anax longipes*), the New England Bluet (*Enallagma laterale*), and the Pine Barrens Bluet (*Enallagma recurvatum*).

Riparian shoreline vegetation has been dominated by dense stands of European grey willow, an invasive species for which an active control project is in progress. Many of these trees have been removed over the last two years. Submergent aquatic vegetation has not been systematically surveyed, but includes water celery (*Vallisneria americana*), waterweed (*Elodea canadensis*), several species of pondweed (including *Potamogeton amplifolius, Potamogeton perfoliatus*,and *Potamogeton robbinsii*), and stonewort (*Nitella flexilis*). Plant growth is limited in many areas by the sandy/cobbly substrate and wave action, but is also greatly affected by low light induced by algal blooms in recent years.

In 2010, while preparing for the aluminum treatment, hydrilla (*Hydrilla verticillata*) was found off an association beach on the west side of the lake by Robert Nichols of the IPA. This highly invasive plant resembles the native waterweed. The monoecious strain is the one that has colonized northern waters to date, and was subsequently found in four nearshore areas of the lake, all in water <5 ft deep, with at least two patches per area. Long Pond in Barnstable also has hydrilla, but there are no more than a dozen lakes in all of New England known to harbor this plant at this time. Birds or boats probably delivered the plant to Mystic Lake, and given the location, an avian source is suspected. Spread to additional, widely separated areas was probably by birds or wind-induced circulation from the initial patch; the other patches are all smaller. Physical control actions including hand harvesting and benthic barrier placement were initiated in late summer of 2010 and continued in 2011. Hydrilla has spread northward from the larger patch, and has been subject to hand pulling by commercial divers, but appears under control in the three areas with smaller patches.

Table 1. Mussel Community of Mystic Lake

Latin Name	Common Name	Status
Alasmidonta undulata	Triangle Floater	**SC***
Anodonta implicata	Alewife Floater	
Elliptio complanata	Eastern Elliptio	
Lampsilis radiata radiata	Eastern Lampmussel	
Ligumia nasuta	Eastern Pond Mussel	**SC**
Leptodea ochracea	Tidewater Mucket	**SC**
Pyganodon cataracta	Eastern Floater	

* SC = species of special concern (MA NHESP).

The MA NHESP, as part of its "Living Waters Project," identified Mystic Lake and Middle Pond as "Core Habitats" because they provide habitat for rare plant and animal species and/or exemplary habitats. In addition, the land upgradient of Mystic Lake is identified as a "Critical Supporting Watershed" since it has the greatest potential to influence habitat conditions within the waterbody. It is not clear how the increasing frequency of algal blooms or infestation with hydrilla affect this designation, but this is a clear case where action is needed to rehabilitate and protect the habitat; it will not remain viable habitat for the valued biological components on its own, as evidenced by the events of the last few years.

Mystic Lake flows into Middle Pond which then flows into the Marstons Mills River and eventually the marine environment (an area known as Three Bays). There are no surface tributaries to Mystic Lake. Precipitation and groundwater in-seepage are the dominant sources of water. Water leaves Mystic Lake as groundwater out-seepage discharging to Middle Pond or as surface water outflow through the narrow connector. Evaporation is a lesser but significant exit route for water in summer. Total hydrologic through-flow was estimated for Mystic Lake and suggests an average annual detention time of slightly greater than 1 year, with the flushing rate varying somewhat over the seasons.

Inspection of the PALS monitoring database (2001-2006), pond assessments (CCC 2003 and Eichner et al. 2006), and additional investigation by AECOM in fall 2007 and spring 2008 have documented consistent patterns of deep water anoxia with regeneration of phosphorus from bottom sediments in areas of > 9 m (30 ft) of water depth. In some cases the anoxia extends upward to depths of 8 m or even shallower; in 2009 anoxia reached 5 m and in 2010 it reached 7 m. Stratification is not strong in the northern half of the lake which is not much deeper than 30 ft in its deepest areas, and wind storms can mix the anoxic, nutrient laden water with the overlying water periodically. Stratification is very strong in the southern deep hole area, promoting reduction of sulfates and formation of hydrogen sulfide. Iron is the most common natural binder of phosphorus in this area, but binding of iron by sulfides is likely to disrupt the natural iron-phosphorus binding cycle and allow available phosphorus to reach the upper waters in significant quantity.

Bottom sediments were collected and analyzed from the lake in 2007. Sediment samples were either fine-grained and highly organic in nature (muck) and contained large amounts of total phosphorus or coarse-grained (sand) and deficient in nutrients. Analysis of the phosphorus fractions indicated that large amounts were contained in the iron-bound phosphorus fraction that would be susceptible to release under low redox conditions that occur during anoxia each summer.

Examination of the conditions that promote phosphorus release included assessment of oxygen demand. Average areal oxygen demand in Mystic Lake is estimated at a range of 521 to 1042 mg O_2/m^2/day, in the range associated with eutrophic lakes with substantial internal phosphorus loading. This demand causes strong anoxia as soon as stratification sets in and bottom waters are denied further atmospheric oxygen inputs. This drastically changes water chemistry in the deep zone, making it unsuitable for the vast majority of aquatic organisms and allowing accumulation of dissolved phosphorus and ammonium nitrogen.

Much of this phosphorus and nitrogen remains in the deepest bottom water layer for the summer, but windy weather can cause some mixing and concentration gradients will allow some diffusion. The Osgood Index was calculated by AECOM at 8.1, with 8 as the dividing line between systems that mixed freely or stayed stratified; the northern portion of Mystic Lake can be expected to destratify in response to strong wind events. Diffusion is driven by the difference in phosphorus levels between water layers, and this difference is typically an order of magnitude by mid-summer; diffusion can be expected.

Where adequate iron is present, much of the phosphorus will recombine with that iron when oxygen is present, and will precipitate back to the bottom. This up and down movement of phosphorus, controlled by iron dynamics, is sometimes referred to as the “ferrous wheel”, an enjoyable play on words but no more enjoyable than the ride when the wheel breaks down. Scavenging of iron by sulfides, which are also produced by anoxic chemical reactions, results in a loss of dissolved iron and more available phosphorus in the upper waters. The phosphorus promotes algal growth, which rains oxygen demanding organic matter onto the lake bottom, fueling further anoxia in deep water and creating a cycle that is both self-sustaining and accelerating. Watershed management will not break this cycle, and algal blooms can be expected without further inputs from land around the lake.

One other phenomenon that is observed in Mystic Lake is the accumulation of algae at the boundary between the upper and lower water layers. This area, known as the thermocline, is fairly narrow (about 1-2 m thick), has some light and a lot of nutrients. Certain types of algae, most notably the cyanobacteria, thrive at this level and can form dense bands. Under mixing conditions, which can occur from summer wind storms or eventual fall destratification, these algae will move upward. Additionally, several types of cyanobacteria form gas vesicles and can become buoyant, floating up to the surface with plenty of nutrients already in their cells and causing blooms. These algae accumulate nutrients from deeper water, float upward to get more light, and grow to the extent that nutrient reserves allow, then producing resting stages that settle to the bottom and facilitate the process in the future. The release of nutrients, especially phosphorus, from bottom sediments can therefore be very influential.

Estimates of the levels of internal recycling were made by AECOM on the basis of observations of the accumulation of hypolimnetic phosphorus and by modeling. The hypolimnion of Mystic Lake accumulated phosphorus with an overall increase of 112.1 kg over the period between May 19 and August 24, 2008. These values translate into benthic release rates ranging from -5.1 to 50 mg P/m^2/day, with an average of 11.6 mg P/m^2/day. The average is very close to the 12 mg P/m^2/day found to be typical for lakes with anoxic bottom layers (Nurnberg 1984, 1987). Only a portion (normally 10-25%) of this phosphorus will reach the surface waters during summer, and this roughly matches the estimate of effective internal loading by Eichner et al. (2006). This phosphorus input is enough to raise the surface water phosphorus concentration sufficiently to support the observed algal levels in the pond. Adding the potential for cyanobacteria to accumulate nutrients near the interface of the two water layers and float upward, bloom probability increases.

A phosphorus budget was prepared by AECOM and shows an estimated load of 120 kg/yr for Mystic Lake. There are many assumptions in this budget, but it is apparent that the annual input from the watershed is not the dominant influence at this time. The internal load is the most important factor in the overall phosphorus dynamics, representing about half the total load and occurring during the summer when inputs are most likely to fuel algal blooms. This trend is unlikely to reverse itself without human intervention. However, it should be kept in mind that the watershed is the ultimate source of the phosphorus and other pollutants and both in-lake and watershed management are appropriate.

Eichner et al. (2006) estimated a lower phosphorus load to the lake, with the internal load calculated as the difference between itemized external loads and the observed mass of phosphorus in the lake at any point in time. As a substantial portion of the external load will enter as particulate forms of phosphorus that will settle to the lake bottom, the internal load is likely to be more important than estimated, but the conclusions drawn by both AECOM and Eichner et al. were similar and justified; the internal load represents the difference between acceptable an unacceptable conditions in this lake and must be addressed to regain the desired level of clarity.

Watershed Features

The Mystic Lake watershed is approximately 421 ha (1140 ac), with a watershed to lake area ratio of approximately 8:1. Delineation of the watershed is complicated by ground water flow patterns, however, and is not exact. With sandy soils, very little of the watershed lands drain directly to the lake; storm water runoff is not a major issue for this system. Land use in the watershed is largely forested, low density residential, and open or protected land, with some cranberry production and upgradient public drinking water wells (Figure 3). There are approximately 37 upgradient residences and septic systems within 300 ft of Mystic Lake (Figure 4), but the 300 ft zone is a somewhat arbitrary limit. Nitrogen will come from much farther away, and it is possible that phosphorus will also reach the lake from distances farther than 300 ft over time, although attenuation of loads will be substantial and the rate of movement will be slow. Ground water quality is an appropriate focus of watershed management in this system.

Figure 3. Contributory land area for Mystic Lake. (from Eichner et al. 2006)

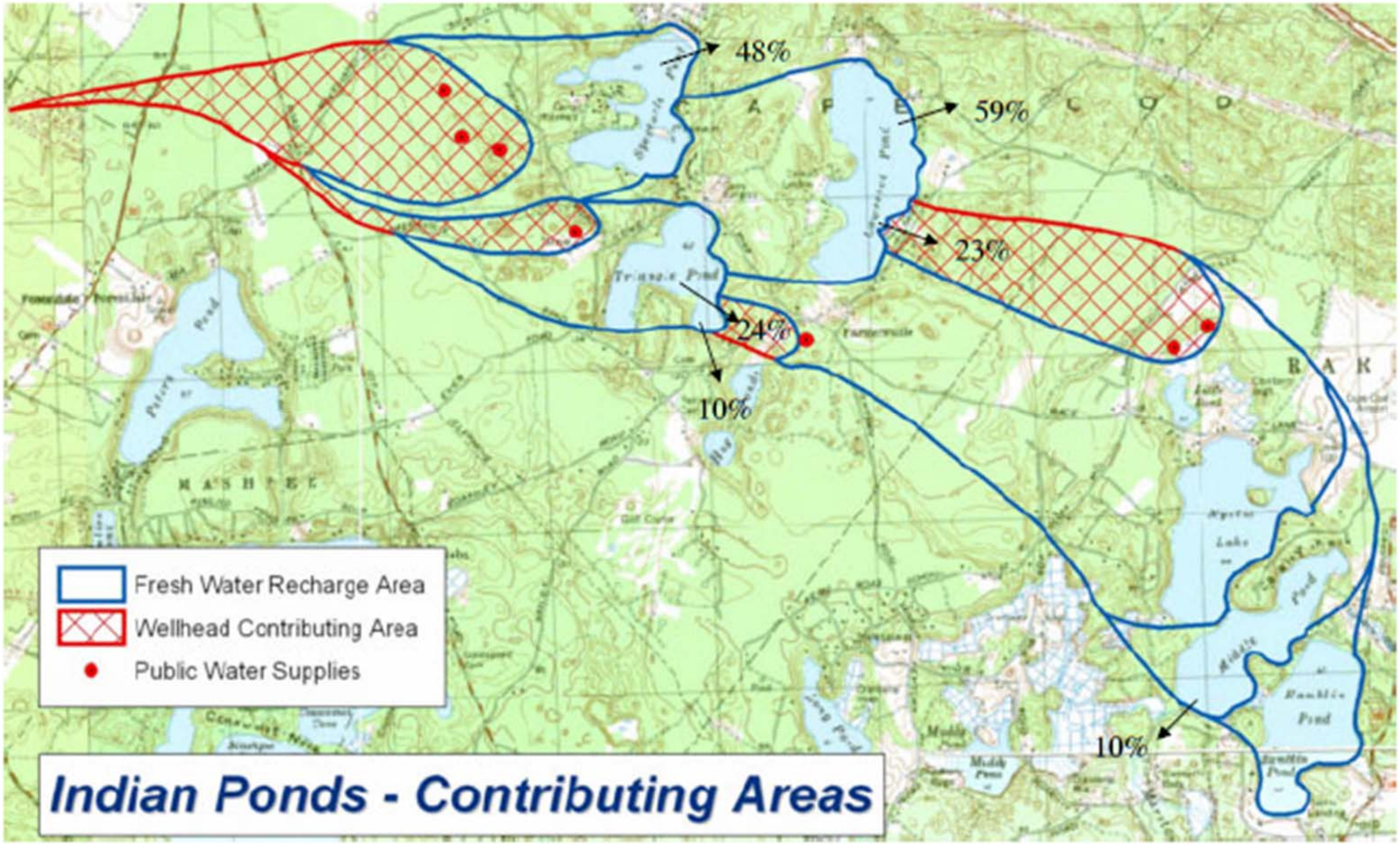


Figure 4. Immediate contributory land area around Mystic Lake. (from Eichner et al. 2006)

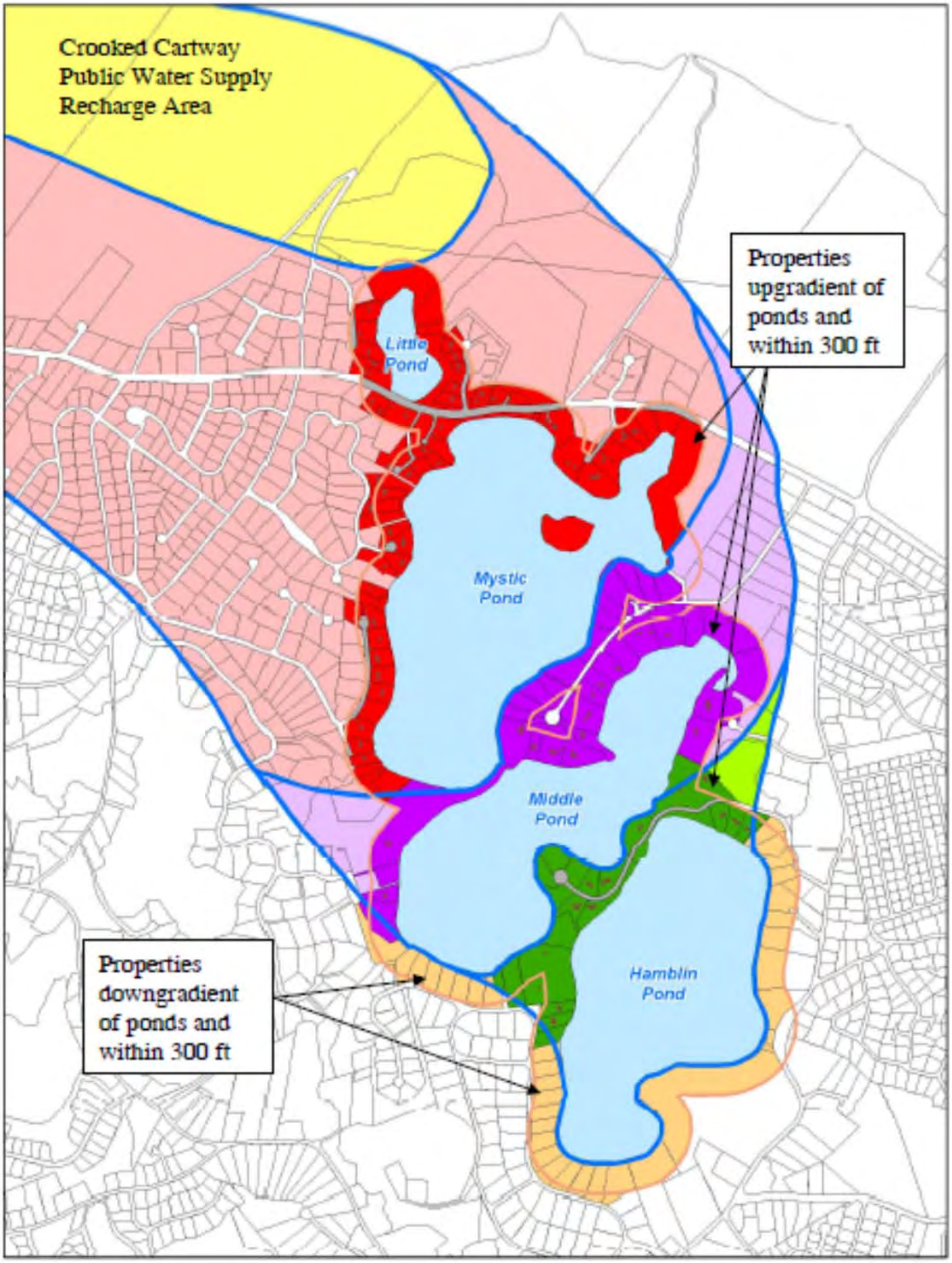


Figure 5. Aerial view of the immediate watershed of Mystic Lake.

Aside from some direct runoff during major storms in areas of steeper slope, the only surface water discharge to the lake is from cranberry bogs. There are bogs to the north and south (Figure 5). The bog to the north covers about 10 acres and has been managed by various parties over time, was not active in 2010, but resumed operation in 2011. Water can be withdrawn from or discharged to Mystic Lake from the northern bog. The very small bog just south of the lake is part of a larger complex of bogs. Water is apparently not withdrawn from the lake for irrigation of these bogs, but they use Mystic Lake water for flooding. However, only the small (about 1 acre) bog near the lake returns flood water to the lake; the remaining bog areas discharge to the Marstons Mills River downstream.

If 11 acres of cranberry bog were flooded to a depth of 1 ft, that would be 11 acre feet of return water, a very small amount relative to the roughly 2700 acre-feet of water in Mystic Lake. Phosphorus concentrations tend to be very high, averaging >200 ug/L and often in the 500 ug/L range (unpublished UMASS Cranberry Station and WRS data). For Mystic Lake, this would equate to 2.7 to 6.9 kg of phosphorus per year delivered from cranberry bogs; this is not a large amount, and would be contributed after the summer season when blooms are prevalent, but it is not insignificant. Bogs contribute to the sediment nutrient reserves that fuel internal recycling, but are not a significant threat to water quality at the time of discharge. The Eichner et al. (2006) and AECOM (2009) reports concluded this as well, but neither report explicitly included cranberry bog operations in the phosphorus loading estimate.

Waste water from on-site disposal systems associated with residences in the watershed was estimated to contribute 8.4 kg/yr of phosphorus, while background ground water was estimated to contribute another 16.7 kg/yr. As the calculated waste water phosphorus input was only for dwellings within 300 ft of the lake, the background ground water load undoubtedly contains some waste water phosphorus, but together the subsurface load is about 21% of the total load as estimated by AECOM in 2009 (Table 2).

Table 2. Estimated phosphorus load to Mystic Lake (from AECOM 2009).

Input Source	TP Load (kg/yr)	% of TP
Atmospheric	15.0	13%
Internal Recycling	54.9	46%
Waterfowl	6.0	5%
Septic Systems	8.4	7%
Watershed GW Load	16.7	14%
Watershed Runoff Load	18.5	15%
Total	119.5	100%

Watershed runoff inputs were estimated from modeling exercises, and may be overestimated based on the lack of storm water drainage systems, but if taken to include cranberry bog inputs, are reasonable estimates. Bird inputs are based on literature values for input per bird and an estimate of the number of birds at the lake on a continual basis. The AECOM report notes that the presence of alewife might attract more birds, including cormorants, and in recent years higher numbers of cormorants have been observed. The associated input estimate (Table 2) is probably low, but not far from accurate.

The atmospheric input calculation is based on precipitation and phosphorus levels measured in other studies from this general area, and is believed to be a reasonable approximation in the absence of direct measurement. This leaves only internal loading, which was calculated from actual phosphorus accumulation and corroborated with rates and loads from other studies. It is a reasonable estimate, but subject to considerable variability based on weather patterns and biological interactions. It should be kept in mind that much of the external load is scattered over the entire year and includes forms of phosphorus that are not immediately available for uptake by algae and plants in the lake. Yet the internal load occurs almost entirely in summer and is highly available, limited only by movement from deep water to the shallower zone, with such movement expected to be substantial. While the watershed is the ultimate source of phosphorus that has entered Mystic Lake, internal recycling processes now dominate phosphorus loading and have moved the lake from a desirable condition to one in which algal blooms and low clarity are common in summer.

Designated Uses

Mystic Lake is listed as Class B waters. Under the Massachusetts system, this means that the water is not intended for direct potable water supply, but is expected to meet water quality standards for recreational and habitat uses. The designated uses of Mystic Lake include swimming, boating, fishing, habitat for fish and wildlife, including support of an active alewife run, and water supply for cranberry bogs. No clear priority has been established, and it appears possible for all of these uses to co-exist with limited interference. However, having alewife in the pond minimizes zooplankton populations and associated grazing on algae, allowing the highest algal biomass supported by the fertility level of the pond. Return of water used to flood cranberry bogs to Mystic Lake represents nutrient inputs that add to that fertility, although the discharges are small relative to pond volume. Watershed activities of concern include storm water and waste water generation and routing, neither of which appears to be a large source at any instant in time, but the cumulative additions over many years help fuel internal loading.

Use Impairment

Low clarity, algal blooms, and deep water anoxia affect all uses but boating. Use of the water in cranberry bogs is not functionally impaired, but the image of berries picked in water with cyanobacterial blooms is not good for marketing. Mystic Lake experiences frequent but not continual algal blooms, including blooms of cyanobacteria in summer, green algae in summer and fall, and diatoms in winter and spring. Mystic Lake is rendered unaesthetic by algal blooms, and much of the algal production winds up in the sediment, both impairing oxygen through decay and limiting the opportunity to fuel productivity of desirable fish.

There has been no toxicity testing of water from Mystic Lake, and hazardous levels of toxins in lakes are actually fairly rare in occurrence (Lindon and Heiskary 2009, Graham and Jones 2009, Bigham et al. 2009), but the threat exists and is most commonly associated with high concentrations of cyanobacteria that coincide with surface scum formation or wind-blown shoreline accumulations. Both of these occur in Mystic Lake. There was also a major die off of mussels in the lake that coincided with a cyanobacterial bloom in 2009, with additional mortality coinciding with another cyanobacterial bloom in 2010. While mussel die offs have been attributed to algal toxins in salt water environments (Curtis et al. 2008), this is not a well-studied phenomenon in fresh water. Tissue samples collected in 2010 did not reveal any microcystin, but other toxins were more likely to be responsible and the timing of sampling was not ideal for detecting toxic impacts.

Key Relationships and Causative Agents

Mystic Lake does not suffer from turbidity induced by resuspended sediments; the shallow area (<15 ft deep) is quite sandy and concentrations of non-algal particulates in the water column are low. While the arrival of hydrilla signals a major threat to lake uses, rapid response by the IPA and Town of Barnstable have minimized its impact to date; rooted plants are not the cause of any use impairment at Mystic Lake. Abundant algae are the primary issue, linked to elevated phosphorus that is linked to low oxygen and large reserves of iron-bound phosphorus in muck sediments under >20 ft of water. Based on an AECOM survey, about 43% of the area covered by Mystic Lake has muck sediment covering the bottom. Testing in 2007 revealed an average of over 600 mg/kg of iron-bound (potentially available) phosphorus in that muck, representing between 16 and 32 g P/m^2. That is a minimum of more than 4000 kg of potentially available phosphorus on the bottom of Mystic Lake, when the quantity of phosphorus in the water column at any one time is only about 70 kg. The transfer of phosphorus from the sediments into the water column is a therefore critical process.

Transfer can occur when oxygen is low at the sediment-water interface. Dissolved oxygen profiles illustrate that anoxia below 9 m of depth is a common occurrence in the summer (Figure 6). Even back in 1948 there was low oxygen in deep water, but the situation has worsened considerably in recent years, with anoxia extending as shallow as 5 m in 2009 and 7 m in 2010. Oxygen demand is a combined function of decaying material in the water column and in the sediment, and algal blooms add to both.

Surface water temperature has increased in recent years, and there are irregularities in the upper water temperature pattern (Figure 7). Warmer waters favor cyanobacteria (Paerl and Huisman 2009), and less sharp thermal gradients facilitate upward transport of phosphorus. Temperature is largely a function of weather pattern, and climate change is moving many lakes toward warmer surface temperatures.

The relationship between phosphorus and either algal chlorophyll or water clarity is well known for lakes in which phosphorus is a limiting factor (Mattson et al. 2004, Cooke et al. 2005), and this relationship holds true for Mystic Lake (Figures 8 and 9). There is considerable variability, much relating to the types of algae, particle sizes, and phosphorus availability, but the pattern is clear; more phosphorus leads to more algae.

Figure 6. Dissolved oxygen profiles over time in Mystic Lake.

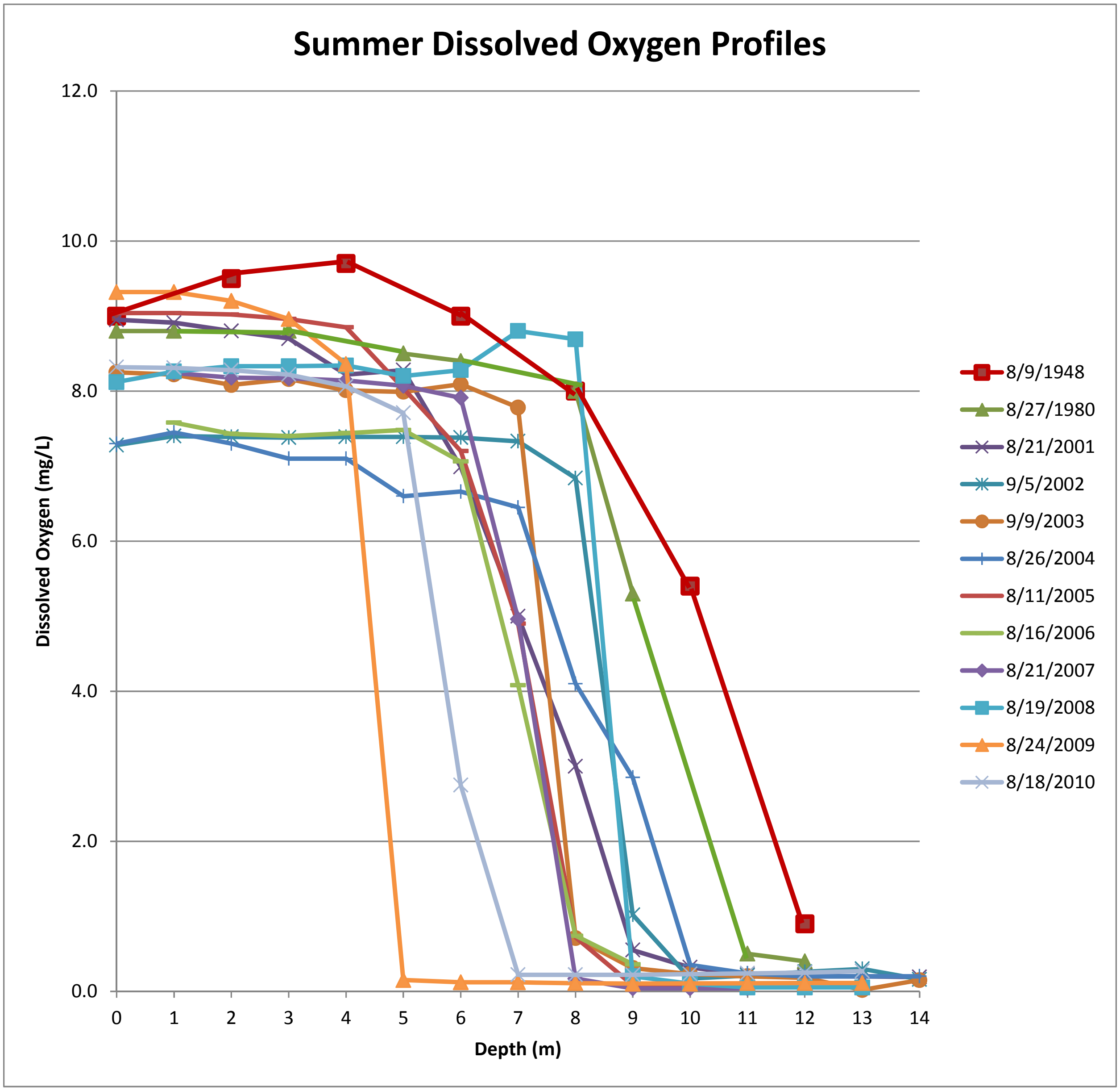


Figure 7. Temperature profiles over time in Mystic Lake.

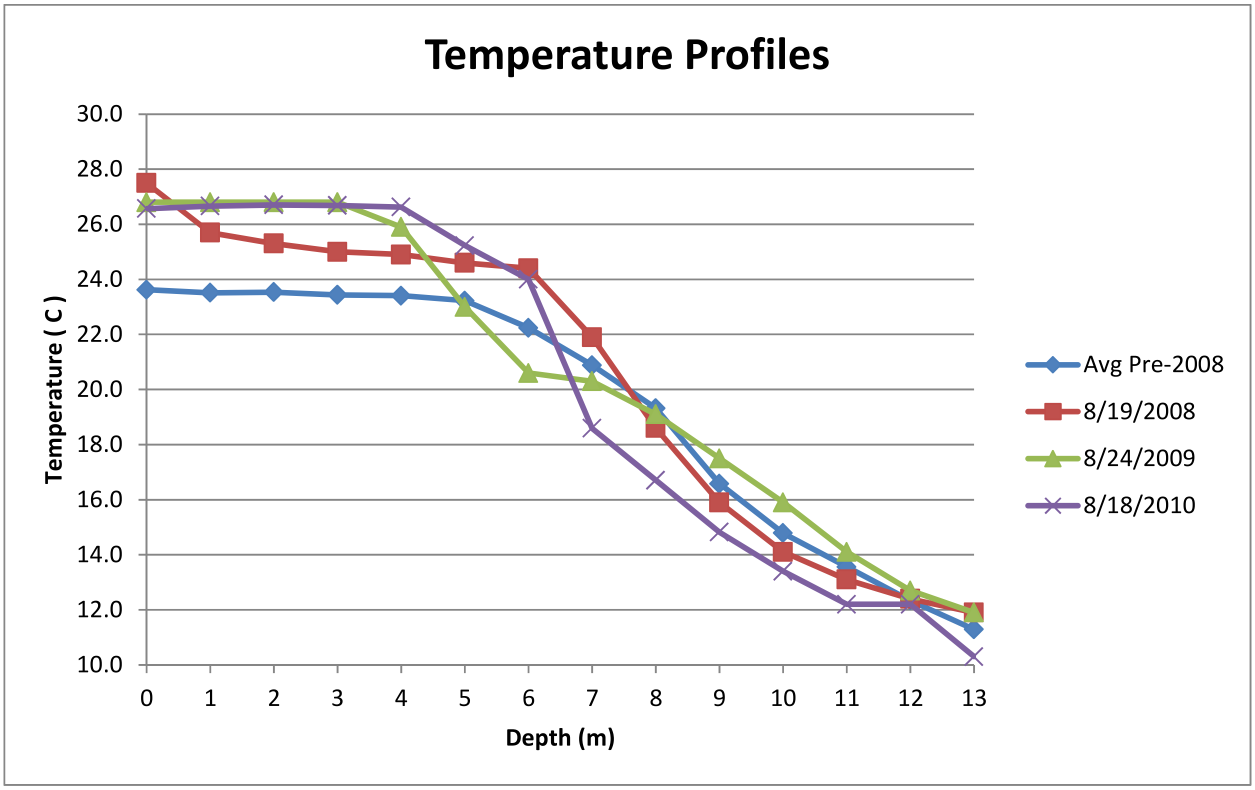


Figure 8. The relationship between phosphorus and chlorophyll-a for Mystic Lake.

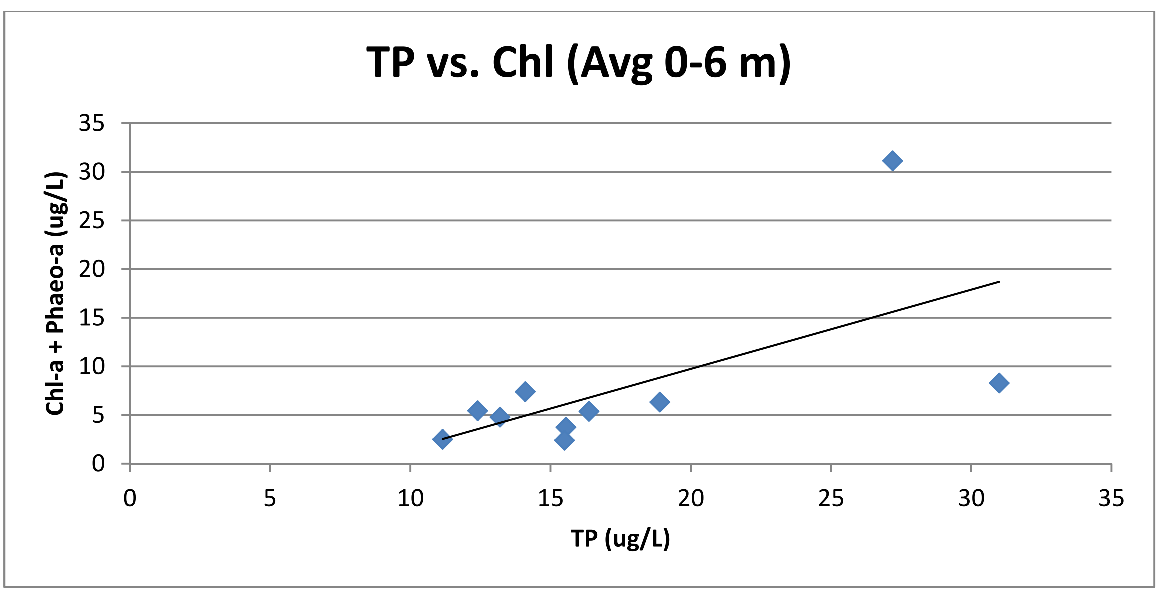
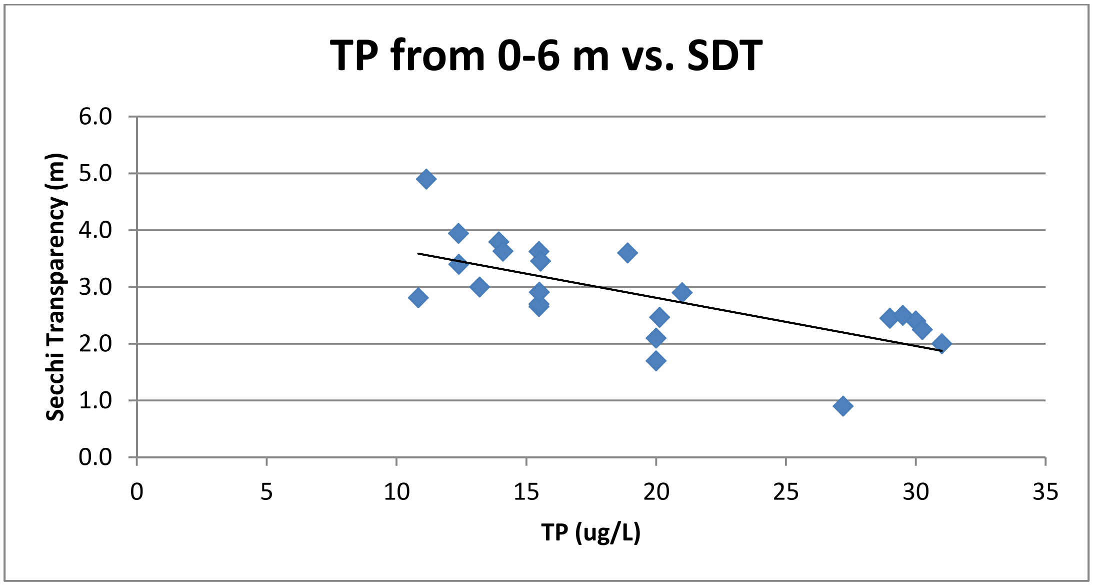
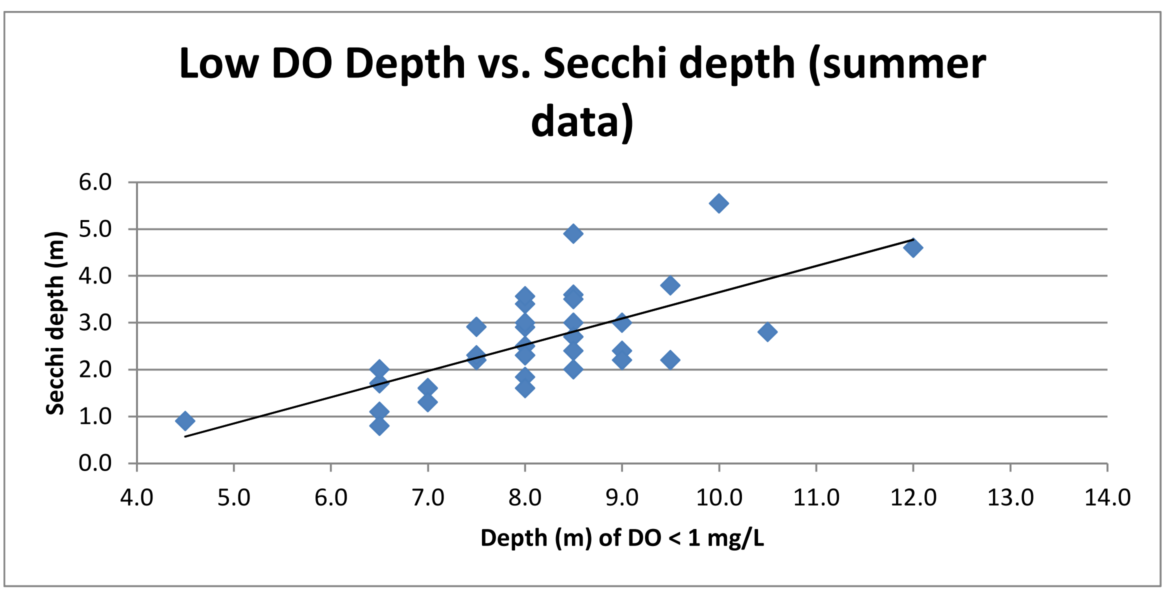


Figure 9. The relationship between phosphorus and water clarity for Mystic Lake.



As clarity is linked to algal biomass, which is linked to phosphorus, which is linked to oxygen, it seemed relevant to determine if the linkage would hold true all the way from clarity to oxygen. Based on comparison of the depth at which anoxia is observed in late summer and corresponding Secchi disk measures of water clarity, the relationship does indeed hold up (Figure 10). When the lake stratifies at a shallower depth, more bottom area is subjected to low oxygen and possible phosphorus release, there is more volume in the lower water layer to absorb that phosphorus (enhancing the concentration gradient and encouraging more release), more phosphorus is expected to reach the upper waters, more algae are produced (either in the upper waters or at the water layer boundary with subsequent upward movement), and water clarity declines.

Figure 10. Relationship between depth at which anoxia is observed and water clarity.



Nitrogen will generally not determine the quantity of algae present, although there is scientific debate over the role of nitrogen in algal dynamics (Welch 2010, Paerl et al. 2011). However, nitrogen is an important determinant of the types of algae that will be present, and the nitrogen to phosphorus (N:P) ratio in a lake provides useful insight into algae likely to dominate blooms (Smith 1983). N:P ratios under about 15, and certainly lower than 10, will favor cyanobacteria, particularly those able to utilize dissolved nitrogen gas, but cyanophytes in general are associated with lower N:P ratios. Most algae require nitrogen as nitrate or ammonium, and these can be scarce in summer in kettlehole lakes. High N:P ratios favor green algae in warmer waters and diatoms in colder waters.

Mystic Lake N:P ratios are high in surface waters and low in deep water most of the summer (Figure 11), with greater concentrations of both phosphorus and nitrogen in the deep water, but disproportionately more phosphorus relative to surface waters, lowering the deep water N:P ratio. This is typical for lakes experiencing high internal loading, and most phosphorus in the deep waters will be in a dissolved, available form, while most nitrogen will be ammonium released by decay but unable to convert to nitrite and then nitrate due to the lack of oxygen. N:P ratios in the mid-depth range (7-11 m) are variable, but was abnormally high in 2009 when the boundary between the upper and lower water layers was so shallow and cyanobacteria formed a bloom higher in the water column. The surface waters of Mystic Lake have moderate to high N:P ratios, but this does not prevent deeper cyanobacteria bloom formation with movement into the upper waters.

This pattern suggests that cyanobacteria will form blooms in the deeper water if light is adequate, near the boundary between water layers, and are likely responsible for the observed oxygen bulges observed near the thermocline in June and early July of many years. Whether that deep bloom reaches the surface waters is most likely determined by the weather, but we have limited chlorophyll or other algal data by which to track this phenomenon. The build-up of deep water chlorophyll at low N:P ratio is apparent in 2001 through 2005 (Figures 11, 12 and 13). Incomplete sampling over depth in 2006 and 2007 limit assessment in those years. In 2008 the boundary between upper (oxic) and lower (anoxic) water levels was as deep as it had been since 1980 (Figure 6) and the bulge in oxygen at the boundary was as large as ever observed, with a corresponding chlorophyll-a concentration that was the highest observed to date (Figure 12). Yet the associated algae did not rise to the top during summer and the layer was too deep to be mixed by wind, so surface water clarity was among the highest observed for Mystic Lake in summer 2008. In contrast, in 2009 that boundary was a full 4 m higher in the water column and the algae did move into the upper waters at very high chlorophyll-a levels.

As the water layers sampled have different thicknesses, putting the values on a volume weighted scale is helpful in visualizing the mass of algae involved (Figure 13); while a lack of deep water chlorophyll data for 2009 limits assessment, it appears that there was more algae in 2008 than any other year on record, but that it remained in deeper water. There were a lot of algae in 2009, and much of it moved into shallower water, probably as a function of wind mixing with such a shallow boundary depth during the wet, windy summer of 2009. The pattern over depth among years (Figure 14) suggests that the deeper algal community is the primary source of higher algal abundance in the surface water, mediated by weather. The influence of weather must be kept in mind when considering the variability that is observed and how management actions might change conditions.

Figure 11. Nitrogen:Phosphorus ratios over time and space in Mystic Lake.

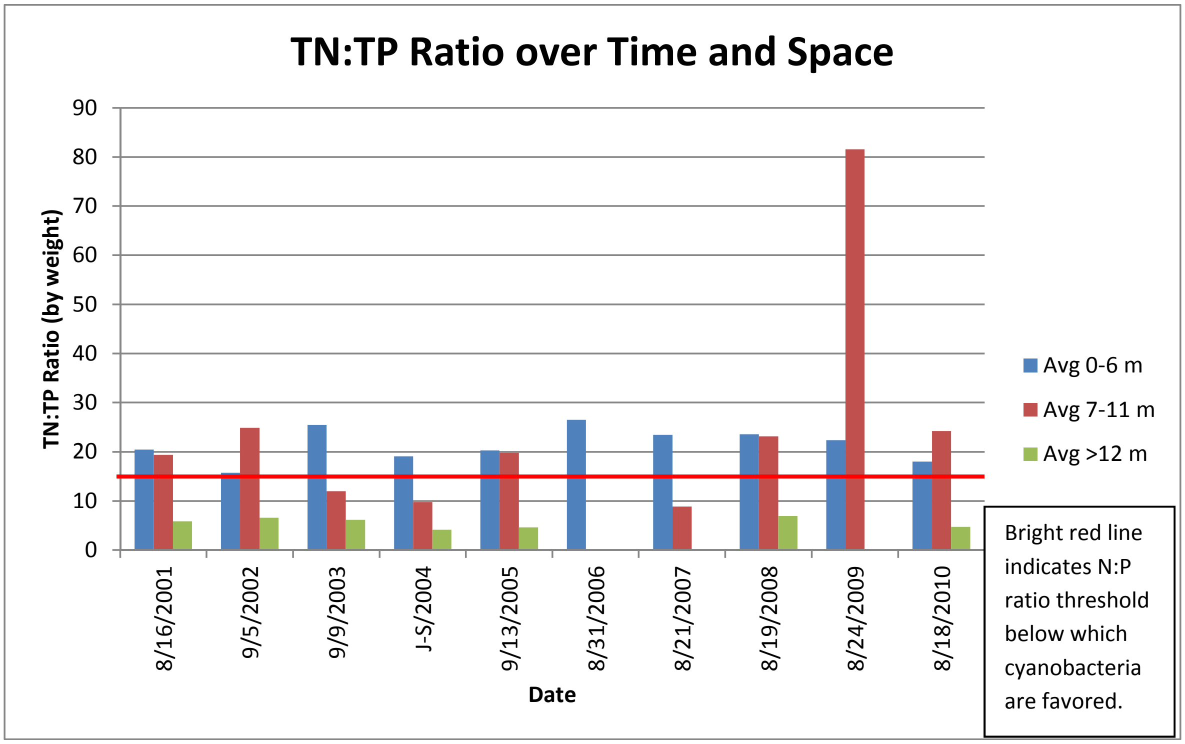


Figure 12. Concentrations of chlorophyll-a over space and time in Mystic Lake

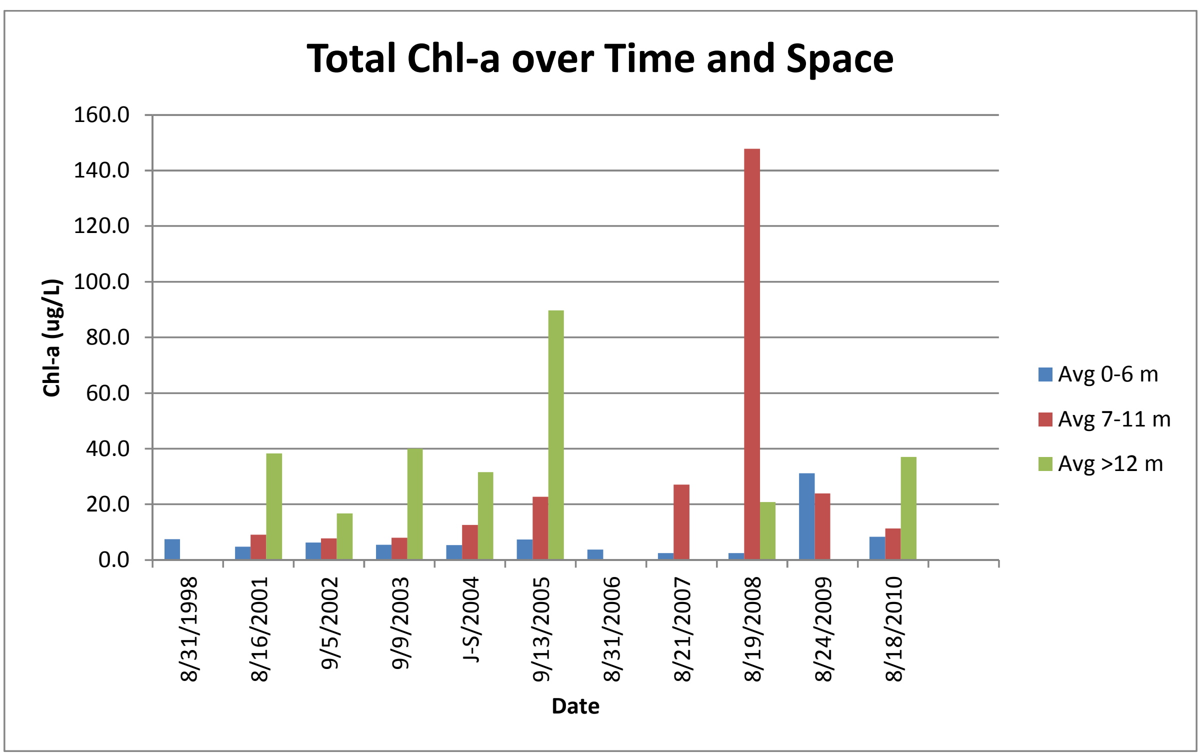


Figure 13. Volume weighted chlorophyll-a concentrations in Mystic Lake.

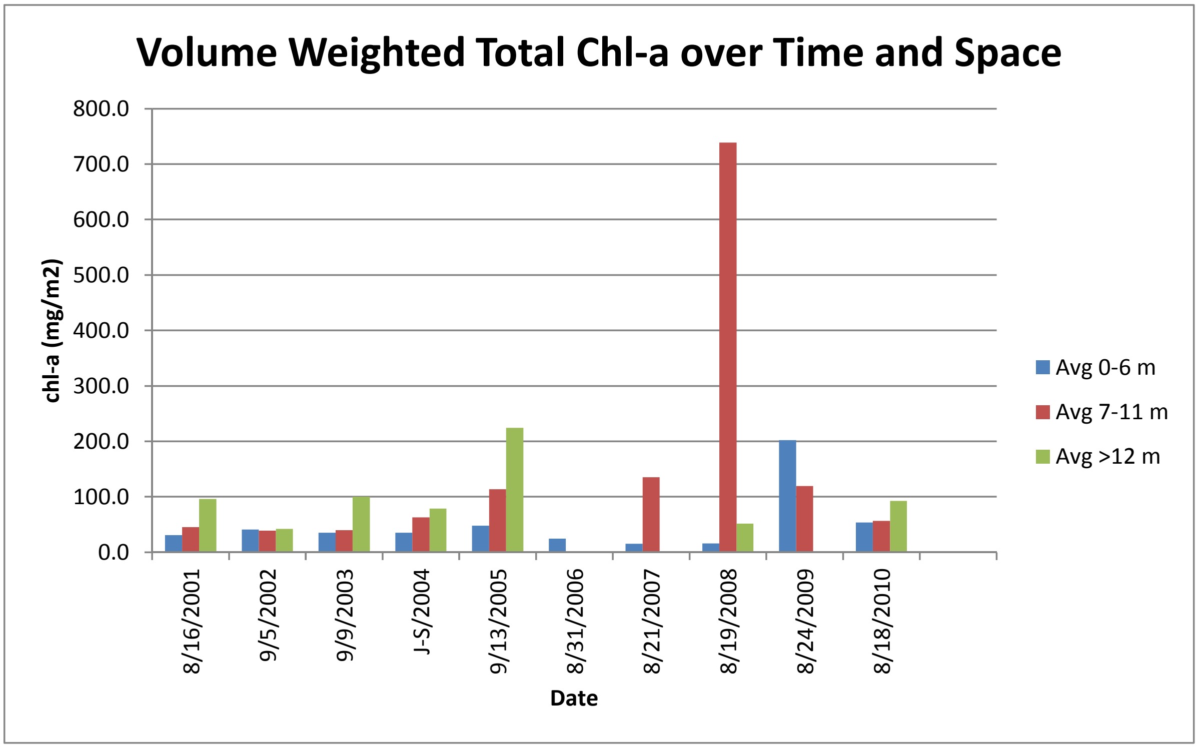
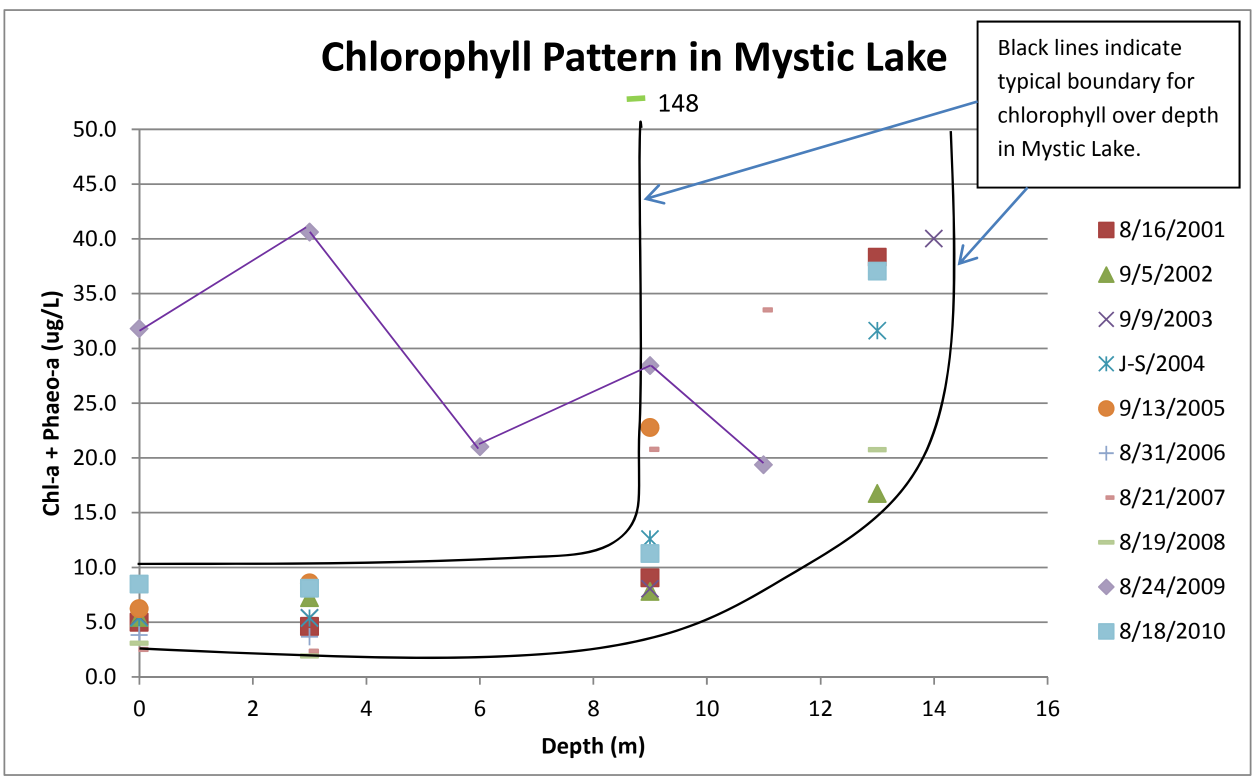


Figure 14. Pattern of chlorophyll over space and time in Mystic Lake.



Rehabilitation Needs and Objectives

Improving water clarity is the most apparent objective, and requires algal biomass reduction, which is best achieved by phosphorus reduction. The average summer chlorophyll *a* concentration for surface waters in the lake averaged 7.7 ug/L from 1998 through 2010, a moderate value, but higher values are often observed. Much higher concentrations have been observed in deeper water during thermally stratified conditions (Figures 12-14), and it appears that this deeper algal community is the source for many of the high chlorophyll values at the surface through mixing or active upward movement of the algae. Reducing chlorophyll levels to <4 ug/L in surface water would be desirable, preferably with a shift in the algal community away from cyanobacteria, and this may require addressing deeper water algal levels.

For most contact uses, phosphorus of <10 ug/L will yield an acceptable average concentration of algae, and will minimize the probability of algal blooms, usually defined as chlorophyll *a* levels in excess of 10, 15 or 20 ug/L; there is no strict definition of a bloom, but low clarity is rarely a problem at phosphorus levels <10 ug/L. Based on studies of many Cape Cod ponds, the Cape Cod Commission has suggested a target of 10 ug/L (CCC 2003) as appropriate for most Cape ponds, which is entirely consistent with ecological understanding of most lakes. This relationship is complicated, however, when a lake is stratified and algae may form a layer in deeper water, where the phosphorus concentration may be considerably higher than in the surface waters. Additionally, where fish production is an important goal, somewhat higher phosphorus levels are desirable, and can be maintained without excessive problems with algae at phosphorus levels <25 ug/L. This depends on having a desirable biological structure that supports a large population of large-bodied zooplankton (especially *Daphnia*) to graze on the algae, and that is difficult to maintain in ponds with alewife runs. It is expected that Mystic Lake will have algal abundance near the upper end of the plausible range for whatever level of fertility it has; to get clear water, the phosphorus level will have to be very low.

A phosphorus target close to 10 ug/L would therefore be desirable, favoring the level of clarity desired by most lake users. The summer phosphorus level averaged 18 ug/L in the surface waters of Mystic Lake from 2001 through 2010 and was considerably higher in water more than 20 ft (6 m) deep (average = 36 ug/L for 7-11 m and 574 ug/L at >12 m). In 2008, with the best clarity observed in a decade, surface phosphorus was 11 ug/L, while in 2009 and 2010 it was relatively steady and averaged 29 ug/L. It appears that the target of 10 ug/L is appropriate here, although if actual surface water phosphorus levels are just slightly higher, clarity will probably be acceptable. Phosphorus and algae come together to the upper water layer, and clarity correlates fairly well with each (Figures 8 and 9). The correlation of clarity with depth at which oxygen drops below 1 mg/L (Figure 10) appears to relate to the ease with which algae and phosphorus can move into the upper waters. Ensuring high oxygen to a greater depth would be one option for management, while simply reducing available phosphorus would be another.

The corresponding Secchi transparency for a phosphorus level of 10 ug/L is between 4 and 5 m, with an average chlorophyll level of 3 ug/L and peaks just over 10 ug/L. There will be variation in these values, but as averages, they would represent very acceptable conditions in Mystic Lake for all uses. Fish productivity might be lowered to some degree, but it is not at all rational to assume that high recent levels of algae, primarily cyanobacteria, are beneficial. That assumption delayed the treatment that is the main subject of this report and appears to have resulted in a major die off of valued mussels. The impact on the fishery is unknown, but as the natural background fertility of Cape ponds is believed to be low (CCC 2003), the 10 ug/L phosphorus target is considered to represent true restoration.

For comparison, the average surface water phosphorus level in Long Pond in Brewster and Harwich since the aluminum treatment in 2007 is just under 20 ug/L, and this has provided acceptable water clarity (summer average since treatment = 4.7 m) while still supporting fishing uses quite well. Clarity did not increase to the level observed in Hamblin Pond, however, where post-treatment Secchi transparency has averaged close to 6 m, and deep water oxygen in Long Pond did not improve to the level observed in Hamblin Pond either. Hamblin Pond was treated with aluminum to inactivate available sediment phosphorus reserves in 1995 and has exhibited both excellent water clarity and supported the best trout fishing on Cape Cod since then. Based on PALS data, surface water phosphorus concentrations have been fairly stable and have averaged <10 ug/L over the last decade, while the deep water level has averaged 61 ug/L with peaks around 200 ug/L. The depth at which oxygen drops below 1 mg/L was greatly altered by the treatment, having been <8 m prior to treatment and routinely deeper than 12 m since treatment. No oxygen was added, but the reduced algal production decreased the oxygen demand, allowing oxygen to remain higher for longer in the summer. There is still anoxia in deep water in Hamblin Pond, and some release of phosphorus still occurs, but the interaction with surface waters has been greatly reduced.

Rehabilitation objectives for Mystic Lake can therefore be summarized as control of phosphorus to yield lower algal abundance and higher water clarity, with some expectation of less oxygen demand in deeper water as a result. The target value for surface water phosphorus is set at 10 ug/L, although slightly higher values (no more than 20 ug/L) are expected to provide generally desirable conditions. It is also desirable to have the depth at which oxygen drops below 1 mg/L be at least 8 m below the surface of the lake; this happens naturally in some years as a function of weather, but could be achieved by management actions on a more regular basis.

Loading analysis by AECOM (2009) indicated that a phosphorus load of 79.5 kg/yr would be expected to yield the desired phosphorus level of 10 ug/L in surface water. This was considered to require a one third reduction in current loading. The difficulty of reducing loads from atmospheric sources, background ground water, and waterfowl was noted. Waste water sources and storm water runoff were noted as offering some reduction potential, but both AECOM (2009) and Eichner et al. (2006) concluded that the internal load was the logical primary target of management. Addressing inputs from on-site waste water disposal systems and storm water runoff was recommended to prolong the benefits of internal load control, but it took decades for the internal load to build to its currently deleterious level, and control of internal loading would be expected to provide extended benefits even without watershed management.

Review of Management Options

Both the Eichner et al. 2006 and AECOM 2009 reviewed management actions that were applicable to Mystic Lake. Both noted the need to address watershed loading, but both concluded that reduction of the internal load was necessary to achieve the desired conditions. It is often difficult for people focused on source control and watershed management to grasp the significance of the internal load, but this has been documented as a major force in many lakes, one that cannot be reversed quickly by watershed management. Yet internal load, when dominant, can be controlled with extended benefits (Mattson et al. 2004, Cooke et al. 2005, NYSFOLA 2009).

There are three well documented ways to reduce internal loading of phosphorus: remove the sediment which harbors the available phosphorus, inactivate the phosphorus in place, or provide enough oxygen to prevent phosphorus from being released to surface waters. Removing the phosphorus involves dredging, which is a truly restorative technique, but extremely expensive and difficult to permit in Massachusetts. Average levels of many contaminants in Massachusetts lake sediments exceed unrestricted disposal thresholds, so extensive testing is needed and in more than half the possible dredging cases disposal becomes complicated, further increasing costs. Adding the technical difficulty of dredging in water more than 25 ft deep, this is not an option that is normally even considered for deeper lakes with any history of anthropogenic inputs and possible sediment contamination.

Inactivation could be accomplished with addition of oxygen if natural phosphorus binders are present in adequate supply. The most common phosphorus binder by far for Cape ponds is iron, and that is what nearly all available phosphorus is bound to in Mystic Lake. Iron bound phosphorus represented almost half of the total sediment phosphorus in the AECOM samples from muck sediments, and iron concentrations were over ten times the total phosphorus concentrations. Phosphorus not bound to iron is largely in organic forms, some of which may decay and release that phosphorus, but very slowly. However, under anoxic conditions iron and phosphorus tend to resolubilize and increase in the overlying water column. By keeping oxygen levels high, the phosphorus stays bound to iron in insoluble compounds. Phosphorus released from organic compounds is likely to be bound by iron fairly quickly where oxygen is adequate. Even if oxygenation is not extended to the sediment-water interface, presence of enough oxygen below the boundary between lower and upper water layers during stratification can cause the iron and phosphorus to recombine and settle downward again. Creation of an oxygenated boundary layer can be achieved anywhere between the sediment-water interface and the bottom of the upper water layer, based on controlling water temperature to create a stable density gradient. The entire lake can also be kept in a mixed condition, circulating oxygen rich water from top to bottom.

Oxygenating all or part of the deeper water layer requires adequate input of oxygen by any of several energy intensive means from at least May through September every year, and success if often variable over space and time, such that internal load reductions are usually not as high as predicted by theory.

The additional benefits of more oxygen in deeper water include better habitat for fish and invertebrates and reduced concentrations of ammonium, sulfides, iron and manganese, with those reduced concentrations highly desired in water supply situations. For most recreational lakes, however, the ongoing expense and load control uncertainty associated with deep water aeration are cause for hesitation, and this approach is less often used for internal load control.

Inactivation by a binder other than iron has been practiced in water and waste water treatment for many decades, with calcium and aluminum most often applied. Calcium only precipitates at higher pH than experienced in a healthy Cape pond, so aluminum would be the binder of choice. Aluminum combines with phosphorus to form an insoluble floc between pH 6.0 and 8.0, settles to the bottom, and interacts with the sediment phosphorus in the upper few inches of sediment, preventing later release. Aluminum comes in several reactive forms, some causing the pH to decline and others causing it to rise, and a balanced addition of two aluminum compounds with opposite pH tendencies can maintain the pH at a desired level. Keeping the pH between 6.0 and 8.0, and preferably between 6.5 and 7.5, maximizes reaction efficiency and minimizes possible toxicity impacts of reactive aluminum (Mattson et al. 2004, Cooke et al. 2005). Once reacted, there is no significant threat of aluminum toxicity, but during the reaction process there is a risk to aquatic organisms. The treatment of Hamblin Pond in 1995 did not have a balanced mix of aluminum compounds, and while available sediment phosphorus was greatly reduced, there was substantial fish mortality during the treatment. A similar situation occurred in a CT lake in 2000, prompting research into causative agents, and avoiding mortality is now easily achieved.

Both the Eichner et al. (2006) and AECOM (2009) studies recommended inactivation of available sediment phosphorus in Mystic Lake with aluminum. Given what was known about Mystic Lake and the success of this approach in achieving clear water in Hamblin Pond, this was a logical management conclusion. The AECOM effort provided the documentation necessary to proceed to the permitting stage.

Project Planning and Permitting

ENSR International, which was absorbed by AECOM at the end of 2008, prepared a Notice of Intent for a proposed phosphorus inactivation project in October of 2008, even before the full report (AECOM 2009) was finalized. The permit was not authorized, however, as the Natural Heritage and Endangered Species Program (HHESP) objected to the treatment as a threat to the health of what was documented as an outstanding freshwater mussel community including seven species, three of them with protected status (Biodrawversity 2007, 2008). There were two expressed concerns: possible direct toxicity from aluminum during the treatment and reduced fertility to support the very abundant mussel community as a result of lower phosphorus and algae levels. AECOM and the Town of Barnstable countered those arguments with information about how aluminum treatments have evolved over time to minimize the probability of any toxicity, and literature regarding the threat of nutrient pollution and excessive algal abundance (e.g., Strayer et al. 2004), but NHESP felt that the risk to an outstanding mussel community was too great to allow the treatment.

At the core of the debate over possible impact of aluminum treatment of Mystic Lake was a lack of certainty regarding the response of mussel populations to increasing lake fertility and unwillingness by NHESP to let Mystic Lake be an experimental system for exploring that relationship. Impairment for human uses is not considered when making determinations under the Massachusetts Endangered Species Act, and there is a tendency for most regulatory agencies to assume that that status quo will remain in effect if no action is allowed. That assumption proved to be deeply flawed.

In August of 2009 the anoxic zone in Mystic Lake reached an apparent shallowness record of between 4 and 5 m (13 to 16 ft) and algae growing in a band near that level moved into the upper water column, creating very high chlorophyll levels in the surface waters (Figure 14). There was an extreme die off of mussels in Mystic Lake during August 2009; later assessment put the kill at close to 90% of the overall mussel community (Biodrawversity 2010), with greater loss of some species and less of others. There was little immediate sampling conducted, leaving the cause of the mortality to speculation. Only a few dead fish were observed, and the normal PALS monitoring on August 24, 2009 did not suggest any issues with oxygen or pH at that time, while mortality was still occurring. As the anoxic zone extended upward from the bottom to 4 to 5 m of water depth, and mussels were known to dwell at depths of up to about 8 m, some mortality due to extended lack of oxygen seems likely, yet some deep water mussels survived, suggesting other reasons for mortality. Millions of mussels died in very shallow water (Biodrawversity 2010), seemingly too shallow to have been a function of any of the normal mortality factors.

The leading hypothesis is that for at least the shallow mussel populations, algal toxicity was the cause of mortality. Chlorophyll-a levels in surface waters were the highest recorded there (Figures 12-14), and the algae that dominated the August 2009 bloom were *Aphanizomenon* and *Planktothrix*, both filamentous cyanobacteria known to be capable of producing neurotoxins. The mussels died more suddenly than would be expected with hepatotoxins, which have generally not proven toxic to mussels anyway. Live mussels collected by Mr. Robert Nichols of the IPA showed signs of severe lethargy, not contracting and closing their shells when picked up. When placed in well water in an aquarium, these mussels recovered, indicating that the problem was in the Mystic Lake water, at a time when oxygen, pH and other water quality features were acceptable for mussel survival. Additionally, mussel mortality was observed in Middle Pond, in the path of noticeably green water moving through the connector channel from Mystic Lake into Middle Pond. But no tests were run for neurotoxins at the time, and the cause of mortality remains unconfirmed.

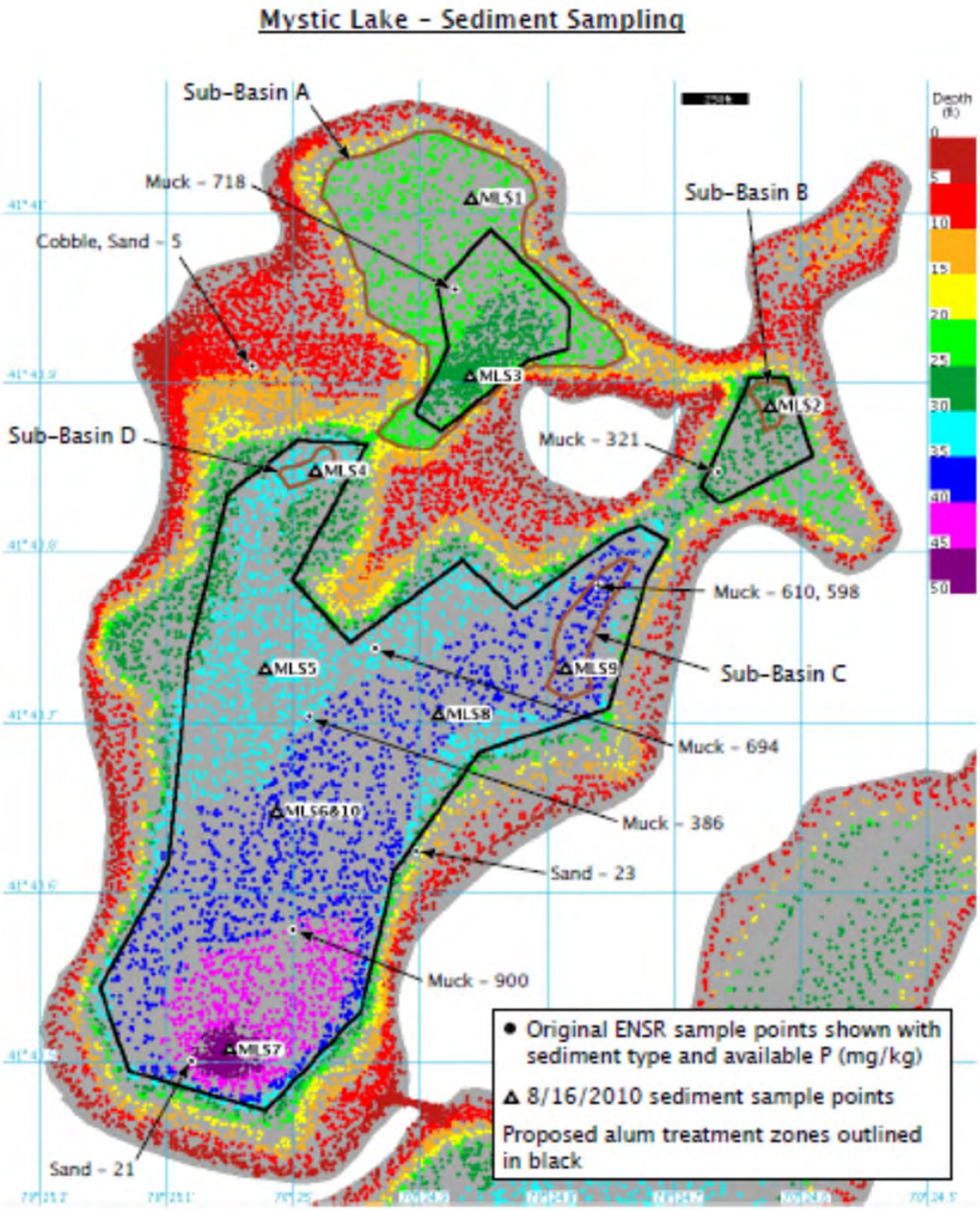
After consultation with the NHESP in the fall of 2009, reconsideration of the inactivation project was initiated. A revised Notice of Intent was filed, outlining the steps to be taken to further define the treatment, the precautions to be taken to avoid undesirable side effects, and the monitoring to be conducted before, during and after the treatment. The project was approved at all levels, with 32 special conditions imposed in the final Order of Conditions. Permit conditions included clauses relating to notifications and pre-treatment meetings, land based preparations, pre-treatment monitoring and final dose determination, treatment method, target area, and timing, and monitoring during and after treatment. Permit conditions also included thresholds for environmental conditions governing when treatment could not occur (e.g., wind and temperature limits) and when treatment would cease (e.g., pH outside of target range, fish or mussel mortality beyond established limits).

A late summer to early fall treatment was specified, to avoid potential issues with spawning alewife and release of glochidia (larval stages) by reproducing mussels, both of which are spring events. This was the approach applied at Long Pond in Brewster and Harwich, but has the disadvantage of treating after much phosphorus has been released into the water column by anoxic sediment during the summer; the reaction whereby aluminum binds phosphorus is not as efficient at the lower phosphorus levels in the water column as at the very high levels in the sediment (James 2011), and it can be expected that some phosphorus will remain in the water column that would have been inactivated by a spring treatment.

The permit specified a dose of 20 to 25 g/m^2, recommended by NHESP, with an option to increase to the 50 g/m^2 level recommended by AECOM if further testing showed that it was appropriate. Lab inactivation assays had indicated that 25 g/m^2 would be adequate to inactivate nearly all phosphorus in the composite sample tested in 2007, but the starting phosphorus level in that sample was only about half of the average concentration from seven samples in the target treatment zone. Sample from non-target zones containing sandy sediment with much lower phosphorus concentrations was apparently mixed with muck samples (from what became the target zone) in the lab to create the test composite, and a higher dose was believed to be needed in the target area of the lake to sufficiently lower available phosphorus.

Since three years had elapsed since sediment samples were analyzed and there was a question regarding the appropriate dose, WRS undertook a follow-up sediment assessment in August of 2010. Samples were collected from nine locations based on the expected target zones (Figure 15), with a randomly chosen duplicate sample for quality control. Note that MLS-1 was located in a shallower area where mussels had been abundant prior to the die off in 2009, as NHESP expressed interest in having an area treated where mussels might be affected. Most selected treatment areas are too deep for

Figure 15. Sediment testing locations for 2007 and 2010 with 2007 results.



significant mussel populations to have been present even before the die off. Additional die off was observed again in summer 2010, again with a mixed cyanobacterial bloom present, so there were relatively few mussels present to be affected anywhere in the lake, but mussels could be placed in a plot in this location prior to treatment. This additional sediment testing was considered necessary to determine the appropriate level of treatment, and followed the methods of Rydin and Welch (1998, 1999). Total, iron-bound and loosely sorbed phosphorus were measured from surficial sediment samples collected at the targeted locations, along with percent solids to allow calculation of the mass of available phosphorus that should be inactivated per unit area. In all New England sediments tested to date, iron bound phosphorus has represented nearly all of the available sediment phosphorus, and Mystic Lake was no exception.

The results of testing of the samples collected in August of 2010 by WRS (Table 3) indicated high available sediment phosphorus in all muck samples collected. There was spatial variability in the results, with the highest values at the north end of the lake just west of the island and in southern end of the lake, in the deepest part of it. Lowest values were at the northwestern end of the lake in the shallow zone chosen for mussel presence/suitability and in the northeastern end, east of the island. Available sediment phosphorus was almost entirely iron-bound phosphorus and represented a major fraction of the total phosphorus in each sample. The solids content of all samples was low, ranging from 11.9 to 15.1%, typical of loose, organic muck samples.

Quality control data by the lab indicated an acceptable level of precision. Combining all the duplicate or replicate analyses available for this project, the variability in iron bound phosphorus ranged from 0.2% to 28% with an average of 6%. Only one difference was >10%, and was observed for a split sample from an area with low available sediment phosphorus (slight differences at low levels yield higher percent differences). Spiked sample recovery was always >90% and blanks were below detection in each case. The data appear reliable.

The lab assay for inactivation of phosphorus by aluminum involves suspending an approximately 5 g sample of sediment in a small amount of distilled water, treating with the aluminum compounds to be used in the lake treatment, allowing the reaction to proceed, and retesting for available sediment phosphorus. It is not a perfect simulation of field treatments, but has proven useful in setting doses in the past (e.g., ENSR 2001a, ENSR 2001b), and appears more reliable than simply multiplying the targeted quantity of available sediment phosphorus by some empirical factor (10-100X, Rydin and Welch 1998, 1999). Treatments simulated doses of 20, 30, 40 and 50 g/m^2 (Table 4), covering the range of doses allowable under the permit issued. Results normally exhibit diminishing returns, with the smallest dose yielding the largest proportional decrease and larger doses yielding successively smaller decreases as the detection limit is approached; Mystic Lake sediments followed the expected trend.

In planning a treatment, one must decide at what point the extra cost and possible adverse environmental impacts outweigh the benefits of additional phosphorus inactivation, but there is no standard rule governing this process. Experience over the last decade has indicated that even high doses

Table 3. Data from 2010 sediment testing.

Station	MLS-1	MLS-2	MLS-3	MLS-4	MLS-5	MLS-6	MLS-7	MLS-8	MLS-9	MLS-10
% solids	14.6	15.1	14.8	15.1	14.1	12.1	14.4	12.2	12.2	11.9
Total Phosphorus (mg/kg dry weight)	643	628	1890	787	820	1010	1410	1090	1000	1020
Iron-bound Phosphorus (mg/kg dry weight)	170	175	1280	300	243	428	1090	518	445	495
Loosely sorbed Phosphorus (mg/kg dry weight)	<3.4	<3.3	<3.4	<3.3	<3.6	<4.1	<3.5	<4.1	<4.1	<4.2
Available Sediment Phosphorus (mg/kg dry weight) after Treatment with 20 g/m2	20.5	31.7	346.0	42.4	29.0	66.6	142.0	76.8	70.3	46.8
Available Sediment Phosphorus (mg/kg dry weight) after Treatment with 30 g/m2	17.3	17.5	234.0	29.9	28.3	36.1	87.6	44.4	31.0	31.2
Available Sediment Phosphorus (mg/kg dry weight) after Treatment with 40 g/m2	<17.2	<16.5	120.0	<16.5	<17.8	<20.6	36.9	<20.5	27.2	25.8
Available Sediment Phosphorus (mg/kg dry weight) after Treatment with 50 g/m2	<17.2	<16.5	109.0	<16.5	<17.8	<20.6	52.5	21.6	<20.5	24.0
Preferred Dose Assignment (g/m2)	30	35	50+	40	40	40	50+	45	50	40

of aluminum can be administered without significant fish mortality, and that while some lateral drift is expected, invertebrate mortality outside the target zone is minimal. While the aluminum floc that forms can negatively impact invertebrates, invertebrate populations in typically anoxic target zones are not normally a concern; these tend to consist mainly of midge (Chironomidae) larvae and oligochaete worms. Relatively rapid recolonization, including additional, more desirable species, has been observed after aluminum treatments (Smeltzer et al. 1999, Cooke et al. 2005). The dose decision is therefore properly based on effectiveness and cost.

Using the data in Table 3, doses were selected for the target areas (Figure 16) based on apparent need and the permit limit of 50 g/m^2. The permit limit was based on the original preparatory work and a desire to minimize adverse environmental impacts, and for two areas (B and E) resulted in a dose less than what would have been prescribed based on the most recent analysis. Yet as the potential increase from an initially proposed maximum of 25 g/m^2 to 50 g/m^2 was gained only after considerable debate, re-opening the permitting process to increase the limit further was not pursued. The testing process suggested that 50 g/m^2 would provide a reasonable level of decrease in available sediment phosphorus, and this is very close to the dose administered to Hamblin Pond in 1995 with excellent results in terms of algae control and increased water clarity.

The dose calculation is viewed in more detail in Table 4. All available data for any defined target area were combined to get an average available sediment phosphorus value for each of those areas; this includes the 2007 AECOM results as well as the WRS data from 2010. There is variability among and within data sets, but the results are compatible and do not include any outliers for the defined target areas. Samples from sandy areas collected by AECOM were excluded, as these represent non-target areas. The range of mean available sediment phosphorus for the six identified treatment areas (Figure 16, Table 4) is 170 to 999 mg/kg. The mass of phosphorus targeted for inactivation is calculated based on either a 4 cm or 10 cm depth of sediment, the typical range applied. The treatment areas range from 2.3 to 20.3 acres, with a total area of 58.1 acres.

Based on the dose selected according to lab assays and permit limits, the range of ratios of aluminum to phosphorus for each treatment area is calculated, using either the 4 cm or 10 cm depth. These ratios range from 11 to 27 for the 4 cm depth and 3 to 11 for the 10 cm depth, all at or below the low end of the range postulated for successful treatments (Rydin and Welch 1998, 1999, James 2011). With high available phosphorus levels, treatment is more efficient and the ratio can be lower, but ratios <10:1 are likely to leave a significant fraction of what is assessed as pre-treatment available sediment phosphorus unbound to aluminum. For Mystic Lake, the doses for areas B and E, which are limited by permit conditions, appear low. This does not mean that the treatment will fail, but it does suggest that less phosphorus will be bound that desired.

The targeted doses yield a range of aluminum being deposited in each area, with a total of 23,025 lbs (10,466 kg) of aluminum being applied over the 58.1 acres. Given the properties of the commercially available aluminum chemicals and the targeted ratio of aluminum sulfate to sodium aluminate(2:1) intended to yield minimal change in pH, the volumes of aluminum sulfate and sodium aluminate are calculated at 20,624 and 10,312 gallons, respectively.

Figure 16. Target areas and doses for the Mystic Lake phosphorus inactivation treatment.

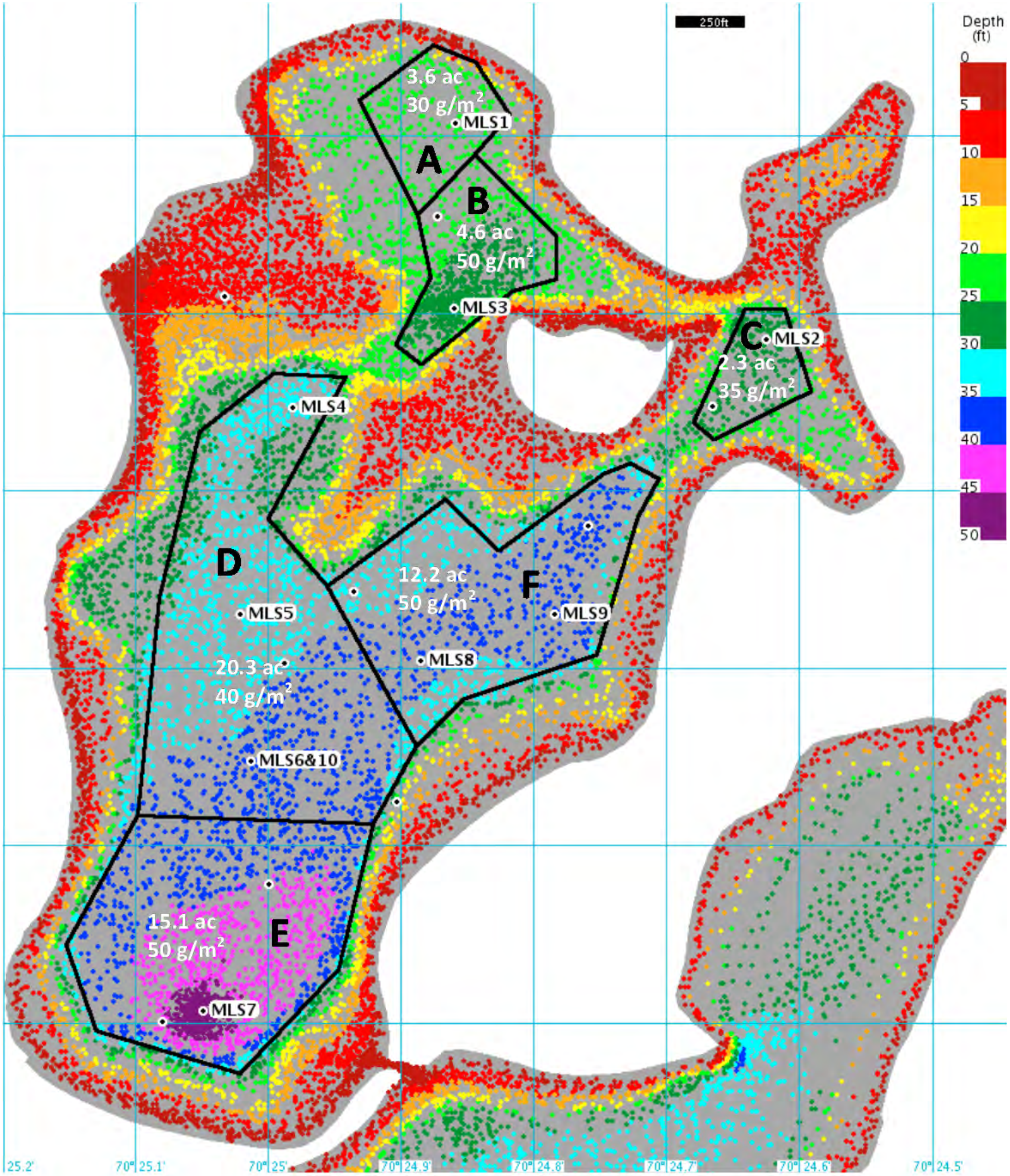


Table 4. Dose calculations for the Mystic Lake phosphorus inactivation treatment.

Designated Area	A	B	C	D	E	F	
WRS Sampling Stations	**MLS-1**	**MLS-3**	**MLS-2**	**MLS-4,5,6,10**	**MLS-7**	**MLS-8,9**	**Total**
Mean Available Sediment P (mg/kg DW) from AECOM + WRS samples	170	999	248	370	995	573	
Mass of P to be Treated (g/m2) assuming 4 cm depth	1.12	6.59	1.64	2.44	6.57	3.78	
Mass of P to be Treated (g/m2) assuming 10 cm depth	2.81	16.48	4.09	6.11	16.42	9.45	
Target Area (ac)	3.6	4.6	2.3	20.3	15.1	12.2	58.1
Target Area (m2)	14516	18548	9274	81855	60887	49194	234274
Areal dose (g Al/m2) based on lab assays and maximum dose limit	30	50	35	40	50	50	
Ratio of dose to P mass at 4 cm depth	27	8	21	16	8	13	
Ratio of dose to P mass at 10 cm depth	11	3	9	7	3	5	
Aluminum sulfate (alum) @ 11.1 lb/gal and 4.4% aluminum (lb/gal)	0.4884	0.4884	0.4884	0.4884	0.4884	0.4884	
Sodium aluminate (aluminate) @ 12.1 lb/gal and 10.38% aluminum (lb/gal)	1.256	1.256	1.256	1.256	1.256	1.256	
Ratio of alum to aluminate during treatment (volumetric)	2.00	2.00	2.00	2.00	2.00	2.00	
Aluminum Load							
Dose (kg/area)	435	927	325	3274	3044	2460	10466
Dose (lb/area)	958	2040	714	7203	6698	5411	23025
Dose (gal alum) @ specified ratio of Alum to Aluminate	858	1828	640	6452	5999	4847	20624
Dose (gal aluminate) @ specified ratio of Alum to Aluminate	429	914	320	3226	3000	2424	10312
Application (gal/ac) for Alum in Alum+Aluminate Trtmt	238	397	278	318	397	397	
Application (gal/ac) for Aluminate in Alum+Aluminate Trtmt	119	199	139	159	199	199	

While the dose decision is best based on a combination of effectiveness and cost, environmental concerns must be appropriately addressed in how the dose is applied. The primary concern is possible aluminum toxicity during treatment, as aluminum in a reactive state can be toxic if the pH is too low or too high. The desirable pH range is usually given as 6.0 to 8.0 SU (Mattson et al. 2004, Cooke et al. 2005), but bioassays have sometimes revealed potential toxicity at pH values between 7.5 and 8.0 as well, and the reaction between aluminum and phosphorus appears to be most efficient at a pH near 7.0 SU. Consequently, we aim for a pH close to 7.0 SU in most treatments, knowing that this will be both effective and minimally toxic.

For the planned treatment of Mystic Lake, bioassays were run with fish (fathead minnow, the standard lab bioassay fish species) at New England Bioassay, now a division of GZA Environmental. A 48 hour static bioassay was run using Mystic Lake water to which aluminum sulfate and sodium aluminate were added at a 2:1 ratio by volume to yield aluminum concentrations ranging from 2.78 to 16.67 mg Al/L. This range was estimated based on the target area depths, expected doses, and anticipated range of initial mixing. Five fish of one week in age were put in each of 5 replicate containers for each of seven aluminum treatment levels plus one control (untreated Mystic Lake water). Water chemistry, fish behavior and fish mortality were observed over 48 hours. It should be noted that the pH of the water delivered to the lab was 8.8 SU, elevated by algae blooming in Mystic Lake at the time. Also, the alkalinity was 15 mg/L, a high value for Cape Ponds but entirely consistent with field measurements on surface waters from Mystic Lake over the last few years.

The bioassay is intended to mimic treatment conditions and assess the potential response of fish to aluminum treatment. As lab conditions are not as variable as field conditions and only one species and life stage of fish is involved, it is not a complete simulation, but has proven useful in adjusting doses and minimizing mortality. No such assays were conducted before the treatment of Hamblin Pond in 1995, when substantial mortality resulted from elevated pH; the ratio of aluminum sulfate to sodium aluminate was only 1.4:1 for that treatment (Town of Barnstable unpublished data). Likewise, no bioassays were conducted before initial treatment of Lake Pocotopaug in 2000, which exhibited similarly elevated fish mortality. Subsequent bioassays allowed adjustment of that treatment, which proceeded in 2001 with no significant fish mortality (ENSR 2001b).

It is common to observe some behavioral stress and/or mortality in bioassays at aluminum >5 to 10 mg/L, or if the pH gets outside of the range of 6.0 to 8.0 SU. For Mystic Lake, however, there was no significant mortality; only one fish died in a mid-range aluminum treatment (5.56 mg Al/L) and one fish died in one of the control containers (NEB 2010), out of 200 fish involved in the bioassay effort. The pH never dropped below 7.3 SU, and was as high as 7.9 SU, but these values represented decreases from the starting pH of 8.8 SU, considered an improvement. No abnormal behavior of fish was observed during the bioassay. Toxicity is almost always observed within the first 24 hours of such bioassays if it is going to occur, and is not expected if not observed after 48 hours, as the aluminum reactions have largely been completed by that time. The relatively high alkalinity of Mystic Lake water apparently buffered pH changes and associated aluminum toxicity; it was concluded that there was minimal probability of fish toxicity at the planned doses in Mystic Lake.

One concern that was raised by the bioassay was that the background pH during treatment could be high. This would suggest that a lower ratio of aluminum sulfate to sodium aluminate would be appropriate, dropping the pH to around 7.0 SU for better reaction kinetics and minimal toxicity threat. This is why flexibility needs to be built into permits for such treatments. It was decided to put off any decision on changing chemical ratios until closer to the treatment time, when the pH might be different. As it turned out, the pH just prior to the start of treatment was 6.8 to 7.2 SU, a favorable range for the planned treatment with the aluminum sulfate to sodium aluminate ratio at 2:1.

Treatment and Monitoring Protocols

Pre-Treatment Monitoring

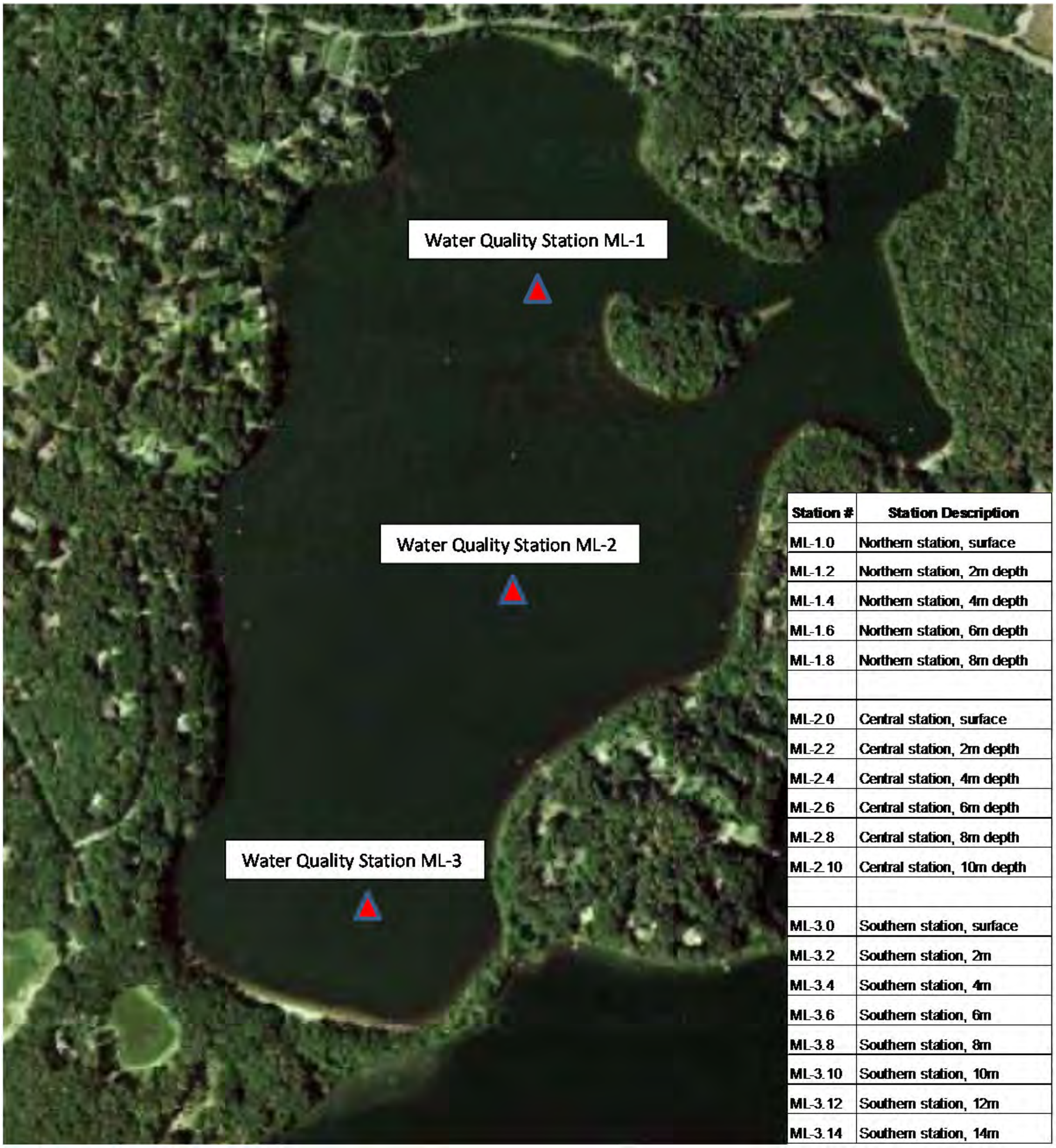
The permit conditions specified that water testing and a mussel survey be conducted prior to treatment. Both the water testing and the mussel evaluation would assess the background conditions for later comparison. As already presented in this report, there are many data points for water quality in Mystic Lake from the last decade, but additional testing before treatment is important for detecting any treatment issues, such as the potential high pH problem noted in the previous section of this report. It also provides more immediate pre-treatment conditions for comparison to conditions during and after treatment. Water quality was therefore assessed at 2 m intervals from surface to bottom at three locations in Mystic Lake (Figure 17) on September 13, 2010, about a week before planned treatment. Water quality sampling stations were set up to match sediment sampling stations MLS-3, MLS-7 and MLS-8 from Figures 15 and 16, and became the standard monitoring locations for follow up sampling.

Tested water quality variables included temperature, dissolved oxygen, pH, alkalinity, conductivity, total phosphorus, dissolved phosphorus, and dissolved aluminum, as specified in the Order of Conditions. All but phosphorus and aluminum were measured in the field with a Hydrolab DS5 multi-probe sonde that was calibrated prior to each use. Total and dissolved phosphorus were measured at Berkshire Enviro-Labs in Lee, MA by the same standard method (colorimetric after digestion), except that the dissolved phosphorus subsample was filtered through a 0.45 um glass fiber filter before testing. Secchi disk transparency was also measured at each of the three stations with a view tube to minimize the effect of any glare or waves, and phytoplankton and zooplankton samples were collected as well. Phytoplankton samples were whole water samples collected about 0.1 m below the lake surface, while zooplankton samples were obtained by pulling a 53 um mesh net through 30 m of vertical water column (multiple tows being needed since the water is not that deep), representing 948 L of water. Phytoplankton and zooplankton samples were preserved with gluteraldehyde, concentrated by settling in the WRS lab, and assessed quantitatively in counting chambers under phase contrast optics at 100 to 400X.

Mussel monitoring required submission of a monitoring plan for approval by NHESP, and the Order of Conditions outlined a number of testing and monitoring elements that were to occur. Some proved infeasible or unjustifiable, such as bioassays for mussels, as there is no standard protocol and a 48 hour assay is not long enough to assess impacts; these organisms can close their shells and avoid interaction with the surrounding water for multiple days. Likewise, the suggested caging of mussels in areas likely to have no oxygen or suspended in the water column would not provide the desired insight into mussel impacts from aluminum treatment, requiring all parties to rethink how to best assess mussel impacts.

ACT and WRS worked with Ethan Nedeau of Biodrawversity to develop a program that would be appropriate under the circumstances. Previous surveys of Mystic Lake had been conducted by E. Nedeau as part of NHESP assessments and in response to a permit application for a dock by a private party owning property at the lake and as a general survey of important mussel ponds in southeastern Massachusetts, so there was considerable background on mussel populations in Mystic Lake

Figure 17. Location of water quality monitoring stations in Mystic Lake.



Station #	Station Description
ML-1.0	Northern station, surface
ML-1.2	Northern station, 2m depth
ML-1.4	Northern station, 4m depth
ML-1.6	Northern station, 6m depth
ML-1.8	Northern station, 8m depth
ML-2.0	Central station, surface
ML-2.2	Central station, 2m depth
ML-2.4	Central station, 4m depth
ML-2.6	Central station, 6m depth
ML-2.8	Central station, 8m depth
ML-2.10	Central station, 10m depth
ML-3.0	Southern station, surface
ML-3.2	Southern station, 2m
ML-3.4	Southern station, 4m
ML-3.6	Southern station, 6m
ML-3.8	Southern station, 8m
ML-3.10	Southern station, 10m
ML-3.12	Southern station, 12m
ML-3.14	Southern station, 14m

(Biodrawversity 2007, 2008). However, the die offs of 2009 and 2010 complicated use of previous data as a baseline for comparison with post-treatment data relating to aluminum addition. A lakewide survey was conducted in June of 2010 by E. Nedeau to ascertain the level of mussel die off for NHESP, but additional die off in August 2010 limited the value of that assessment as a baseline as well. Clearly a very large portion of the mussel population was lost, and general surveys covering defined areas would not be completely adequate.

It was determined that it would be most appropriate to utilize defined plots for mussel assessment. Mussels could be placed in those plots if mortality in 2009 and 2010 had been too extreme, which it was, with prescribed numbers and types of mussels added as available. Plots could be monitored before, during and after treatment to assess any impacts from treatment, although continuing mortality from non-treatment causes might interfere with interpretation of results. The details of the program are described in a separate report by Biodrawversity (2011a), but the basic elements included establishment and "stocking" of pairs of mussel survey plots (Figure 18) with 35 mussels of three species (15 eastern elliptio from Mystic Lake plus 10 tidewater muckets and 10 eastern lampmussels transplanted from Middle Pond). There were 8 plots, 2 in an area directly treated, 2 in on edge of a treatment area and likely to receive some aluminum floc, and 4 in areas not expected to receive any additional aluminum.

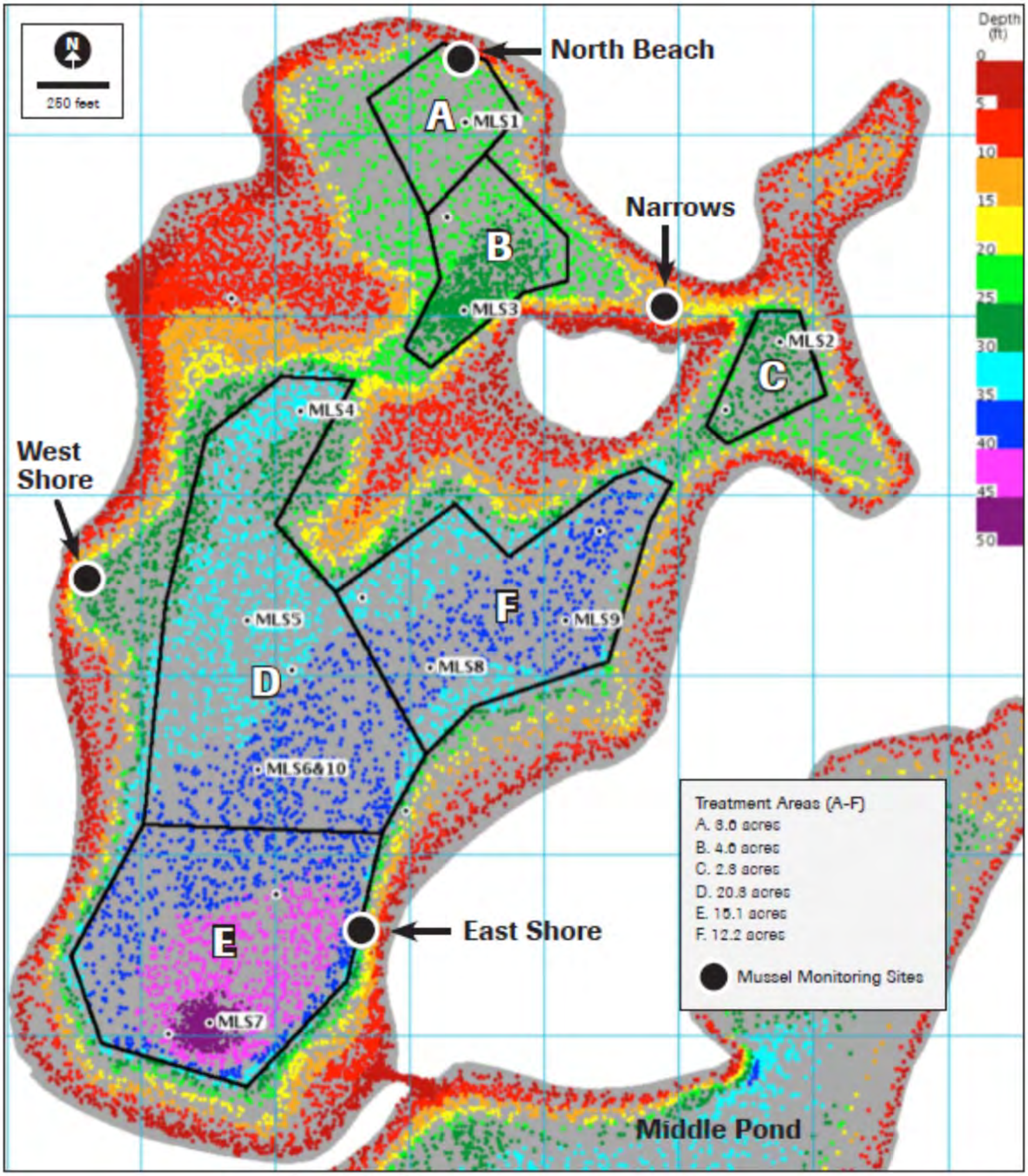
Assessment of physical (shell length and condition), behavioral (responsiveness to touch, embeddedness, filtering, movement) and mortality features potentially relevant to aluminum additions or other influences was conducted before treatment, shortly after the initial day of treatment (which included Area A in which two mussel plots were located), and several weeks after all treatment was complete. Treatment was extended into part of the northern area (Area A in Figures 16 and 18) that was shallower than would normally be treated, at the request of NHESP, to allow direct assessment of treatment impacts on mussels. Aluminum treatments usually target deeper water with anoxia that is not suitable as mussel habitat, but direct treatment was conducted to help define potential impacts.

An additional monitoring effort, not directly connected to the aluminum treatment, involved a lakewide survey in 2011 to allow comparison with 2007 and 2010 lakewide survey results. The 2007 survey represented conditions prior to any major die off of mussels, while the 2010 survey illustrated the massive die off of 2009. The 2011 survey illustrated any additional die off in 2010, from any possible source, including but not limited to aluminum treatments, with some mortality known to have occurred prior to aluminum addition. The 2011 survey could also have documented the start of any recovery, although continued mussel population decline was observed (Biodrawversity 2011b). An additional 2011 survey of Middle and Hamblin Ponds was also conducted and provides comparative data (Biodrawversity 2011c).

Treatment Protocols

All appropriate parties, as identified in the Order of Conditions, were kept informed of project planning progress during the summer as this process proceeded. A number of actions were necessitated by certain clauses in the Order of Conditions, including approval of a mussel monitoring program discussed above, but extending to treatment protocols as well. Approval of the applicator and monitoring team and finalizing the dose for each defined treatment area were straightforward permit compliance actions.

Figure 18. Locations of mussel monitoring plots in Mystic Lake (from Biodrawversity 2011a).



Several needed project adjustments required permit changes, however, necessitating consideration by both the Conservation Commission and the NHESP. Some modifications were simple adjustments, such as altering deadlines to better support project objectives or to allow more appropriate response times. Some changes were slightly more complicated, such as eliminating the requirement for stratification during treatment and quantifying the threshold for treatment cessation in response to floc drift.

Treatment was originally mandated under stratified conditions, which in fact did occur, but do not represent a critical factor in treatment effectiveness and impact minimization under the formulated plan. The unqualified requirement for treatment to cease if there was any drift of floc outside the target zones was appropriately modified to allow up to 1 cm of accumulation outside the planned treatment footprint; precise control of drift is not feasible and small accumulations were not expected to result in any negative impacts. The desirability of requiring approved plans in Orders of Conditions without getting too specific in the actual permit before those plans are developed is underscored by the needed changes.

The intended project start date, commencing with the pre-treatment meeting and followed by treatment of areas A and C, was planned for September 21, 2010. Mobilization and preparation occurred on September 17 -20, 2010, and all agencies and interested parties were notified accordingly. Very few agency representatives attended the September 21st morning meeting; hopefully this was a function of having been kept well informed during the planning process and being comfortable with the approach and adjustments.

Actual treatment followed protocols developed in other aluminum application projects, as modified by the Order of Conditions governing the Mystic Lake treatment. Access for equipment was at the south end of the lake, on private property that includes the cranberry bog at that end of the lake. Some upland landscape had to be damaged to gain access for larger equipment, but restoration was easy. The actual entry point was all sand with no vegetation, so there was no significant bank damage. Chemical storage and delivery were based at the north end of the lake, on public property off Race Road. Appropriate signage was posted, letting potential users of this property know that no parking was allowed during the treatment, which occurred on only 6 days over 15 days of calendar time.

Tanker trucks arrived on each day of treatment and the driver remained with the truck until all liquid was used on that day; treatment was set up to use what was delivered on each day whenever possible. The treatment barge and support boats were stored at either the north or south ends of the lake overnight; there were no vandalism problems. Signage at other locations informed residents and lake users of the treatment, but the late September to early October time frame minimized use conflicts.

Buoys were used to mark start and end points for treatment, but the treatment was guided mainly by a GPS system. The barge ran along parallel transects spaced to match the areal pattern of aluminum chemical application, depositing about half of the intended dose for the day when the target area was covered. Transects were then run perpendicular to the first set of transects, covering the same area with the second half of the intended dose.

Treatment was prohibited when wind speeds exceeded 20 miles per hour (mph), ostensibly to control floc drift, but safety would dictate cessation of treatment at such wind speeds anyway. The last day of treatment was delayed because of either high winds or prediction of high winds; windy conditions did not occur on one date as projected by the weather forecast, but the chemical delivery had already been cancelled for the day.

There are four possible approaches to minimizing toxicity during aluminum treatments: keep the concentration of reactive aluminum below 5 mg/L, keep the pH between 6.0 and 8.0 SU, treat in a checkerboard pattern with no two adjacent areas treated in succession, and treat at or below the thermocline (ENSR 2001b, Mattson et al. 2004). The first three approaches are all workable in virtually any aluminum treatment, but the last option is both difficult and counterproductive in most cases. Treating only the bottom water layer is not possible if the lake is not stratified (in which case there is no discernible bottom layer), and if the treatment is later in summer, the phosphorus in the upper water layer is not subject to any treatment and may have considerable phosphorus as a result of internal loading prior to treatment. Deep treatment is a potentially viable option for early summer treatments, before internal loading has become severe, but is technically difficult and raises project costs. The intent is to minimize impacts, as treating an anoxic bottom water layer should present minimal risk to aquatic organisms, which avoid that zone, but the first three approaches listed above have proven adequate to minimize impacts without deep treatment.

For the proposed treatment areas and doses at Mystic Lake, the expected aluminum concentrations under complete mixing are all less than 6.6 mg/L, with most under 5 mg/L, the desirable threshold. Yet as mixing may not be complete, it was assumed that for areas where the concentration might be higher that the dose would be split into two separate treatments at least one day apart to allow flocculation reactions and settling to run to completion and clear the water of reactive aluminum. Exceptions were made for Area A, where it was desired that the maximum dose be applied to determine if there would be any toxicity to mussel plots in this area, and for Area C, where no threat of toxicity was expected even at the full dose.

Treatment of the six defined areas was orchestrated to minimize treatment of contiguous areas. Treatment of sub-areas facilitates spatial separation of sequential treatments on some dates, but the GPS guidance makes this a relatively easy task as long as it is not windy, in which case treatment would not occur anyway. Additionally, the balancing of aluminum sulfate and sodium aluminate volumes to keep the pH close to 7.0 greatly minimizes the probability of any toxicity.

Treatment and Monitoring Conduct and Results

Aluminum Application

Aquatic Control Technology of Sutton, MA was the prime contractor for the Mystic Lake project and conducted the treatment. ACT has considerable experience in aluminum application and has developed equipment and techniques to maximize treatment effectiveness and limit adverse impacts. Distribution of aluminum compounds was from a barge outfitted for the treatment with two chemical holding tanks, two vertically adjustable manifolds (one for each aluminum compound) and instrumentation to control the flow of chemical to the manifolds. The chemical tanks were filled via hoses from the tanker trucks parked at the north end of the lake, where a temporary dock was installed. The driver used GPS to move the barge along tracks within targeted treatment zones after lowering the manifolds to about 10 ft below the water surface while support staff on the deck controlled the flow of chemical (Figure 19). The speed of the treatment barge was maintained at close to 3 mph and the application rate was close to 20 gallons per minute for aluminum sulfate and 10 gallons per minute for sodium aluminate, yielding the desired 2:1 ratio by volume.

Actual treatment occurred on six days and resulted in a total application of 21,002 gallons of aluminum sulfate and 10,553 gallons of sodium aluminate (Table 5). The first day of treatment involved Areas A and C, smaller areas at the north end of the lake where the entire planned dose could be delivered in less than a day. Area A was the shallower area with monitored mussel plots in it, and those plots were monitored a day after treatment. Three days were allotted after that initial treatment day to detect any adverse impacts, without further treatment, as required by the permit and practiced in some other aluminum treatments as a precaution. Both visual monitoring and chemical testing were conducted to detect any treatment impacts. With no impacts detected and proper documentation filed, regulatory approval was quick and further treatment was allowed to proceed on the following Monday, September 27th. If not for a prediction of high winds on September 30th, the rest of the treatment could have been completed in that week. The high winds did not materialize, but treatment for that day was cancelled and conditions were not suitable again until October 5th, at which time the treatment was completed.

Aluminum treatments require detailed logistics, contingency plans, preparation, and flexibility. A lot of equipment issues are possible, deliveries of chemical or parts can be delayed, and weather is an uncontrollable factor. There were no substantial equipment problems during the treatment period, chemical deliveries arrived regularly, and everything proceeded on the fastest possible schedule until the high wind warning for Friday, October 1st forced cancellation of the conclusion of treatment on that day. Treatment resumed on October 5th and was completed by early afternoon.

Figure 19. Aluminum application process at Mystic Lake.

Loading the tanks from onshore tanker trucks.

Manifolds before lowering to treatment depth.

Driver steering harvester along a treatment track.

Rear view during treatment.

Table 5. Aluminum compound application record for Mystic Lake.

Date	Areas Treated	Load	Aluminum Sulfate (gal)	Sodium Aluminate (gal)	Start Time	AM/PM	End Time	AM/PM	Total Aluminum Sulfate for Day (gal)	Total Sodium Aluminate for Day (gal)
9/21/2010	A and C	1	500	250	11:32	AM	12:15	AM	1501	753
		2	500	250	12:35	PM	1:21	PM		
		3	501	253	1:44	PM	2:25	PM		
9/27/2010	B, D and E	1	500	250	8:26	AM	9:00	AM	4509	2252
		2	500	250	9:19	AM	9:49	AM		
		3	500	250	10:00	AM	10:37	AM		
		4	500	250	10:51	AM	11:32	AM		
		5	500	250	11:57	AM	12:36	PM		
		6	500	250	1:08	PM	1:47	PM		
		7	500	250	2:00	PM	2:41	PM		
		8	500	250	2:57	PM	3:36	PM		
		9	509	252	3:56	PM	4:21	PM		
9/28/2010	E and F	1	500	250	8:33	AM	9:07	AM	4508	2277
		2	500	250	9:25	AM	9:59	AM		
		3	500	250	10:18	AM	10:53	AM		
		4	500	250	11:17	AM	11:49	AM		
		5	500	250	12:07	PM	12:39	PM		
		6	500	250	1:28	PM	2:05	PM		
		7	500	250	2:23	PM	2:57	PM		
		8	500	250	3:11	PM	3:42	PM		
		9	508	277	3:56	PM	4:28	PM		
9/29/2010	B, D, and F	1	500	250	8:05	AM	8:38	AM	4527	2270
		2	500	250	8:53	AM	9:29	AM		
		3	500	250	9:45	AM	10:24	AM		
		4	500	250	10:40	AM	11:15	AM		
		5	500	250	11:30	AM	12:07	PM		
		6	500	250	1:45	PM	2:22	PM		
		7	500	250	2:35	PM	3:14	PM		
		8	500	250	3:25	PM	3:52	PM		
		9	527	270	4:03	PM	4:37	PM		
9/30/2010	B, E and F	1	500	250	7:15	AM	7:45	AM	3015	1518
		2	500	250	8:00	AM	8:28	AM		
		3	500	250	8:44	AM	9:15	AM		
		4	500	250	9:32	AM	10:29	AM		
		5	500	250	10:44	AM	11:18	AM		
		6	515	268	11:29	AM	12:02	PM		
10/5/2010	E	1	500	250	7:34	AM	8:07	AM	2942	1483
		2	500	250	8:24	AM	8:58	AM		
		3	500	250	9:15	AM	9:50	AM		
		4	500	250	10:10	AM	10:43	AM		
		5	500	250	11:00	AM	11:37	AM		
		6	442	233	12:00	PM	12:42	PM		
Total									21002	10553

Treatment Monitoring

Visual monitoring and field chemistry assessment were employed to track the treatment and detect any impacts. Key elements included monitoring of pH, alkalinity, temperature, dissolved oxygen, and water clarity in treated areas and surface and underwater video-aided visual assessment for physical conditions and any dead or behaviorally altered organisms after treatment. The detailed log of data and observations during treatment and immediately afterward is provided in Appendix A.

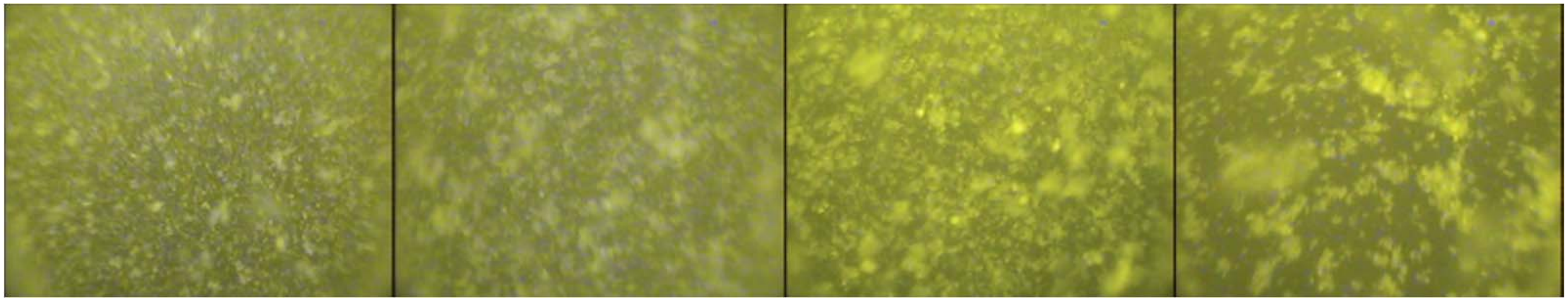
Alkalinity did not change substantially as a result of treatment and all pH values remained within the target range. The balance of aluminum sulfate and sodium aluminate minimizes related changes in alkalinity and pH, and worked well during this treatment. Temperature data indicated that stratification was breaking down during the treatment, but there was still a substantial thermal gradient between surface and bottom locations all over the lake at the end of treatment. Northern areas are shallower and stratification breaks down sooner than in the southern part of the lake; oxygen was >2 mg/L by the end of treatment, but still much lower than at the surface. The southern area remained stratified through the treatment period, but the thermocline was sinking. Still, oxygen remained negligible in deep water in the southern portion of the lake throughout the treatment period. Water clarity increased somewhat in treated areas, but did not exhibit major increases immediately after treatment. Given substantial wind mixing during this period and treatment of <40% of the lake area, this is not surprising.

The visual assessment required considerable time and involved complete circuits around the lake to look for distressed or dead fish and transects through treated areas to assess the appearance of the lake bottom and any evidence of stress on biota that might be in the area by remote video. Very few dead fish were found, less than one might expect in an untreated lake. Dead fish included two alewife, one of which was clearly an adult used as bait (hook marks) and the other had been cut up by a boat prop or other sharp object. One dead yellow perch was found, along with several dead white suckers, at least one of which was very likely to have been dead before the treatment started. There was no widespread or significant fish mortality, and the thresholds set in the Order of Conditions for intolerable fish mortality were never approached. The dosing at any instant in time kept the aluminum concentration below 5 mg/L and the pH did not move outside the 6.0 to 8.0 SU range, so no toxicity was expected. Underwater video showed fish swimming through the floc during treatment without apparent effect.

It was harder to assess any impacts on mussels by the visual survey, as so many mussels had perished in the major die off of 2009 and subsequent die off of 2010, and the bottom at depths <25 ft was littered with shells of dead mussels. No obvious death or stress was observed, but these observations did not constitute a definitive assessment. Mussel impact assessment therefore depended upon the monitoring program conducted by Biodrawversity, which is detailed in a separate report (Biodrawversity 2011a) and summarized in the next section.

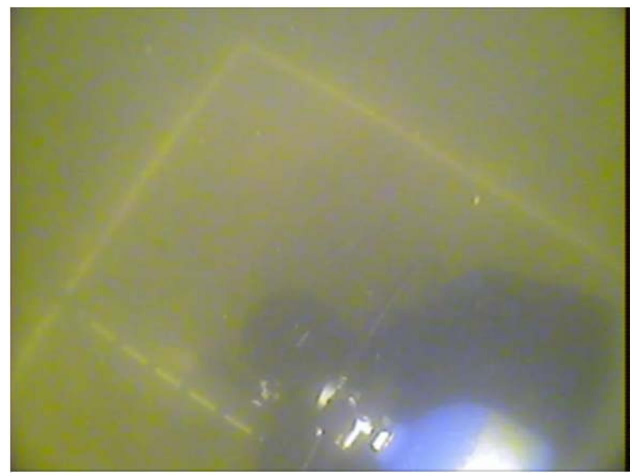
It is difficult to convey in words the observations from many hours of visual monitoring, but the photographs in Figures 20 and 21 provide some visual reference for the subsurface aspects of the treatment.

Figure 20. Subsurface photo capture from underwater video collected during treatment monitoring.



Floc formation from near the surface (left) to near the bottom (right), showing increasing size of particles as flocculation proceeds.

Floc landing on bottom in Area A on day 1 of treatment.



Mussel plot in Area A being surveyed the day after treatment

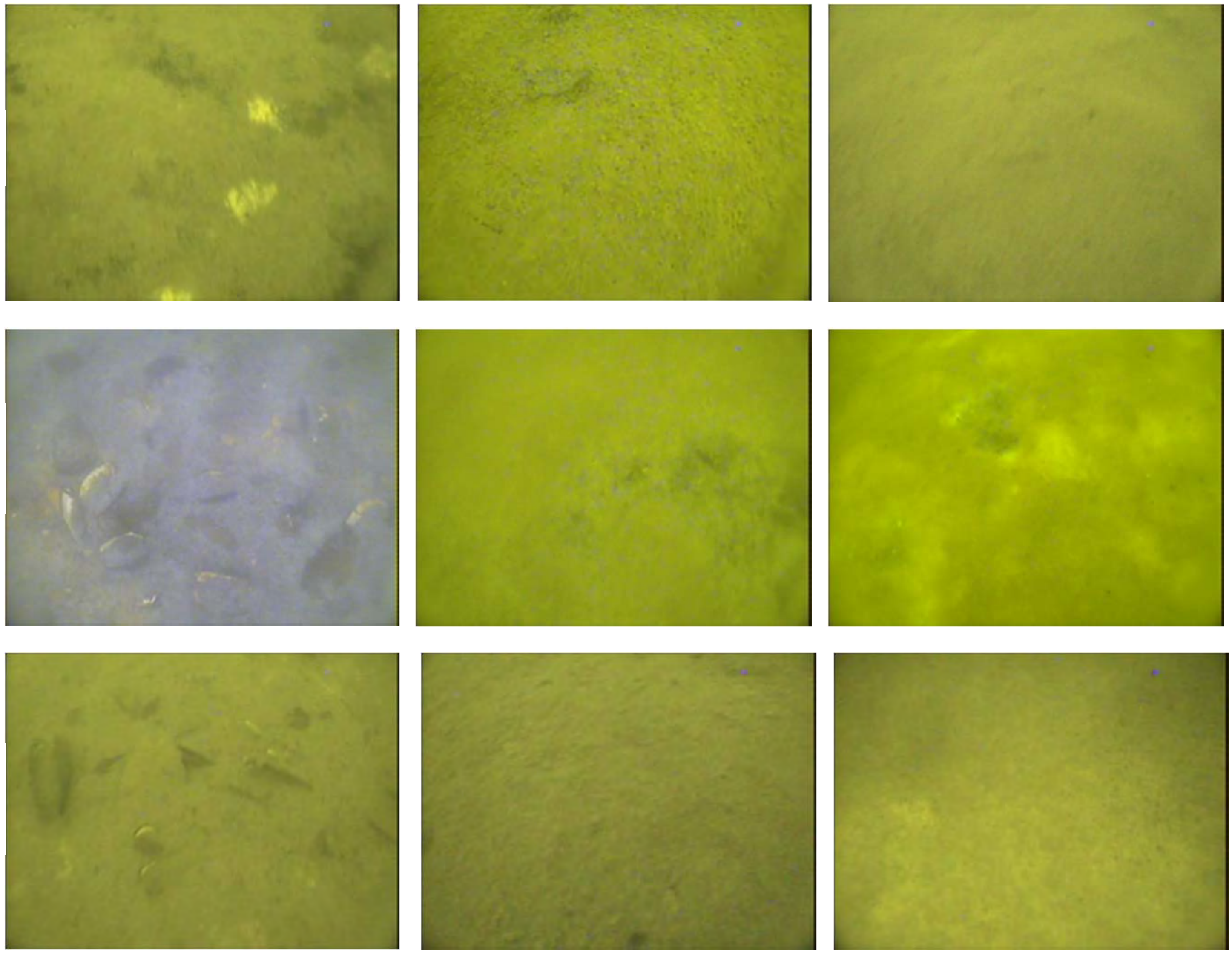
Figure 21. Bottom conditions over depth and time in Mystic Lake.

15 ft depth (Area A) | 25 ft depth (Area D) | 35 ft depth (Area E)

Pre-treatment

1 day post-treatment

3 weeks post-treatment



Mussel Impact Assessment

The mussel monitoring program established for this project involved assessment of mussel features in multiple established plots around Mystic Lake, with two plots in an area directly treated with aluminum, two plots in an area on the edge of a treatment zone (unlikely to receive any direct aluminum, but some exposure to floc drift was likely), and four additional plots divided among two areas not targeted for treatment and unlikely to receive significant amounts of floc by drift (Figure 18). Monitoring of mussel condition and behavior was conducted before, during and after treatment. A lakewide survey was also conducted, but with the high mortality in 2009 and 2010 prior to treatment, the assessment of stocked plots was considered to represent a better means of assessing aluminum impacts.

Paraphrasing from the Biodrawversity (2011a) conclusion, monitoring of mussel plots in Mystic Lake during and after treatment with aluminum did not detect any difference in the behavior or mortality of mussels between treatment and control plots. Added aluminum did not create any short-term adverse impacts on the mussels. The study did not investigate longer term effects on processes such as survival, growth and reproduction, and assessment of such impacts by aluminum treatment is confounded by the mass mortality of mussel populations prior to treatment and possibly ongoing unstable conditions. Yet it is clear that the actual treatment did not result in significant mortality of mussels and did not appreciably change their behavior with regard to position or feeding.

Toxicity was a primary concern of regulatory agencies and was not observed as a result of the Mystic Lake treatment. As the aluminum becomes an inert material after reaction, which occurs over several months but is most intense in the few hours immediately following treatment, no direct long-term impacts seem likely. The vast majority of the aluminum was deposited in deep areas not inhabited by mussels. Where treatment intentionally overlapped with mussel habitat, assessment of established plots indicated no adverse impacts. Drift allowed some floc to reach potential mussel habitat over a broad peripheral area of Mystic Lake, but the layer was thin and no impacts were noted in either established mussel plots or the subsequent lakewide mussel survey (Biodrawversity 2011b). However, the mass mortality of more than 90% of the overall mussel community in 2009 and 2010 prior to treatment minimizes the potential for detecting impacts from the aluminum treatment by lakewide surveys. The approach embodied by the established mussel plots provides the most reliable indications we have, and suggests that the aluminum treatment, as conducted, did not produce adverse impacts on mussels.

There was concern expressed by the NHESP during the initial permitting process that the aluminum treatment would reduce the fertility of Mystic Lake and compromise support of what was considered an outstanding mussel community prior to the 2009 die off. As the purpose of the aluminum treatment is to reduce internal recycling of phosphorus and limit the potential for algal blooms, this concern is best evaluated in the context of changing algal abundance and associated water clarity. The short term mussel monitoring program does not provide useful data for evaluating this issue; it simply but conclusively indicates that the aluminum treatment had no short-term negative impacts on mussels.

While not directly related to the aluminum treatment, observations on the condition of mussel shells bear mention in association with monitoring activities. Substantial shell “erosion” was noted on a high percentage of mussel shells in both Mystic Lake and Middle Pond during the course of work at these lakes. E. Nedeau of Biodrawversity noted that the degree of erosion and portion of the populations exhibiting it had increased over the 4 years he has done surveys in this lake system. This is not a matter of dissolution of shell material upon death; many live mussels exhibit shell erosion, to the point where the outside of the shell appears much like the inside when opened, with the pearly coating exposed (see photo on report cover). This phenomenon was observed before aluminum addition. Given higher pH and alkalinity in Mystic Lake, this is not an acidification issue. The reason for this condition is unknown, and any linkage to the algal blooms or mussel die off is speculative, but it is clearly not a healthy condition for the mussels.

Water Quality

Monitoring data are available for shortly before the treatment began in mid-September 2010, on the last day of treatment in early October 2010, three weeks after the end of the treatment in late October 2010, mid-February 2011, and then once in each of May, June, July, August and September of 2011. The August sampling was conducted by IPA volunteers under the PALS program, with analyses conducted by the SMAST program at UMASS Dartmouth. All other samples were collected by WRS with help from the IPA, with field analysis where possible and lab analyses conducted by Berkshire Enviro-Labs of Lee, MA. Before, during and immediately after treatment, water quality monitoring focused on phosphorus, water clarity and aluminum, with supporting data for temperature, dissolved oxygen, pH, alkalinity, and conductivity collected. Algae and zooplankton were also assessed. After it was apparent that post-treatment aluminum levels were low, that analysis was discontinued in accordance with the conditions of the permit, but it was thought prudent to add forms of nitrogen (ammonium, nitrate and TKN) to the program voluntarily. Data collected by WRS in fulfillment of the monitoring requirements of the permit, along with selected relevant data from the IPA/SMAST, are provided in tabular form in Appendix B.

The primary goal was to improve water clarity, although this involves a fairly complex chain of events to reach that goal. Secchi transparency for all summer assessment dates in the entire period of record (Figure 23) demonstrate that water clarity in 2011 was improved over 2009-2010 levels, was also better than 2005, was roughly comparable to 2004, but was not as high as in years prior to that. Water clarity in Mystic Lake improved from between 2 and 3 m to between 3 and 4 meters in late June and early July as stratification set in, but declined to just above 2 m in mid-July and remained fairly stable at that level the rest of the summer. Chlorophyll data are limited (Figures 24 and 25) to August of each year through the PALS program, and shed little light on this transition. However, the chlorophyll data do indicate a decrease in 2011 over recent years at all sampled depth levels, but especially in the deep area, where algae build-up has been very high in some recent years. This is consistent with the major decline of deep water phosphorus levels.

Review of thermal profiles for summer data (Figure 26) suggests that while the pattern among years is similar, there are small but important differences. All August profiles after 2007 have higher than average surface water temperatures, and the pattern of temperature decline varies among years. The density gradient set up in 2008 provided the most water in the upper water layer and a strong enough difference between upper and lower levels to minimize mixing. The profile for August 2010 was similar, with an even sharper gradient but with the boundary level slightly shallower. The pattern for 2009

Figure 22. Secchi disk transparency over time in Mystic Lake.

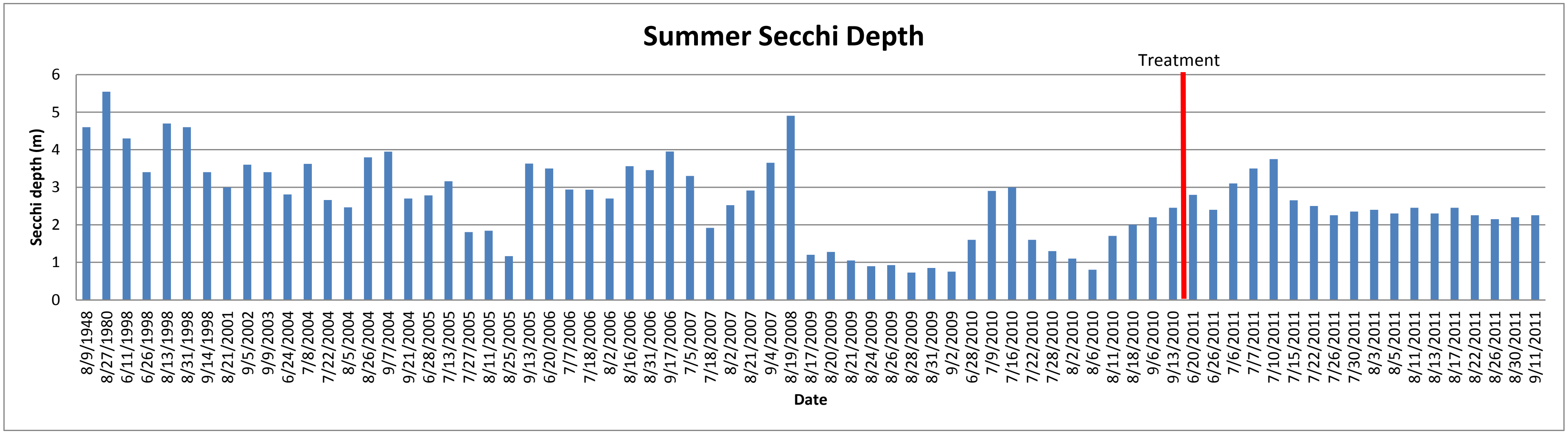


Figure 23. Total chlorophyll-a concentration in Mystic Lake.

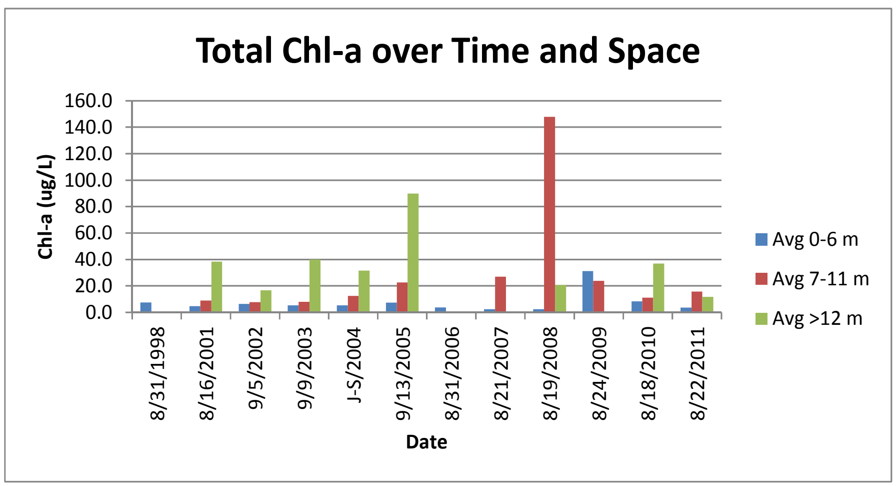


Figure 24. Volume weighted total chlorophyll-a in Mystic Lake.

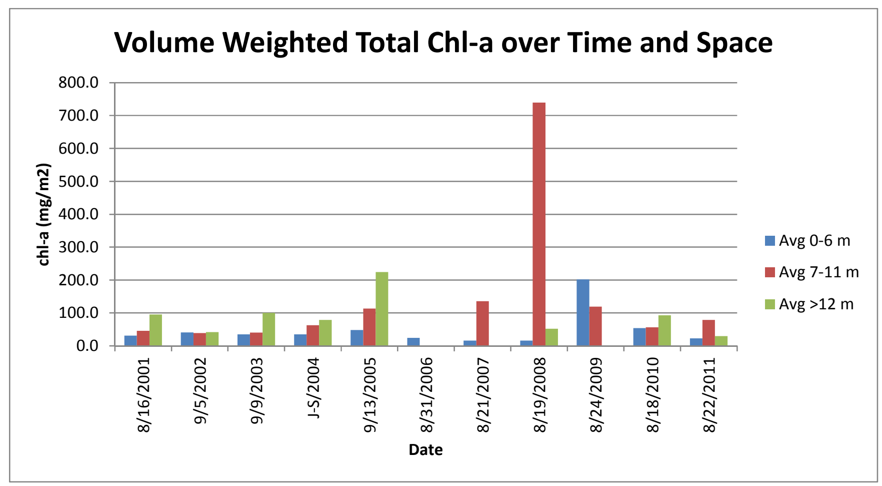
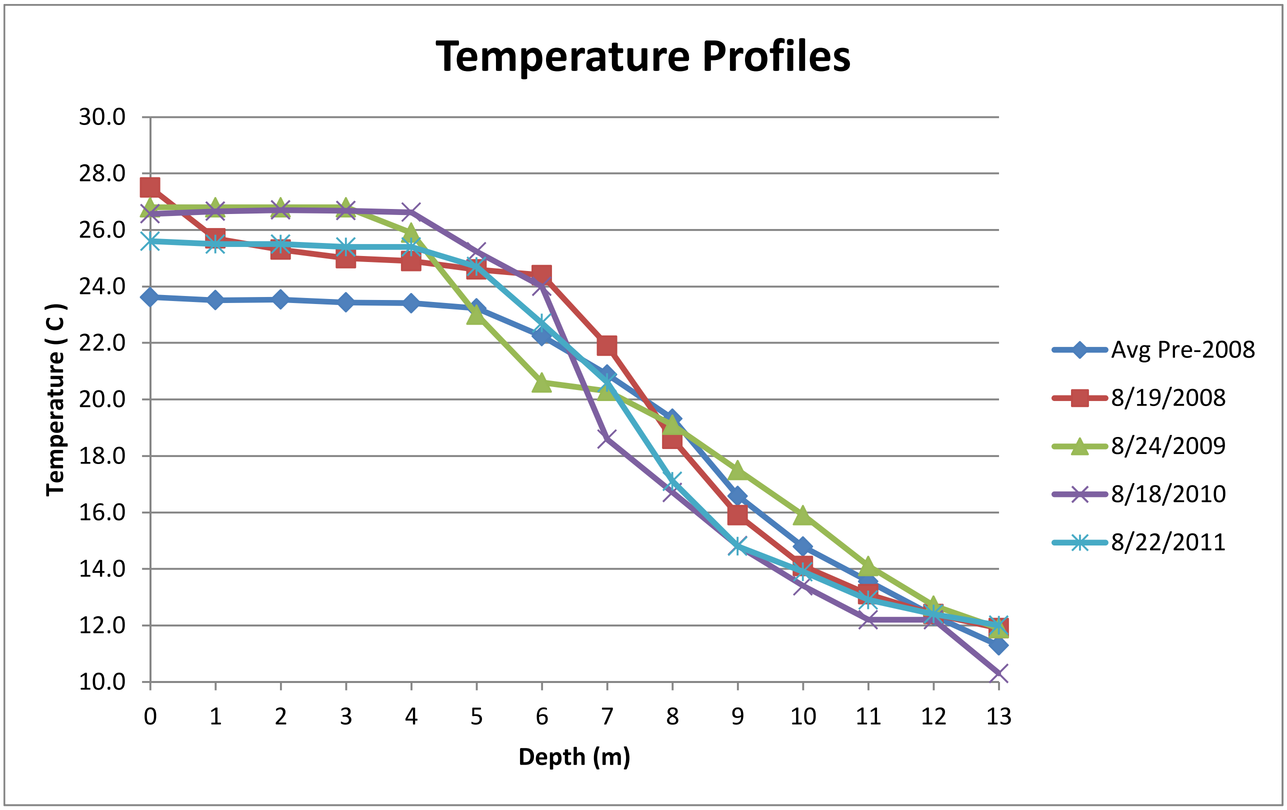


Figure 25. Summer temperature profiles for Mystic Lake.



showed stratification starting at much shallower depth (4 m) than the other years, with a warmer deep layer as well. The profile for 2011 was similar and intermediate to 2008 and 2010. The thermal pattern is a function of weather pattern at Mystic Lake, with both sun and wind playing major roles, and the timing of each is important.

The damaging bloom conditions of 2009 appear related to strong winds that mixed the water much deeper than usual before stratification set in, after which stratification occurred as a shallower depth. It takes less temperature difference to create a boundary layer at warmer temperatures, so a little bit of strong sun after a very windy period in 2009 was likely responsible. This allowed nutrients from the bottom to mix further upward than they might otherwise, facilitated algal growth at a level with more light, and allowed later mixing to pull at least the top of that lower water layer upward. The result is evident in the chlorophyll pattern (Figure 14). Both 2009 and 2010 exhibited the warmest temperatures recorded for the upper water layer, but in 2010 the lake stratified about 2 m deeper, limiting light to the deep algae layer and restricting nutrients for shallow algae growths. Some nutrients and algae did make it to the upper water layer, as there was a bloom in 2010, but it was not as severe as in 2009. The much deeper depth of the boundary layer in 2008 kept what was a huge quantity of algae at that boundary deep enough to minimize impacts on surface water during the period of stratification; has stratification been shallower in 2008, the problems encountered in 2009 could very well have occurred in 2008.

The 2010 bloom occurred and ended earlier than in 2009, and appears related to the stratification pattern. Stratification was weak in early July (Figure 27), allowing more mixing and upward movement of nutrients and algae. A sharper separation of upper and lower water layers developed between mid-July and mid-August, at a depth of about 6 m, and the bloom ended in mid-August. Likewise, the development of stratification in 2011 tracks the decline of water clarity in this post-treatment period (Figures 22 and 28). Initially high clarity began to deteriorate in early July, then leveled off once stronger stratification at a more substantial depth set in by mid-July. The very cloudy, wet, windy conditions in summer 2011 were likely responsible, with stable stratification not set up until late July with a boundary between upper and lower water layers at about 6 m.

Stratification strongly affects oxygen levels in Mystic Lake, and the depth at which oxygen disappears shows a high correlation with water clarity; higher oxygen in deeper water correlates with greater clarity (Figure 10). Summer dissolved oxygen profiles (Figure 29) show that 2009 had the shallowest depth of low oxygen (<5 m), followed by 2010 (6.5 m) and 2011 (7.5 m). The depth at which oxygen became <1 mg/L in 2008 was among the deepest recorded (9 m) with only 1980 and 1948 clearly better. Some years in the early 2000s exhibited deeper depths of complete oxygen loss, but the loss was more gradual, and oxygen levels higher in the water column were lower than in 2008. It is apparent that where anoxia develops at a depth shallower than 9 m, the potential for algal blooms rises substantially and water clarity declines. This is entirely consistent with Mystic Lake morphometry (Figure 2); much of the lake is <9 m deep, so having oxygen to a depth of at least 9 m limits the area that will contribute phosphorus through release from iron compounds under anoxia.

In 2010 the depth at which oxygen was low to negligible rose in the water column by over a meter between July 9 and August 11 (Figure 30), a very warm period; warmer water holds less oxygen, so there was a decline in surface water oxygen during this time, but oxygen demand in deep water is a major factor in oxygen loss in the lower water layer. Concurrently, wind mixing homogenized the shallower water and thermal stratification became stronger during this period (Figure 27), setting a fairly broad boundary between 4 and 6 m, although anoxia did not occur until about 7 m. Yet by this time an algal bloom had occurred, although it subsided somewhat in August, possibly due to limited movement of phosphorus upward from the bottom water layer with the stronger stratification.

Conditions in 2011 appeared very favorable at the start of summer (Figure 31). Low oxygen was not encountered above 11 m on June 20th, but oxygen was low at 9 m by July 22nd. This is a typical seasonal pattern, with stratification limiting oxygen supply from the atmosphere to the deep waters by mid- to late June, and oxygen being depleted as successively shallower levels for 2-3 months thereafter. However, there was an oxygen “bulge” that formed at about 5 m in early July and was then reduced and pushed deeper until it disappeared in late July. Such bulges are usually indicative of algal growth and oxygen release during photosynthesis. As thermal stratification became stronger in July 2011, these algae were mixed (or floated) upward into the upper water layer and reduced water clarity (Figure 23). A deeper layer of algae did not appear to form, presumably due to lack of phosphorus related to the aluminum treatment, yet to be discussed. The algae increase in surficial waters in July remained fairly stable, based on water clarity data (Figure 23), but involved changes in species composition yet to be discussed.

Figure 26. Temperature profiles during the 2010 bloom.

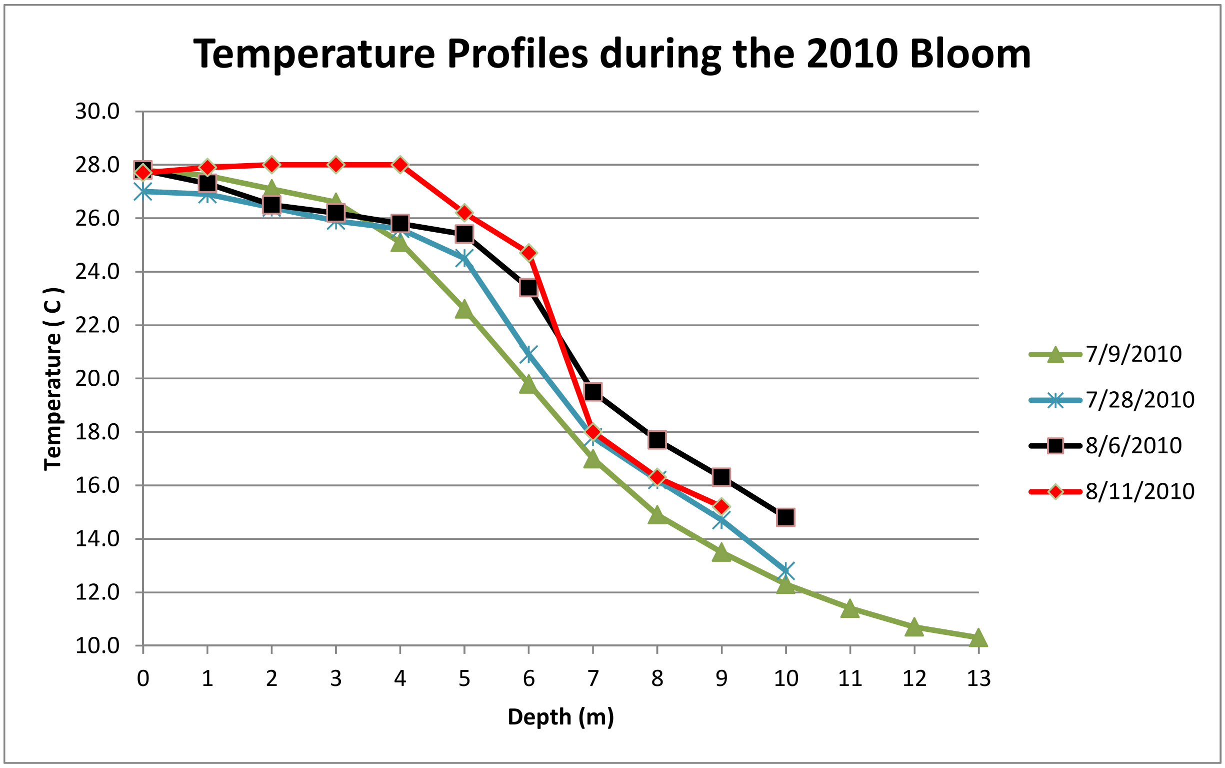


Figure 27. Temperature profiles during the 2011 period of declining Secchi transparency.

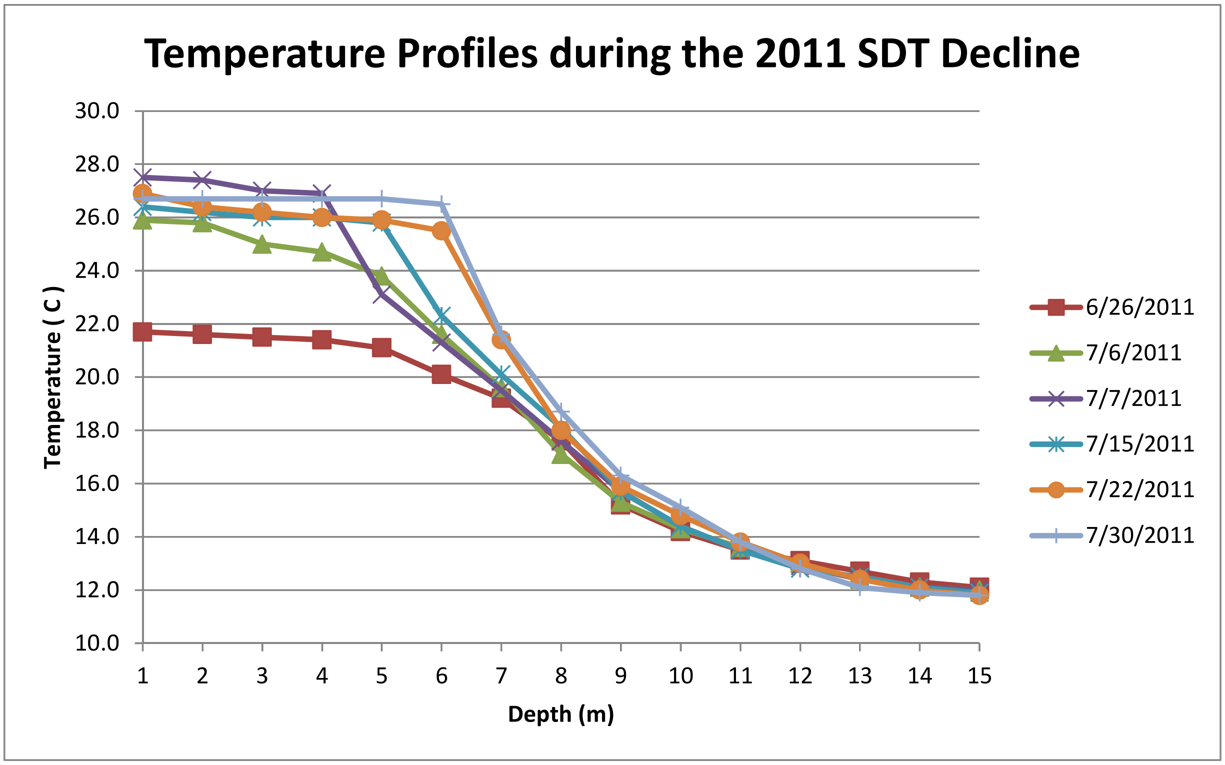


Figure 28. Summer dissolved oxygen profiles for Mystic Lake.

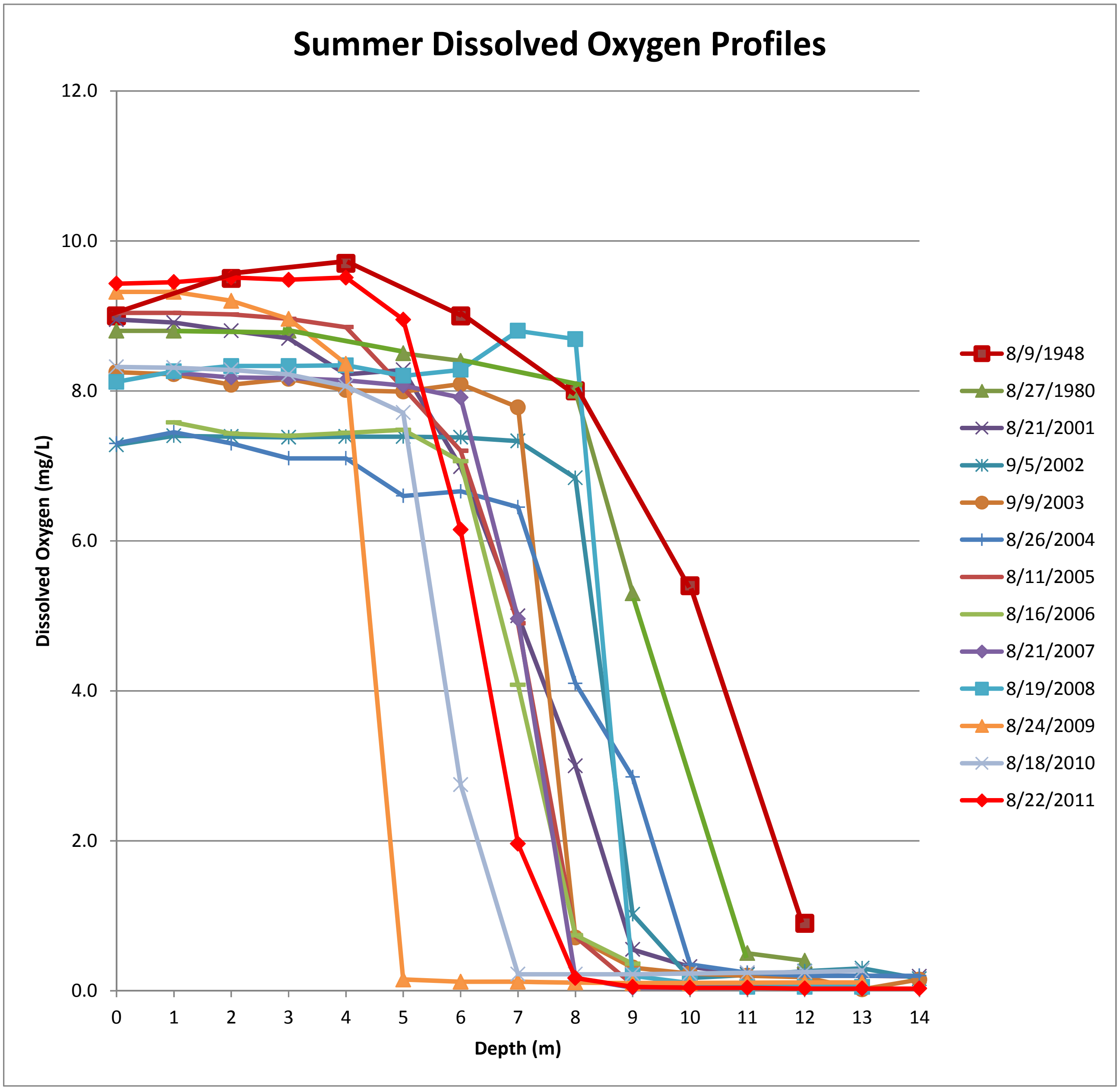


Figure 29. Oxygen profiles during the 2010 bloom.

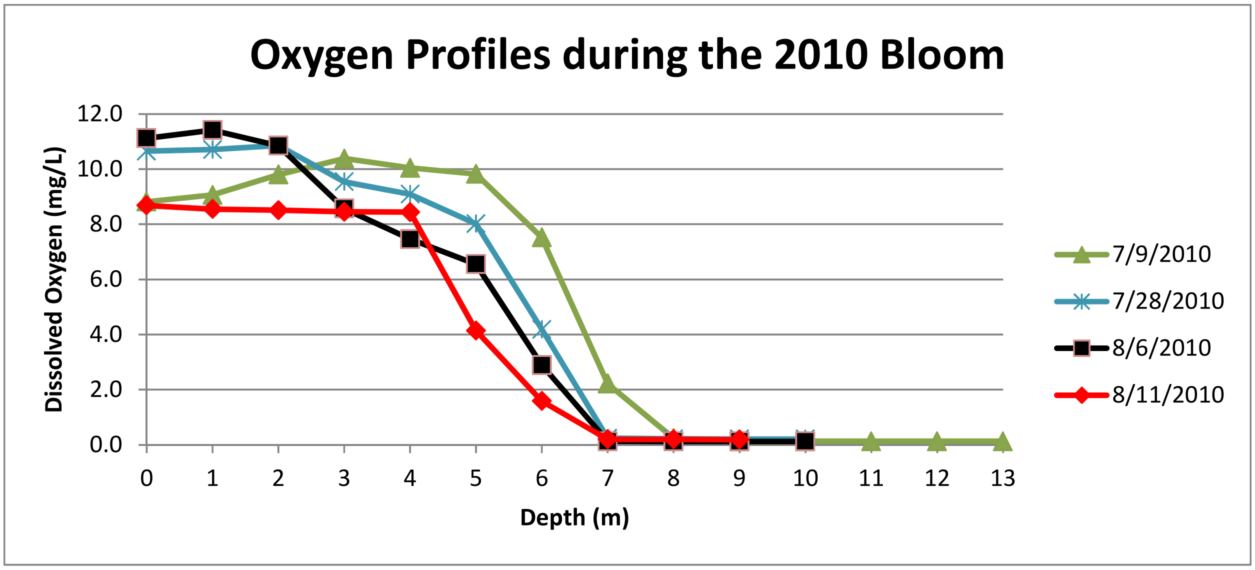
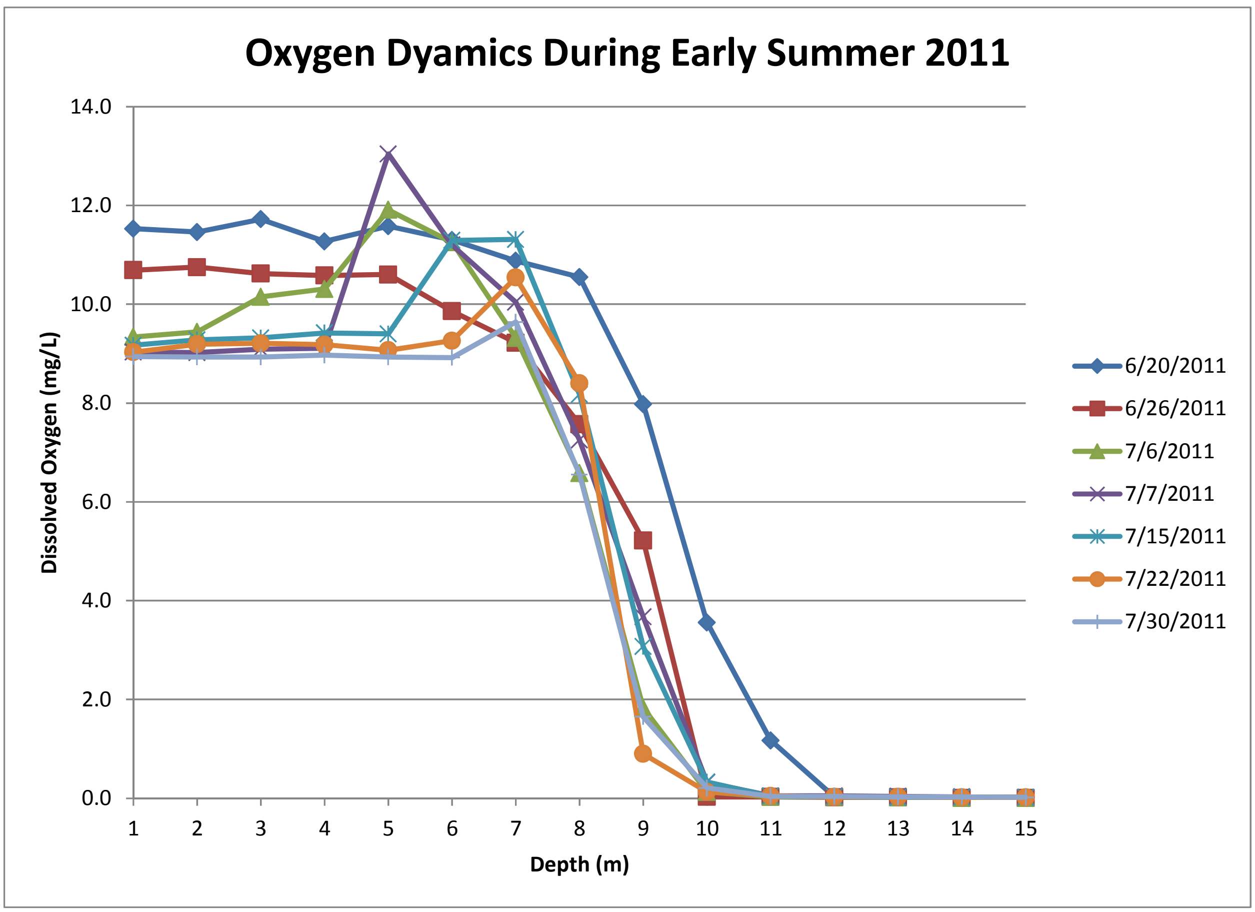


Figure 30. Oxygen dynamics in Mystic Lake during early summer 2011.



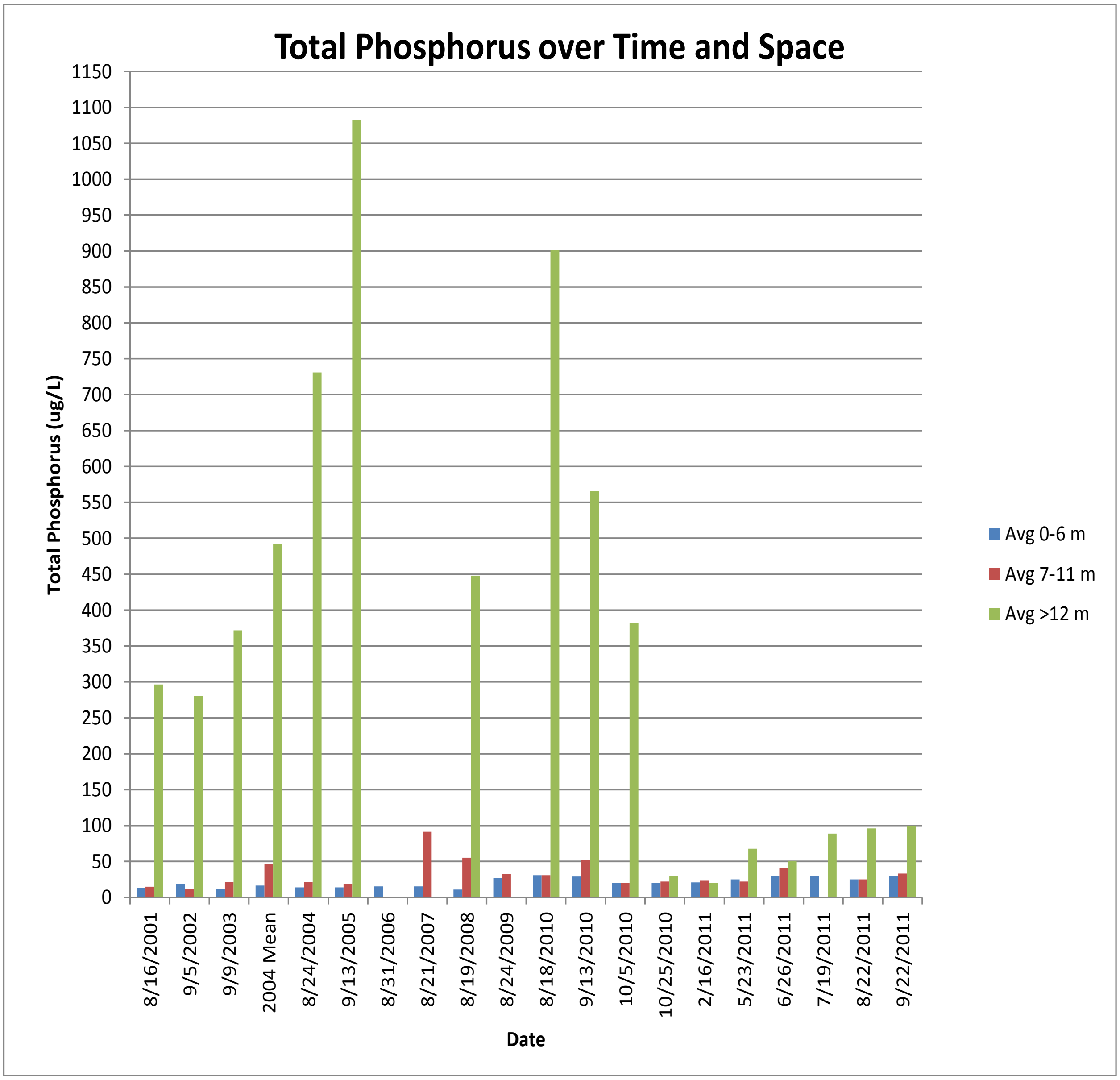
The key information to be gained from the data presented so far is that the late spring-summer processes of stratification and deep water oxygen depletion exhibit substantial variation among years in Mystic Lake. These influences will be superimposed on nutrient availability and algal growth to help determine water clarity. If not for the very different weather pattern in 2008, the low clarity and elevated cyanobacteria abundance in the upper water layer observed in 2009 might very well have occurred in 2008. Likewise, 2011 appeared similar to 2009 in terms of weather pattern, but the greater depth of anoxia (as well as the aluminum treatment) may have prevented a repeat of 2009 blooms conditions.

The distribution of chlorophyll in the water column and related water clarity appear to be strongly influenced by weather pattern during late spring and summer. To the extent that weather influences light and temperature, it can also influence algal abundance. Yet there is enough light in even the cloudiest summer to support algal blooms, and while growth is slower when the water is cold, this will not prevent growth. The presence of a dense layer of algae at depths of 9 or more during summer demonstrates this. The availability of nutrients tends to be the primary determinant of algal quantity, with the ratios of various nutrients favoring different types of algae. The ratio of nitrogen to phosphorus is very important, with low ratios (<10:1) favoring cyanobacteria and high ratios (>20:1) favoring green algae. Other nutrients, such as silica, may control the abundance of diatoms and golden algae which utilize silica in the construction of cell walls. Shifting nutrient ratios lead to succession of algal types, but the maximum quantity of algae attainable is largely linked to the concentration of phosphorus (Wetzel 1983, Reynolds 2006).

The purpose of the aluminum treatment was to increase water clarity, but unlike in water and waste water treatment where aluminum addition is used to coagulate and settle particles (thus clearing the water), in lake management we usually seek to inactivate phosphorus and make it unavailable for algal uptake and growth, thereby preventing loss of water clarity. The mechanism by which the aluminum treatment works is to bind phosphorus, to some extent in the water column, although this is not an especially efficient process, but mainly in the surficial sediments, where phosphorus bound to iron can be converted to aluminum bound forms that are not released into the water column when oxygen is depleted. By limiting the amount of phosphorus entering the water column from sediment sources, the internal load of phosphorus is decreased. Where the internal load represents a major fraction of the phosphorus load to the lake, such inactivation can provide lasting relief from algal blooms. Hamblin Pond represents a prime example of the success of this approach, which is discussed in detail by Mattson et al. (2004). Where aluminum treatment has not provided lasting benefits, it has been demonstrated that either the external load from the watershed was far more important than the internal load, or that the dose of aluminum delivered was insufficient to inactivate enough phosphorus to make a lasting difference (Mattson et al. 2004, Cooke et al. 2005).

The change in deep water phosphorus in Mystic Lake after aluminum treatment is striking (Figure 32). Although the deep water value did rise gradually from the lows of 30 ug/L three weeks after treatment and 20 ug/L in February 2011 to a high of 100 ug/L in September 2011, values were much lower than the

Figure 31. Total phosphorus data for Mystic Lake.



range of 280 to 1093 ug/L from 2001-2010, with a mean of 574 ug/L. Not all of this phosphorus will reach the upper water layer during summer, but with a surface water target concentration of 10 ug/L to prevent blooms and an expected limit of 20-25 ug/L to avoid severe and recurrent blooms, the deep water phosphorus levels represent a very large threat. The reduction to <100 ug/L is encouraging, but with lower than desired clarity in summer 2011, there is still cause for concern. Phosphorus levels in the upper water layer were not lower in 2011 than in recent previous years, suggesting a carryover effect or alternative source of phosphorus. Further analysis of phosphorus data is in order.

Although the change in measured concentration is often insightful, the potential for redistribution of phosphorus vertically in a lake and the importance of summer surface layer phosphorus levels makes it necessary to look at the change in phosphorus on a mass balance basis. The phosphorus profile for each monitored station from just prior to treatment until the end of summer 2011 is complete at 2 m intervals for most sampling dates; missing points have been filled in by interpolation to allow this analysis. Considering the mass of phosphorus in milligrams for each of 5-8 discrete "blocks" of water that can be stacked to represent the water column at each sampling station (Tables 6-8), there has been a change in the deepest block at ML-1 and ML-2, at the sediment-water interface, but not at shallower depths (Tables 6 and 7). For the deeper ML-3 the change is recognizable over the bottom three blocks, with shallower water not showing a lasting change in total phosphorus after treatment (Table 8). All three stations match up in that the change occurs at about 9 m and deeper, while shallower areas continue to exhibit elevated phosphorus levels.

There are multiple possible explanations for this situation:

- High flux from insufficiently treated areas – If the treatment was insufficient, additional release of phosphorus would be possible, but the lack of accumulation in the deepest zone indicates that if such release is occurring, the resulting phosphorus must be passed rapidly upward. This is counter to all observations prior to the treatment and seems very unlikely.
- Inadequate iron binding of phosphorus – Iron is the natural phosphorus binder in Mystic Lake and indeed most Cape ponds, and figures into normal phosphorus cycling. Sulfides are produced in at least the deepest part of the lake to the south, and may be produced elsewhere but lost with periodic aeration in areas marginally deep enough to stratify temporarily. Sulfides permanently bind iron much as aluminum permanently binds phosphorus, and can create a shortage of soluble iron. It would be quite a coincidence that iron binding capacity diminished just when the aluminum treatment was conducted (aluminum does not bind iron, so that is not a mechanism of iron loss), and ground water flow on the Cape tends to carry a lot of iron into lakes, so this seems like a very unlikely mechanism. However, iron levels have not been measured in the waters of Mystic Lake in any recent years.
- Carryover levels from previous loading to surface waters – The aluminum treatment will strip some phosphorus from the water column, but with low efficiency at lower phosphorus concentrations. The process is very efficient in surficial sediments with phosphorus levels in the range of 170 to 1000 parts per million (ppm, equivalent to mg/kg or mg/L), and is moderately effective in the deep waters with phosphorus levels in the range of 0.3 to 1.1 ppm, but will remove only a minor fraction of the phosphorus in shallower waters with concentrations <0.1

Table 6. Phosphorus mass under one square meter of water at ML-1.

	TP mass per square meter at ML-1 (mg)					
Depth Stratum (m)	9/13/2010	10/5/2010	10/25/2010	2/16/2011	6/26/2011	9/22/2011
0-1	27	31	25	22	29	24
1-3	50	40	38	38	66	54
3-5	62	32	34	36	72	70
5-7	46	30	46	50	60	70
>7	104	38	38	52	110	60
Total	289	171	181	198	337	278
0-5 m	139	103	97	96	167	148
0-7 m	185	133	143	146	227	218
0-9 m	289	171	181	198	337	278

Table 7. Phosphorus mass under one square meter of water at ML-2.

	TP mass per square meter at ML-2 (mg)					
Depth Stratum (m)	9/13/2010	10/5/2010	10/25/2010	2/16/2011	6/26/2011	9/22/2011
0-1	29	17	17	22	23	30
1-3	52	40	22	34	60	62
3-5	72	38	40	40	62	62
5-7	58	38	38	34	68	72
7-9	78	42	44	46	70	70
>9	140	34	44	38	56	84
Total	429	209	205	214	339	380
0-5 m	153	95	79	96	145	154
0-7 m	211	133	117	130	213	226
0-9 m	289	175	161	176	283	296

Table 8. Phosphorus mass under one square meter of water at ML-3.

	TP mass per square meter at ML-3 (mg)												
Depth Stratum (m)	2001-2007 Mean	8/19/2008	8/24/2009	8/18/2010	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	8/22/2011	9/22/2011
0-1	15	7	23	23	26	13	16	16	23	18	23	19	27
1-3	30	30	67	78	62	54	44	38	48	56	56	50	56
3-5	36	36	60	72	58	44	46	52	54	56	58	62	60
5-7	40	40	49	66	62	42	46	54	66	78	72	56	60
7-9	66	111	48	62	60	46	48	54	60	66	62	56	60
9-11	174	180	83	120	78	36	46	46	62	76	68	50	60
11-13	900	896	?	800	726	206	62	46	52	56	102	120	154
>13	727	?	?	901	768	660	28	17	68	74	89	96	122
Total	1987	1300	331	2122	1840	1101	336	323	433	480	530	509	599
0-5 m	80	73	150	173	146	111	106	106	125	130	137	131	143
0-7 m	120	113	200	239	208	153	152	160	191	208	209	187	203
0-9 m	187	224	248	301	268	199	200	214	251	274	271	243	263
9-14 m	1800	1076	83	1821	1572	902	136	109	182	206	259	266	336

Table 9. Phosphorus mass in Mystic Lake over time.

			TP mass in lake (kg)												
Depth Stratum (m)	Volume (m3)	Fraction of Total Volume	2001-2007 Mean	8/19/2008	8/24/2009	8/18/2010	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	8/22/2011	9/22/2011
0-1	510000	12.0%	7	4	12	12	13	7	8	8	12	9	12	10	14
1-3	1050000	30.5%	16	16	35	41	33	28	23	20	25	29	29	26	29
3-5	790000	18.0%	14	14	24	28	23	17	18	21	21	22	23	24	24
5-7	610000	16.3%	12	12	15	20	19	13	14	16	20	24	22	17	18
7-9	530000	14.5%	18	29	13	16	16	12	13	14	16	17	16	15	16
9-11	150000	4.4%	13	14	6	9	6	3	3	3	5	6	5	4	5
11-13	160000	3.3%	72	72	?	64	58	16	5	4	4	4	8	10	12
>13	60000	1.1%	44	?	?	54	46	40	2	1	4	4	5	6	7
Total	3860000	100.0%	196	160	105	245	214	136	86	88	107	117	121	111	125
0-5 m	2350000	60%	37	34	71	81	69	52	49	49	58	61	64	60	67
0-7 m	2960000	77%	49	46	86	101	88	65	63	65	78	84	86	78	85
0-9 m	3490000	91%	67	75	99	118	104	77	76	79	94	102	102	92	101
9-14 m	370000	9%	129	85	6	127	110	59	10	8	13	15	19	19	24

ppm. Consequently, phosphorus released from sediment prior to a fall treatment has a low probability of being removed. Spring treatments tend to be more efficient, with most phosphorus still in the surficial sediment prior to most internal release for the year. Surface water concentrations in 2009 and 2010 (27-31 ug/L) were high relative to desirable levels for control of algae, but were low relative to aluminum efficiency, making it very likely that surface water phosphorus levels would not be directly reduced by the treatment. Adding that aluminum was injected about 3 m below the water surface, carryover of surface phosphorus levels after treatment would be expected. Deep water phosphorus removal appears to have been more effective, with major declines in areas >9 m deep.

- Compensatory release from untreated areas – With reduced phosphorus levels at least in deep water after treatment, there will be a tendency for chemical equilibrium to pull more phosphorus from the sediment. Where treatment occurred, such release would be limited, but only about 40% of the lake bottom was treated, and both Eichner et al. (2006) and AECOM (2009) indicate that around half of the lake has organic muck sediments that could be large sources of phosphorus. It has not been considered likely that areas not subject to anoxia contribute major quantities of phosphorus, but some contribution is possible, and it may increase as the phosphorus levels in overlying water decline. Additionally, decay of organic matter will contribute some phosphorus, and release from this source may also increase in what has been termed a “rebound” effect. This tends to be temporary, however, and should not sustain high phosphorus levels for multiple years.
- Decay of mussels – The rebound from organic matter noted above may have gotten a major boost from the death of as many as 24 million mussels in 2009 and 2010 (Nedeau 2011b). This is very likely a factor in the rise of phosphorus in 2009 and maintenance of high levels in 2010. As the mussels were almost entirely outside the treatment area, and die off was still in progress only a month or two prior to treatment, a carryover effect from the die off is possible. Based on an average dry tissue weight of 2.2 g/mussel (Nalepa and Gauvin 1988) and a tissue content of 0.6% phosphorus (Kuenzler 1961), every million mussels would equate to 13.2 kg of phosphorus. The change in phosphorus mass in the upper 9 m of the lake since 2009 (Table 9) averages 24 kg, equivalent to <2 million mussels when many more than that died. The maximum difference (August 2010 vs. pre-2009) is 44 kg, <14% of the amount of phosphorus potentially released by dead mussels.
- Watershed loading – The watershed of Mystic Lake is responsible for about half the annual load to the lake, but with substantial variability among seasons and years. Spring and summer of 2011 were very wet, and may have added more nutrients to the lake than usual. Runoff is not nearly the factor in phosphorus loading to Cape Cod lakes that it is in less sandy areas, but there were major storms that did create significant runoff in 2011. The associated load would enter from the top of the lake, potentially raising the phosphorus level in the upper water layer.

Any or all of the last four of these factors may contribute significantly to the continued high phosphorus levels in the upper water level of Mystic Lake after the aluminum treatment, and while the first two factors are unlikely to be important, they cannot be completely ruled out. However, the death of so many mussels and release of so much phosphorus from that long-term repository appears to be the most likely mechanism for elevated phosphorus in the lake since 2009. Mussel deaths can more than account for the observed increase, even considering slow release and potentially large losses of phosphorus to fish, birds and outflow. In 2011, there appears to be some increase in phosphorus that might be related to releases from sediment in all areas; this could still be related to mussel deaths from 2009 and 2010, or could be a function of sediment in which phosphorus was not completely inactivated.

Natural processes could lower phosphorus levels over time if internal loading remains low, as some percentage of the phosphorus becomes bound in organic matter that builds up on the lake bottom and does not release its phosphorus, and some travels to Middle Pond via the surface flow connector channel. Yet the mussel die offs represent double jeopardy; a large quantity of available phosphorus is likely to have added to the water column, and the loss of filtering capacity reduced algal loss rates. As zooplankton grazing in Mystic Lake is very low, the maximum algal biomass per unit of available phosphorus is to be expected.

Putting aside the current issue with the vertical distribution of phosphorus in Mystic Lake, a central question involves whether the treatment reduced the overall load to the lake to the desired degree. Fortuitously, detention time is about one year, yielding a flushing rate of one year as well (AECOM 2009). This means that mathematically the mass of phosphorus in the lake at any point in time is roughly equivalent to the annual load. Calculation of the mass of phosphorus in the lake by summing values for defined water strata of 1 or 2 m thickness (depending upon data availability) yields annual estimates of loading for 2001 through 2009 and for several points in 2010 and 2011 (Table 9). The mean of 2001 through 2007 was taken as a reasonable estimate of average loading in recent years, with 2008 and 2009 having incomplete data that preclude complete mass estimation (shaded cells in Table 9). Data from two pretreatment dates in 2010 provided estimates, as did data for the last day of treatment and three weeks after treatment in October 2010. In 2011 it was possible to estimate loads from sampling in February and monthly in May through September. In some cases missing data for specific water strata were estimated by interpolation, but the resulting estimates of phosphorus mass appeared reliable.

Results suggest that the mean load for 2001-2007 was on the order of 196 kg/yr, higher than the expected mean of 120 to 125 kg/yr and outside the range of 88 to 173 kg derived by AECOM (2009), but those estimates were based on a lake volume about 9% less than what is now considered accurate. The 2008 and 2009 estimates are incomplete due to missing deep water data, but values for upper strata in 2008 are consistent with the 2001-2007 mean. Values for upper strata in 2009 are elevated, possibly from mixing of deep water in that year of very shallow stratification or decaying mussels, as the 2009 data are from the time of the major die off. Values prior to treatment in 2010 are higher still, with deep strata values similar to past years for which there are data and shallow strata with higher values, yielding total mass (and annual load) estimates of 214-245 kg. Mixing of deep water seems less plausible in 2010, while decay of mussels remains a very plausible explanation.

The mass on the last day of treatment was 136 kg, and this declined to 86 kg three weeks after treatment, about when most reactions would be considered complete. This is close to the target of 80 kg set by AECOM in accordance with Cape Cod Commission recommendations for achieving an average total phosphorus concentration of 10 ug/L, with commensurate chlorophyll of 3 ug/L and Secchi transparency of 4 m, along with a very low probability of algal blooms occurring. Given the smaller lake volume used by AECOM, a more appropriate loading target is now about 87 kg/yr, so the treatment did achieve the desired level of load reduction, even with much more phosphorus in the lake than indicated by historic sampling.

Unfortunately, the vertical distribution of phosphorus mass is not quite what was desired; values at the top remain elevated while the mass at the bottom is very low. The treatment greatly reduced phosphorus in lower strata where concentrations were high, but did little to reduce phosphorus mass in the upper strata (Table 9, October 25, 2010 data).

The pattern of lower phosphorus mass in deeper water strata with higher masses in shallower strata holds up all through 2011, with slight increases at all strata over time. Total phosphorus mass rises from 88 kg in February to 107 kg in May, and gradually to 125 kg in September. This still represents an improvement over recent years, but is less than desired or initially expected. The internal load in treated areas was initially depressed by more than 90% and by the end of summer 2011 the mass reduction estimate was still at least 77%, but phosphorus inputs are still sufficient to maintain shallow water phosphorus that can support algal blooms. However, most of the increase in phosphorus occurred prior to stratification in 2011, so it is not at all clear that the increase is from deep sediment release. Clarity was better than in the previous two years, but still lower than desired. Algae on most dates were of desirable quality, but a short-lived bloom of *Aphanizomenon* occurred in August. One or more of the phosphorus input mechanisms already discussed appears to be reducing the benefits of the aluminum treatment, which clearly reduced phosphorus release from sediments in target areas.

Total nitrogen data (Figure 33, Appendix B) also indicate a reduction in deep water nitrogen after treatment with some fluctuation in 2011. Aluminum treatment does not target nitrogen directly, but nitrogen-bearing particles are coagulated and settled, so some decrease is expected. It is not nearly as dramatic as for deep water phosphorus, but deep water total nitrogen values in late 2010 and all but the August deep water value in 2011 were lower than any others recorded for Mystic Lake for the last decade. The August value for total nitrogen is from the PALS program, while the other 2011 values were obtained from WRS samples sent to another laboratory, with some differences in testing methodology that might account for the inconsistency.

Shallow water total nitrogen was not high in an absolute sense, but was almost twice as high as on average for years prior to 2009 (0.5 to 0.6 vs. 0.25 to 0.4 mg/L). The one available 2009 value and pre-treatment value for 2010 were similar to post-treatment values, so the increase may not be a function of aluminum treatment. Decay of millions of mussels seems the most likely explanation, but we have no detailed data upon which to base an evaluation.

Data for forms of nitrogen are not routinely collected from Mystic Lake, but WRS added nitrate, ammonium and total Kjeldahl nitrogen (TKN) to the testing program after treatment to provide some background data (Appendix B); cessation of aluminum monitoring provided some budget funds that could be used for this purpose. Nitrate is routinely low, never exceeding 0.3 mg/L and below the detection limit of 0.01 mg/L in nearly all samples during summer of 2011. Ammonium is also low in most

Figure 32. Total nitrogen data for Mystic Lake.

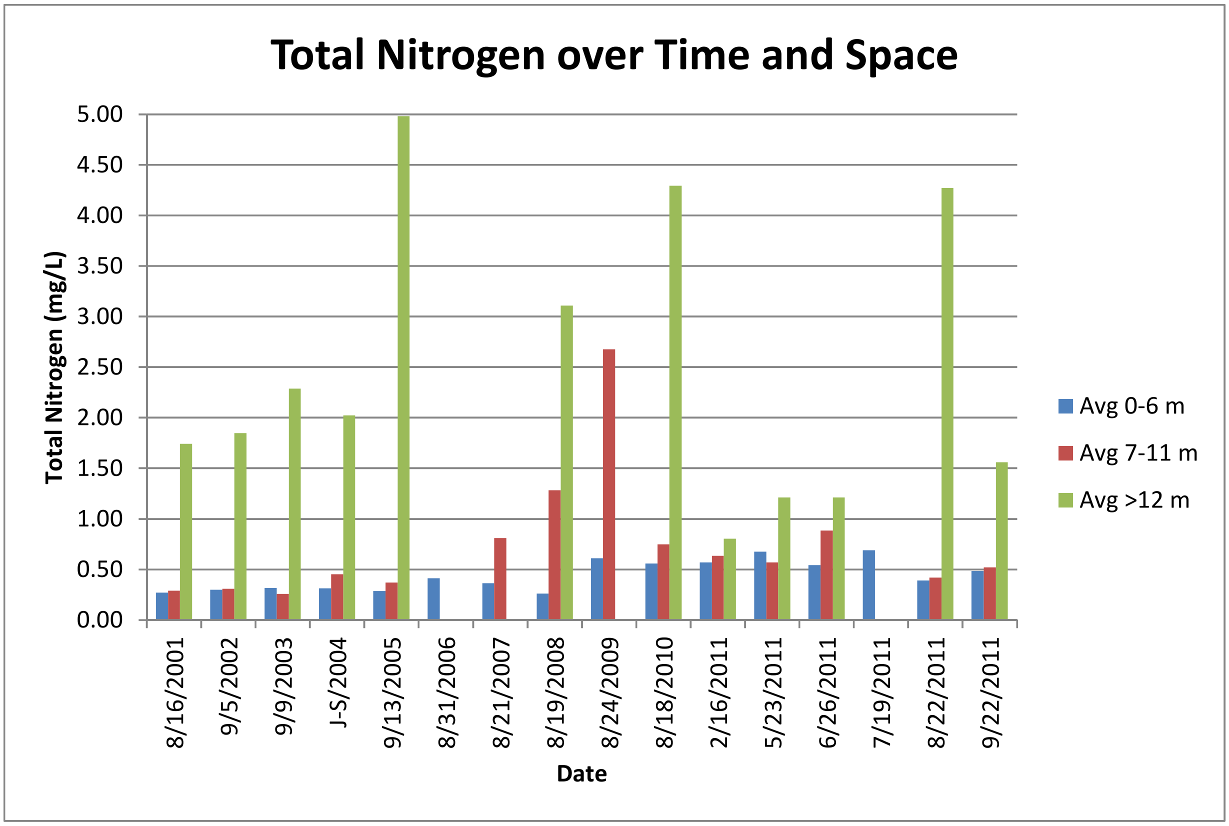
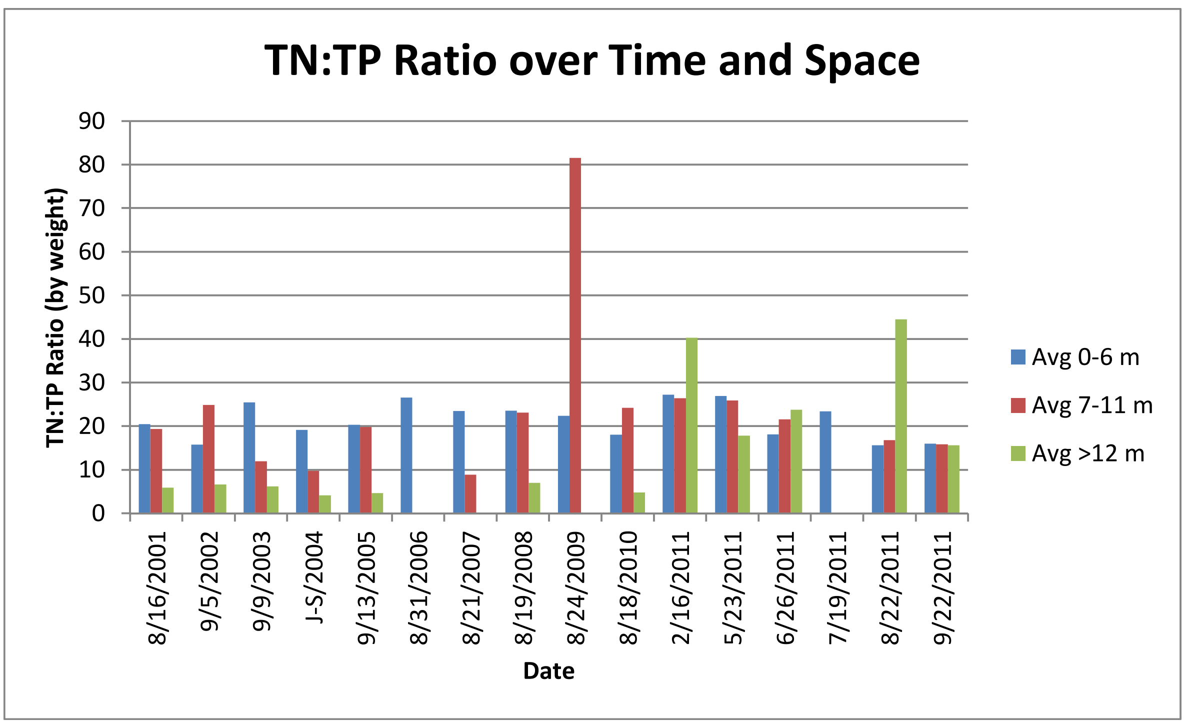


Figure 33. Total nitrogen to total phosphorus ratios for Mystic Lake.



samples; except in June of 2011 and at the deepest point at ML-3 (14 m), values were consistently less than the detection limit of 0.02 mg/L. In June 2011, values ranged from 0.2 mg/L at the surface to 0.8 mg/L at the bottom, suggesting a lot of decomposition and possibly upward movement of ammonium before thermal stratification completely set in. The deepest point at ML-3, which is usually low in oxygen, routinely exhibits high total nitrogen values that reflect ammonium release from sediments. Ammonium is a common decomposition form of nitrogen, but cannot convert to nitrite and then nitrate without oxygen. Values in deep water were as high as 1.5 mg/L, but assuming that past total nitrogen values from deep water are comprised mainly of ammonium, this is lower than usual, except for the August PALS sample.

Most nitrogen is in the TKN form, which includes ammonium and particulate nitrogen, mainly organic matter. As ammonium is low except near the bottom, TKN values reflect organic nitrogen in the upper water layer, and ranged from 0.22 to 0.76 mg/L in water <8 m deep, generally moderate values. Available inorganic nitrogen may remain a factor in favoring cyanobacteria, some of which can utilize dissolved gaseous nitrogen, unlike other algae.

Nitrogen:phosphorus ratios (Figure 34) have not changed much in the upper water layer, but have increased markedly in deep water. While some decrease in nitrogen was observed, a much greater decrease in phosphorus was evident from the treatment, such that N:P ratios have risen from a level that clearly favors cyanobacteria to between 15 and 40 for deep water samples. Values are between 15 and 25 for all surface to mid-depth samples since the treatment occurred. Such ratios will favor green algae during summer, and we do observe a lot of green algae, but there was still a cyanobacterial bloom involving *Aphanizomenon* (a genus that utilizes dissolved nitrogen gas) in August. Also, *Microcystis*, which is not known to be a major user of dissolved nitrogen gas, was found at visible concentrations at the surface in late June; increasing the N:P ratio is beneficial, but will not prevent all cyanobacteria from growing in Mystic Lake. Reductions in phosphorus remain the primary means to limit algal growth overall.

Alkalinity (Figure 35) and pH (Figure 36) have not changed appreciably since treatment with aluminum, but have increased in recent years. Alkalinity in Mystic Lake is higher than average for Cape Cod ponds (CCC 2003), and no one apparently knows why. But the very high alkalinity in deep water at ML-3 is quite striking, possibly higher than in any lake on the Cape. The elevated pH appears related to algal biomass and attendant photosynthesis, which raises the pH by removing carbon dioxide. At the alkalinities observed, the pH should not be higher than 7.0 SU, and for most Cape Cod ponds, pH values tend to between 6.0 and 7.0 SU, with more values closer to 6.0 than 7.0 SU. The elevated pH shifts the forms of inorganic carbon to carbonates from carbon dioxide, which also favors cyanobacteria (Reynolds 2006); this may be one reason that Mystic Lake is still suffering cyanobacterial blooms.

Figure 34. Summer alkalinity in Mystic Lake.

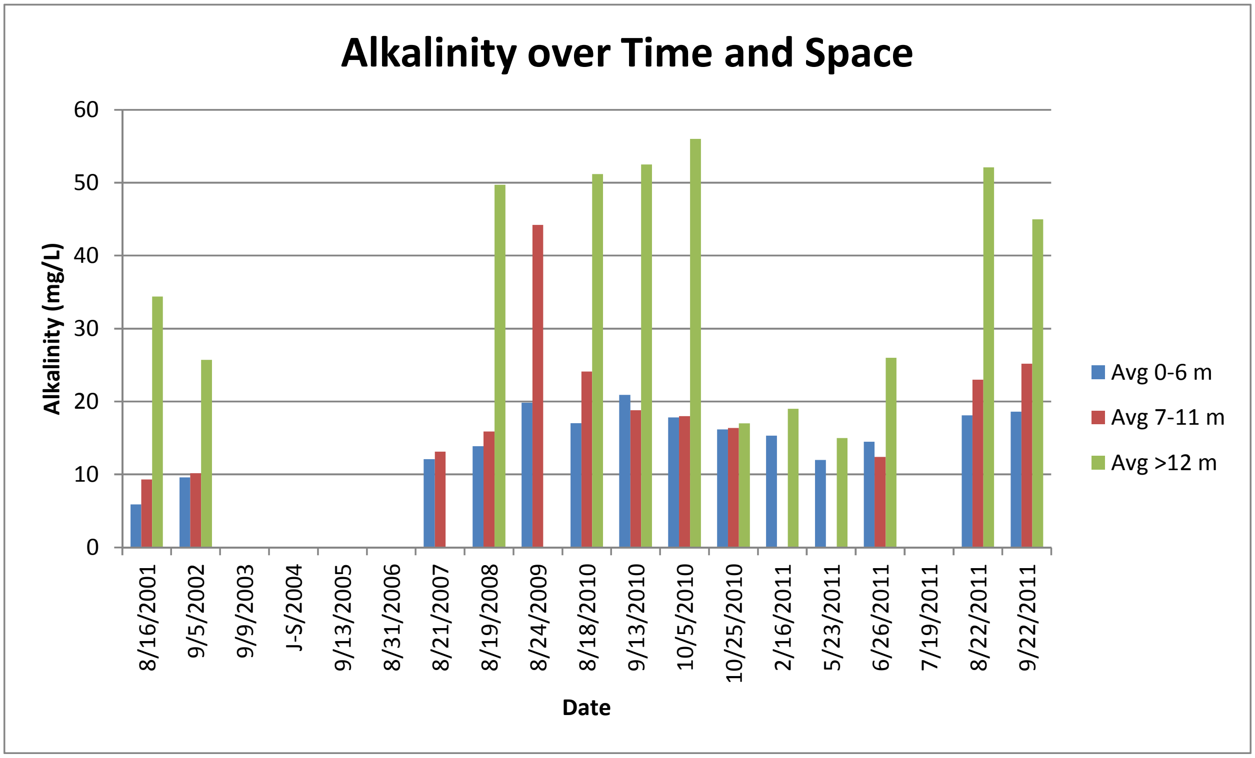
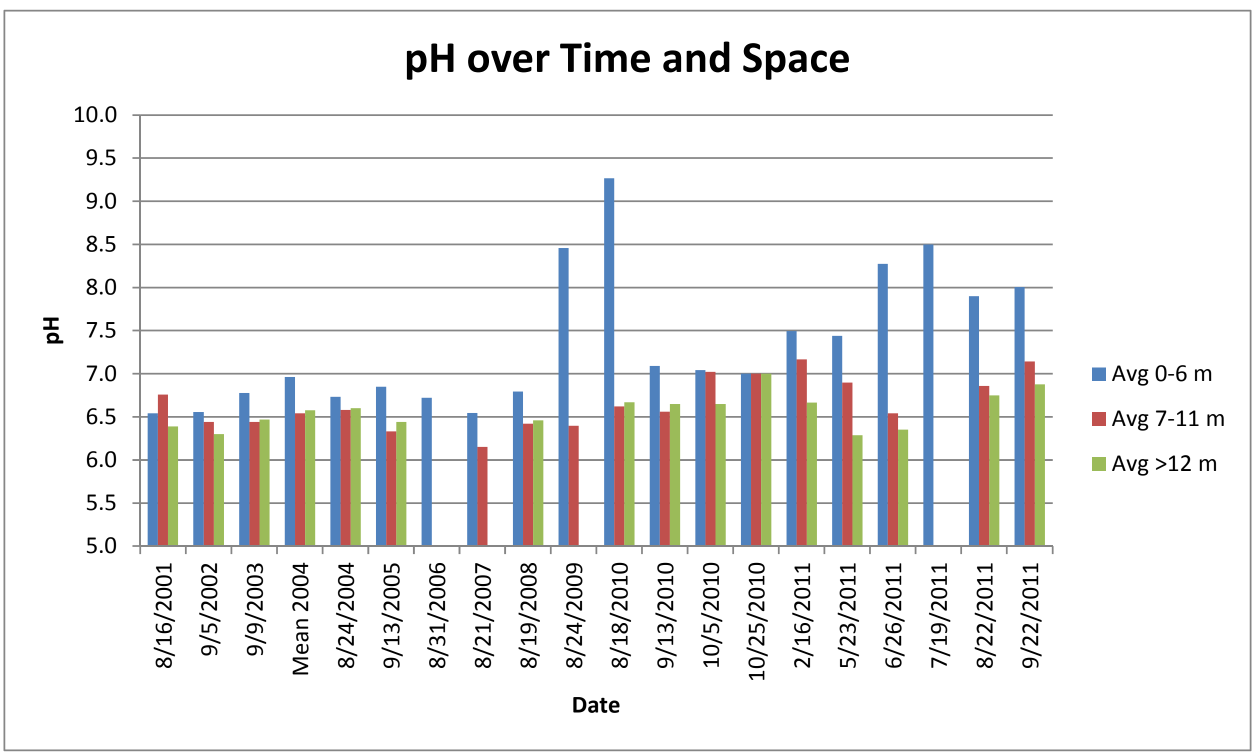


Figure 35. Summer pH in Mystic Lake.



Dissolved aluminum levels were monitored just before treatment, on the last day of treatment, and three weeks later (Appendix B). There were some odd higher values in the pre-treatment sample set; it is surprising that any detectable values were observed, and this remains unexplained. It is possible naturally, but not typical. Values were clearly elevated on the last day of treatment, especially at ML-3, where treatment had finished only minutes before sampling occurred. Almost no detectable values were obtained from samples collected three weeks later, indicating no lasting dissolved aluminum in Mystic Lake waters. Sampling for aluminum was then discontinued, in favor of other water quality features of greater interest, such as forms of nitrogen.

Phytoplankton

Phytoplankton samples were collected with water quality samples, plus on several other dates as convenient, since very little identification of phytoplankton has been performed for Mystic Lake. A single annual sampling for chlorophyll-a at multiple depths at one station provides limited quantitative algal data, but knowledge of the types and relative abundance of algae is very important to understanding ecological impacts and possible use impairment. Complete data from this program are provided in Appendix C. The only other data of which we are aware are from the MA DEP, which examined samples collected after the major mussel die off of 2009. Those samples found high concentrations of *Aphanizomenon* and what was listed as *Oscillatoria*. The species of *Oscillatoria* that form gas vesicles and float were moved to the genus *Planktothrix* in a taxonomic revision, and we use the genus *Planktothrix* to describe those algae. Both *Aphanizomenon* and *Planktothrix* can produce toxins, specifically nerve toxins, the action of which is consistent with the mass mussel deaths and observations on remaining living mussels. No tests for those toxins were completed, however, and sampling response to the die off was very slow, so there are no data for a more careful analysis of potential toxicity as a factor in that die off.

Phytoplankton form the base of the food web, and are therefore necessary to produce fish and wildlife that depend on the lake for food. High productivity is not a negative attribute, as long as the produced algae are consumed by the next trophic level, mainly zooplankton, which are small animals that include several types of crustaceans (especially cladocerans and copepods), rotifers and protozoans. Zooplankton are consumed by small fish, particularly alewife in Mystic Lake, which are in turn consumed by larger fish such as bass. Problems arise when algal production is not utilized in the open water food web, and algal biomass both accumulates in the water column and settles to the bottom, adding to organic sediment and becoming part of the "detrital" food web (governed mainly by bacterial decomposition of organic matter). High algal biomass reduces water clarity, increases pH, causes fluctuations in oxygen levels, and may impart taste, odor and even toxicity to the water. It drives the oxygen demand in deep water that leads to anoxia and phosphorus release where iron is the primary phosphorus binder, thereby fueling more algal production.

Major algal group include cyanobacteria (Cyanophyta), green algae (Chlorophyta), diatoms (Bacillariophyta), golden algae (Chrysophyta), euglenoids (Euglenophyta) , cryptomonads (Cryptophyta) and dinoflagellates (Pyrrhophyta). Samples from July 2010 into October 2011 were analyzed microscopically and organized by major group for display (Figure 37). The tail end of the cyanobacterial bloom of 2010 was detected, followed by dominance by green algae during and after treatment in 2010.

Diatoms become most abundant in the winter, followed by a mixed assemblage in the spring. Green algae became dominant in early summer and remained an important component of the phytoplankton until the end of sampling in October 2011, but there was a pulse of cyanobacteria in August.

The July-August cyanobacteria present in 2010 included a wide variety of genera, including *Aphanizomenon* and *Planktothrix*. Also present were *Aphanocapsa, Dactylococcopsis, Microcystis, Anabaena, Planktolyngbya,* and *Pseudanabaena*. Only the *Aphanizomenon* and the *Pseudanabaena* contributed substantial biomass to the assessed samples, however. The late June 2011 cyanobacteria were mostly *Microcystis*, which were barely evident in open water samples, but did accumulate in wind-driven lines along shore and did form a visible scum at the surface when winds were absent. The August cyanobacteria were mostly *Aphanizomenon*, one of the two types linked to the mussel die offs of 2009 and 2010, but not at very high densities. It is common for species like *Aphanizomenon* and *Anabaena*, which utilize dissolved nitrogen gas, to give way to filamentous non-nitrogen fixers like *Planktothrix, Planktolyngbya* and *Pseudanabaena*. It may be significant that this succession occurred in 2010 prior to treatment, but did not occur in 2011 after treatment. Small amounts of *Microcystis, Planktothrix* and *Pseudanabaena* co-occurred with the *Aphanizomenon,* but did not bloom.

Green algae of importance in Mystic Lake are nearly all of the order Chlorococcales, which tend to prefer high N:P ratios and can discolor the water at bloom densities, but do not form surface scums. These algae are responsible for much of the lower water clarity when cyanobacteria are not blooming in summer. Of these small green algae, the most abundant by far was *Tetraedron*, a unicellular form that makes excellent zooplankton food. If Mystic Lake had an abundant zooplankton population, this alga would not be expected in such high numbers.

Diatoms and golden algae store food as oils, which are metabolized better at cold temperatures, making these algae best suited to dominate in late fall through early spring, which is what is observed in Mystic Lake. The centric diatoms *Aulacoseira, Stephanodiscus* and *Urosolenia* were commonly found, as were the pennate diatoms *Asterionella, Fragilaria, Synedra* and *Tabellaria*. There were some shifts in diatom composition between pre- and post-treatment assemblages, but there are not enough data to determine if this is related to the treatment. These algae can discolor the water brown to olive green, but are generally not summer bloomers and do not represent a significant threat to ecology or human uses of the lake. The only abundant golden algae were *Dinobryon* and *Mallomonas*, two common forms. These can sometimes impart taste and odor to water, but were not that abundant in Mystic Lake.

The dinoflagellates *Ceratium* and *Peridinium* and the euglenoid *Trachelomonas* were occasionally abundant enough to show up on the graph of biomass (Figure 37). These are not ecologically harmful, but tend to be indicators of high levels of dissolved organic substances in the water. Many Cape Cod ponds have high levels of dissolved organic matter, so this is not unique or unusual.

Figure 36. Phytoplankton data from Mystic Lake, 2010-2011.

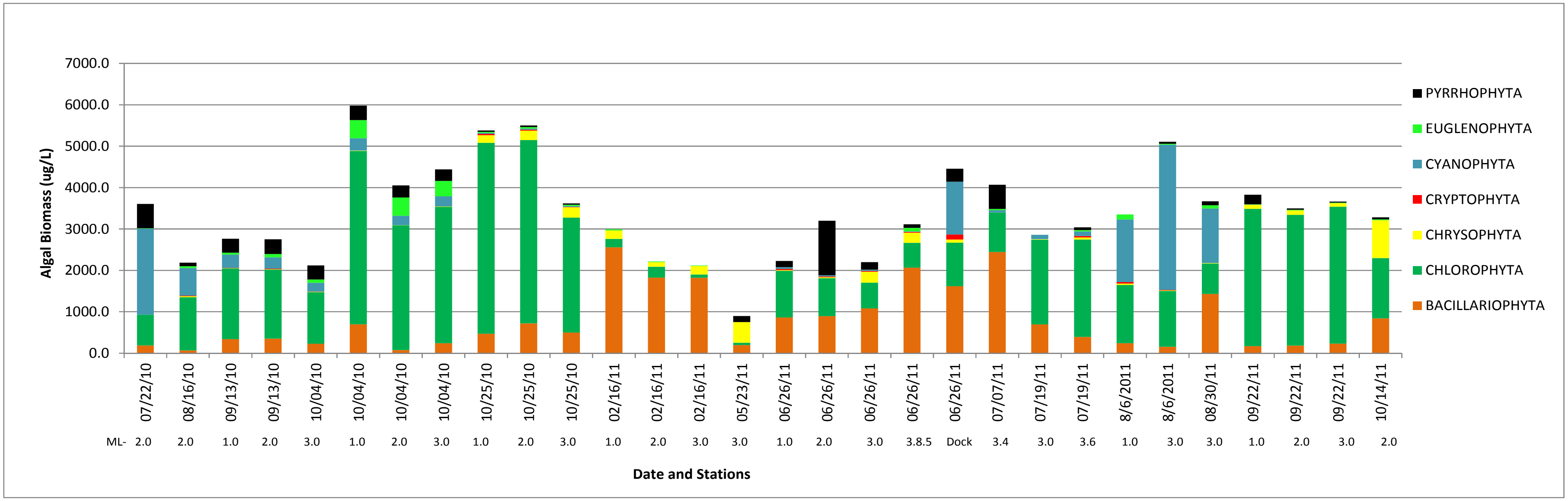
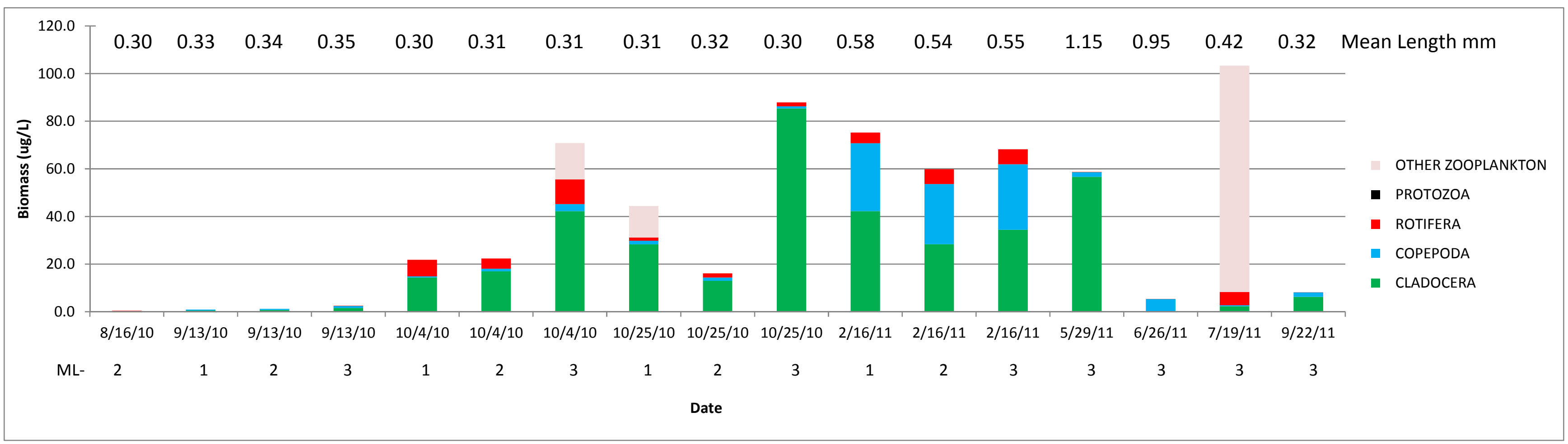


Figure 37. Zooplankton data from Mystic Lake, 2010-2011.



Biomass estimates were almost always in the moderate range (2000 to 5000 ug/L) during this monitoring program. They were undoubtedly higher during the mussel die off of 2009, with the highest chlorophyll levels yet measured in the upper water layer. The highest value since July 2010 was during treatment in October 2010 and was a result of abundant *Tetraedron* (a green alga) at the north end of the lake. The lowest value occurred in May 2011, when mostly diatoms and golden algae were present.

It is too soon to tell if there has been a substantial shift in the phytoplankton, and continued monitoring is warranted. Biomass has not declined substantially, but has been more stable and except for the minor *Aphanizomenon* bloom in August 2011, composition has been favorable. There was evidence of an algal layer in the lake in summer 2011, presumably the *Aphanizomonen* before it moved upward, but it was not detected by algal samples (rather, it was implied by oxygen data). It should also be noted that there are resting stages of most algae in the bottom sediments, including in untreated areas. It is to be expected that there will be some carry over effect of past algal assemblages, but it should be altered over several years as water chemistry dictates which algae thrive and how much biomass can be sustained. No assumption of longer term conditions can be well supported by the amount of data we have at this time.

Zooplankton

As with phytoplankton, zooplankton have not been assessed in any substantial detail in Mystic Lake prior to this project. As Mystic Lake and Middle Pond support an annual anadromous (coming from the sea to spawn) herring (alewife) run, population dynamics will be strongly affected. It has long been known that fish and zooplankton community structures are linked (Mills et al. 1987) and more recent investigations have elucidated the impacts of herring species on lake zooplankton (Post et al. 2008); one would expect few zooplankton and small body size in summer in a lake with an anadromous alewife run, while the winter community would have more zooplankton of larger body size. This is precisely the pattern observed in Mystic Lake (Figure 38), with low summer/fall biomass (<10 ug/L except for July 2011 due to water mite presence) and low mean body length for crustacean zooplankton (0.3 to 0.4 mm). Higher biomass (mostly 40 to 80 ug/L, a moderate range) and larger body length (0.5 to 1.2 mm) characterized winter/spring zooplankton.

The difference in grazing capacity and fish food value between summer/fall and winter/spring zooplankton communities is very large. Filtering capacity increases with the cube of body length, so doubling from 0.3 to 0.6 mm equates to an eightfold increase in water volume filtered by a zooplankter. Biomasses of <10 ug/L are negligible, while values >100 ug/L are large. So Mystic Lake never has a truly large zooplankton community, but the grazing potential can be substantial at times. In May 2011 the *Daphnia* (cladoceran) dominated community at about 60 ug/L and mean body length of 1.15 mm exerted much grazing pressure, resulting in the clearest water observed in years in the lake; one May Secchi reading was close to 9 m.

Maintaining this level of zooplankton presence would produce much clearer water in the summer, and even a lower biomass of large bodied zooplankton can reduce the amount of chlorophyll-a present per unit of phosphorus available (Pace 1984, Stemberger and Miller 2003). Yet the May spawning of alewife in the lake leads to a large population of young of the year alewife and a major decline in biomass and mean body length of zooplankton by early July; there is almost no grazing capacity in summer. The consumption of *Daphnia* and other large zooplankters by the young of the year alewife (and fish of other species as well) fuels fish production, but the effectiveness of the alewife (which filter feed with gill rakers spaced at about 0.3 mm) leads to almost complete decimation of the zooplankton community. Zooplankton resting eggs that sink to the sediment in late spring provide population resurgence in the winter, but there are relatively few zooplankton in Mystic Lake during the summer.

Comparison with Middle Pond

Middle Pond is just south of Mystic Lake and connected to it by a small surface channel; water flows from Mystic Lake into Middle Pond. Much of the ground water outseepage from Mystic Lake is also expected to enter Middle Pond (Eichner et al. 2006). Middle Pond is somewhat smaller in area (105 acres) and shallower (maximum depth = 35 ft) than Mystic Lake, with a volume of just over 1800 acre-feet. A herring run starts at the outlet of Middle Pond, and the spawning alewife enter from this point each spring while the young of the year depart by this route each fall. This herring run joins with the Marstons Mills River about 1000 feet downstream. Water quality data are collected by the IPA under the PALS program with some independent effort, and the Cape Cod Commission and SMAST have supported monitoring and assessment for over a decade.

Temperature and oxygen profiles (unpublished IPA data, WRS data in Appendix D) indicate similarity to parts of Mystic Lake outside treatment area E, the deepest part of Mystic Lake. Where depths of at least 25 ft occur, stratification can occur and phosphorus release is possible. Mixing may occur with strong enough wind and the bottom water layer is relatively thin (<10 ft or 3 m), but that layer is more stable than might be expected. Anoxia can persist in a portion of Middle Pond at a thickness of up to 7 ft (2 m). In 2010, anoxia lasted from late June until mid-September in a bottom layer between 3 and 7 ft thick, increasing over the summer. In 2011, a 5 ft (1.5 m) deep layer was anoxic from the third week of June until the end of August.

Sampling on July 26, 2011 and September 22, 2011 was conducted separately from the Mystic Lake project by WRS with IPA representatives to provide data additional to that provided by PALS sampling in August of each year (Appendix D). Water quality appears very similar to that in Mystic Lake outside Treatment Area E; phosphorus concentrations were elevated, but there is no extremely high value near the sediment-water interface. Middle Pond is more like Mystic Lake at ML-1 or ML-2 than at ML-3, consistent with depth. Nitrate was not detected and ammonium was only detected on the late June sampling date, the same date on which detectable ammonium was found in Mystic Lake outside Area E. TKN levels ranged from 0.33 to 0.86 mg/L, about 0.1 mg/L higher than in Mystic Lake.

PALS data reviewed by Eichner et al. (2006) from 2001-2004 suggest that Middle Pond was in generally desirable condition, but over the last few years this pond has started to behave more like Mystic Lake than Hamblin Pond. Cyanobacteria can grow near the sediment-water interface in this pond and produce the same effect as in Mystic Lake. Phosphorus levels (Appendix D) were elevated in Middle Pond in 2011, potentially supporting blooms to the same level as Mystic Lake, although water clarity was higher in Middle Pond.

Phytoplankton are very similar between Mystic Lake and Middle Pond (Appendix D); the same genera of algae are found in temporally corresponding samples, although the biomass tends to be lower in Middle Pond, yielding higher water clarity by as much as 2 m. But the same cyanobacteria are present, and the same threat to aquatic life and human uses of the pond exist in Middle Pond as for Mystic Lake. There were mussel deaths in Middle Pond in 2009, but these appeared related to the path of surface water moving from Mystic Lake into Middle Pond during and after the die off period in Mystic Lake. However, there was mussel mortality in 2010 that matched a bloom in Middle Pond that appeared to arise at the same time as the one in Mystic Lake. Finally, there were reports of mussel mortality in Middle Pond in 2011, while no significant mortality was observed in Mystic Lake.

Zooplankton community features in Middle Pond (Appendix D) are also similar to those in Mystic Lake, with low summer biomass and declining body length after the alewife spawn. There were no Daphnia found in Middle Pond, but the samples were collected at times when there were none in Mystic Lake either. Given the dominant influence of alewife on both lakes, the similarity is expected.

An examination of mussel populations in Middle Pond suggests that more survived the die off events of 2009 and 2010 than in Mystic Lake, but continued die off in Middle Pond in 2011 remains a threat. Equally disturbing is the high degree of shell erosion observed for live mussels in Middle Pond. Very few mussels did not exhibit at least some erosion, and many had most of outer layer (the periostracum) missing, with the pearly layer exposed and shell structure compromised. E. Nedeau of Biodrawversity undertook an additional survey of Middle Pond mussels in 2011 (Biodrawversity 2011c) and concludes that "In Middle Pond, mussel diversity and density declined within almost all quadrats between 2007 and 2011. Species richness of live mussels dropped from seven to five, and average species richness per quadrat dropped from 5.6 to 2.9. It is likely that two species not detected in 2011—alewife floater and eastern pondmussel—do still occur in Middle Pond, but are only present at much lower densities and are therefore more difficult to detect with quantitative surveys and limited sample sizes….In Middle Pond in 2011, two more late-summer die-off periods were observed after this field survey was completed, and thus current population numbers are likely even lower than reported here. The small and dwindling populations of mussels that remain in these lakes may not be able to recover. We recommend annual monitoring to further understand population trends, creative thinking about ways to preserve the mussel fauna of Middle Pond and Mystic Lake, and a full suite of chemical and biological testing to determine the causes of the recent mussel die-offs."

Comparison with Hamblin Pond

Hamblin Pond was studied in 1992 (BEC 1993), with follow up monitoring and planning for an aluminum treatment in 1993 and 1994. The Town of Barnstable conducted monitoring during and after treatment for 10 years, with a phasing in of CCC, IPA, PALS and SMAST involvement. Much of the history and water quality is covered by Eichner et al. (2006). Hamblin Pond covers about 115 acres and has a maximum depth of 63 ft, with a volume of 3150 acre-feet, almost the same volume as Mystic Lake. It had a duck farm where the current town beach is now for over 30 years until about 1954, and the accumulated excrement over that period created a strong oxygen demand and very high internal phosphorus loading. There were also cranberry bog inputs and a minor amount of runoff and wastewater from a small amount of developed area, but the external load in the 1990s was rather small. The internal load was the cause of severe algal blooms that had occurred for decades and the first aluminum treatment on Cape Cod was planned to address that internal load.

The treatment process has never been documented in writing, which has created a number of misconceptions that persist today. Some of what we know now about how to conduct aluminum treatments was learned at Hamblin Pond, so not all problems were avoidable at that time. The issue was too low a ratio of aluminum sulfate to sodium aluminate (1.4:1), resulting in an increase of pH to over 9 SU. Aluminum is toxic at that pH while reacting, and the shock of a thousand-fold change in pH over about 24 hours may have killed fish as well. The entire target area of 80 acres, over 2/3 of the pond, was treated over two days at full dose (45 g/m^2) with only limited monitoring; virtually everything about the conduct of that treatment would not be part of a current treatment protocol. Many yellow perch, smallmouth bass and rainbow trout were killed just a few days before Memorial Day in 1995.

There were various estimates of mortality, with about 16,500 fish as the best one available (MA DFW unpublished data). As only visible fish were recorded, the number may have been more, but the reaction of most fish when experiencing aluminum toxicity is to come to the surface, so that is where most dead fish would be in such a situation. Reports of mortality of turtles and mussels surfaced later but were undocumented. This has caused considerable controversy over aluminum treatments in some circles; it was assumed that the treatment killed mussels since none were found later. Had there been substantial mussel mortality, there would have been plenty of shells as in Mystic Lake and Middle Pond, but none were noted. A Biodrawversity survey of Hamblin Pond in 2011 did not find any mussels, or even any evidence of past mussel populations. What has frequently not been considered is that there were likely no mussels in Hamblin Pond before the treatment. The 1993 BEC report notes many invertebrates that were present and indicates generally low abundance except for midges (Chironomidae). If mussels were abundant, they would have been mentioned. If there were populations anything like what existed at that time in Middle Pond and Mystic Lake, they would have been considered in treatment planning just as the fish were.

Interestingly, the 1993 BEC report documents the algal community, which was dominated by cyanobacteria during summer and fall. The most abundant phytoplankter was *Aphanizomenon*, the same organism that occurred coincidentally with the major die off of mussels in Mystic Lake. Intense blooms were known from Hamblin Pond for many years, with water clarity <1.2 m (4 ft); it was this lack of clarity and impact on the town beach at Hamblin Pond that prompted the town to have BEC study the pond. It is entirely possible that what happened at Mystic Lake in 2009 and 2010 had occurred at Hamblin Pond for many years earlier, eliminating mussel populations. The urgency to manage Mystic Lake and Middle Pond is underscored by this informed conjecture.

The MA DFW was requested not to stock the pond with trout in spring of 1995, but the stocking was performed anyway. Correspondence in the MA DFW files in the Westborough field office document the resistance of the former Department of Fish and Game to stock trout in Hamblin Pond for many years, citing lack of summer trout water (too warm in the upper water layer, no oxygen in the lower water layer). The fishery agency finally gave in to political pressure to create a put and take trout fishery at Hamblin Pond in the 1960s. Additionally, the yellow perch population was considered stunted in 1980s surveys by the MA DFW, but with rotenone no longer used to reclaim ponds as it was so often in the 1950s and 1960s, nothing had been done to alter community structure. The aluminum treatment did not have to kill fish, but Hamblin Pond has recovered very nicely and is now considered the premier trout pond on Cape Cod.

Note that Hamblin Pond does not have a herring run or a landlocked alewife population, a significant difference from Mystic Lake and Middle Pond that has ramifications for zooplankton and algal grazing that favor Hamblin Pond for greater clarity. Zooplankton in spring 1992 exhibited a high biomass (148 ug/L), with 88 ug/L in *Daphnia*, with a mean body length of 0.56 mm. While biomass did decline to 38 ug/L over the summer, average body length remained high (0.64 mm) and conditions were far better than observed in Mystic Lake. Hamblin Pond zooplankton have not been examined in years, but the potential for high grazing capacity exists.

The treatment of Hamblin Pond worked; phosphorus was dramatically reduced in the deeper waters and movement into the upper waters was curtailed. Phosphorus in the upper water layer has averaged <10 ug/L since the treatment 17 years ago (unpublished town data and PALS data from Eichner et al. 2006). Deep water phosphorus levels have slowly climbed, reaching maxima of about 200 ug/L, but that is much lower than the values up to 1100 ug/L recorded in 1992-1994 (BEC 1993 and unpublished data). Algal chlorophyll-a averages 2.0 ug/L. With lower algae production, the oxygen demand on deep waters during stratification was reduced and oxygen suitable to support trout extended more than 10 ft below the boundary between the upper and lower water levels, sometimes as much as 20 ft.

BEC (1993) estimated an oxygen demand of 860 mg/m^2/day, while Eichner et al. (2006) estimated the demand at that time to be about 219 mg/m^2/day, ascribing a 75% reduction in oxygen demand to the treatment. The threshold for anoxia induced by eutrophication is usually set around 550 mg/m^2/day (Hutchinson 1957), so the change makes quite a difference. Hamblin Pond still experiences deep water anoxia, but the oxygen loss does not extend throughout the deeper water layer. There is an oxygen bulge at the boundary of the upper and lower water layers in Hamblin Pond, suggesting that algae may be accumulating there, and chlorophyll-a data support this contention. However, that layer is normally 9-11 m below the surface and does not interact with the upper waters during summer. Water clarity has averaged >6 m during summer for 17 years since the Hamblin Pond treatment, up from 0.9 m as long ago as 1948, with similar values measured in 1992.

The rehabilitation of Hamblin Pond represents a set of conditions sought for Mystic Lake, and would seem achievable. However, there are differences that must be kept in mind. Hamblin Pond now has fewer watershed influences than Mystic Lake. Hamblin Pond receives ground water already processed by Mystic Lake and Middle Pond, along with a substantial iron load that helps control phosphorus. Hamblin Pond does not have alewife and did not have a major mussel die-off anytime near the treatment. It is deeper and less likely to have deep waters mixed with shallow waters by wind during stratification. Mystic Lake represents the intermediate physical condition in the gradient presented by the Indian Ponds; it is deeper than Middle Pond but has areas that do not strongly stratify, unlike Hamblin Pond. There are simply more factors and complications involved in controlling internal loading in Mystic Lake.

Implications of Results

Considerable commentary has been provided during the presentation of results, but a number of points warrant reiteration, emphasis, and/or expansion.

The condition of Mystic Lake has been improved by the aluminum treatment conducted in fall of 2010, but not to the desired level with regard to water clarity, algal biomass, and types of algae that become abundant. Further improvement may occur without further human action, as the lake must find a new equilibrium condition after so many extreme changes over a period of about two years.

As the treatment occurred at the same time the lake was destratifying, it was not absolutely certain at the time that the reduction in phosphorus concentration and mass were entirely a function of aluminum treatment; oxygenation upon fall turnover should cause phosphorus to decline in the water as a function of precipitation with iron. However, evaluation of phosphorus mass over the following year indicates clearly that internal loading from treated areas has been greatly reduced. The increase in total phosphorus mass in Mystic Lake during the 2011 stratified period (late May through September) is only 18 kg, possibly within the margin of error for the estimate. Assuming it is accurate, the total load estimated from end of summer data is 125 kg, compared to an estimated mean of 196 by the same approach prior to 2008. The reduction of 71 kg is more than the internal load of 60 to 65 kg/yr as adjusted for the difference in lake volume from the estimate by AECOM (2009), although the difference in load is within the expected margin of error. Comparing the pre-treatment mass for the 9-14 m stratum before and shortly after treatment suggests >90% reduction. Comparing pre-treatment phosphorus mass to end of summer 2011 mass for just the 9-14 m stratum, the reduction is 80%. Phosphorus release in the target areas was definitely suppressed to a very large degree.

Considering the potential release from Areas B and E, which may have been under-treated with aluminum due to permit limitation on dose, they account for less than 7 kg over the summer period. This is 39% of the perceived increase in total lake phosphorus mass over that period, but is not a very large load. Over half the increase occurs between February and May of 2011(19 kg) when there is no anoxia at the bottom and iron bound phosphorus should not be released. And the majority of the change during the stratified period occurs between late May and late June (10 out of 18 kg), suggesting a fairly fast rate of change that then tails off substantially for the rest of the summer. It appears that most of the increase in phosphorus mass is occurring outside of the period of stratification and outside the treated areas, but the exact source is not known. Alternative sources and mechanisms have been explored; some combination of watershed loading and internal load from untreated areas is certainly possible, but the ongoing processing of all the dead mussel phosphorus remains a possible dominant influence. All of 2011 was wetter than usual, and the decay of millions of dead mussels in 2009 and 2010 is expected to have added phosphorus to surface waters that the aluminum would not efficiently remove. Pre-treatment shallow water phosphorus levels were moderately high, and remained so after treatment. It is not clear how long this situation will last.

The aluminum treatment has resulted in a phosphorus distribution that differs from what is normally observed in stratified lakes with anoxic bottom waters and substantial internal loading via release of phosphorus by iron. There is more phosphorus in the upper waters than the lower strata. As the lower strata involve less volume than the upper ones, the concentration may still be higher in deeper waters, but the difference is far less than usual. Algal blooms are unlikely to be supported by phosphorus that might become available from deeper waters after the treatment, but enough phosphorus remains in the upper waters to support blooms already. Blooms have not been severe since treatment, but the dramatic shift observed after treatment of Hamblin Pond has not been observed.

The overall mass of phosphorus in Mystic Lake has been reduced, hitting the adjusted target of 86 kg shortly after treatment and holding that level through winter, but rising substantially in the spring and somewhat more during summer. Resulting summer water clarity was higher than in 2009 and 2010, but not as high as desired, with a detectable presence of cyanobacteria in late June and a mild cyanobacteria bloom occurring in August. Conditions were more stable through summer than in recent years. It is uncertain if the phosphorus level will increase or decrease or oscillate going forward, as we do not know the exact mechanism by which increases are occurring. It seems likely that as the large pulse of phosphorus added by dead mussels is processed, overall phosphorus mass in the lake will decline, but that is only informed speculation. Many influences on Mystic Lake are superimposed on the treatment and aftermath, with seasonal and annual variations potentially important and not well quantified in this case. A “wait and see” approach is justifiable in this case, with appropriate monitoring to detect changes and elucidate important mechanisms, including watershed inputs.

The impact of cranberry bogs is more visible than for other land uses, with discharges of water high in phosphorus occurring after harvest and sometimes in spring if flooding is practiced to provide frost protection. However, the actual mass of phosphorus is usually not all that high relative to what is in the receiving lake already. Yet the inputs add up over time, becoming part of the sediment phosphorus reserves and contributing to the internal load. Anything that can be done to reduce inputs from bogs is worth pursuing to protect the lake in the future.

Other watershed inputs should not be ignored. Storm water runoff is not typically a large source to Cape Cod ponds, given very sandy soils and limited storm water collection and discharge systems. Yet some storm water does reach lakes, and slopes around Mystic Lake are great enough to allow runoff during intense storms, so attention to land uses and phosphorus control is warranted. The trend of people retiring to the Cape and trying to recreate suburban landscapes has the potential to increase loading from fertilizers, although high phosphorus in lawn fertilizers is being phased out, so perhaps this will be less of a threat in the future.

Waste water remains an issue all over Cape Cod, and Mystic Lake is no exception. Understanding potential phosphorus inputs from waste water is more difficult than for nitrogen, as phosphorus is adsorbed to soil particles and movement is strongly affected by the oxygen status of the ground water. Eichner et al. (2003) provide a detailed discussion of ground water movement and waste water issues at Mystic Lake, concluding that phosphorus will arrive mainly from the northwest and that break out is still years away, so this does not appear to be a major issue now. Assessment of inputs through seepage surveys in the lake is advised, to more definitively assess current inputs and provide a baseline for future comparison.

Another influence from outside the lake is climate. Whatever one might choose to believe about the causes of climate change, its existence is undeniable, and it does not appear that there is anything we can do about it in the short term. Warmer average temperatures and greater variability in weather, with greater extremes, are now to be expected. Surface water temperatures have been warmer in recent years, and these favor cyanobacteria. Variability in wind will induce variability in the depth of stratification, with 2008 vs. 2009 standing out as a stark example. The only apparent difference between the very favorable conditions of 2008 and the disaster of 2009 is the depth at which the lake stratified. Any interpretation of trends relating to nutrient loads must also consider the variability induced by climate change.

Even without a major decrease in surface water phosphorus concentrations, the treatment did alter N:P ratios to favor algae other than nitrogen fixing cyanobacteria. There was still a mild bloom of nitrogen-fixing *Aphanizomenon* in 2011, but with so many resting stages in the sediment, continued presence by algae of the past decade or more is to be expected until the changed water chemistry fosters a longer term change in species composition. Nitrate and ammonium remain low in surface waters, but the shift in N:P ratio should gradually foster a move away from the objectionable algae that have been so problematic in recent years. Algal quality may be just as important as algal quantity in this system.

Concern was initially expressed by the NHESP that the aluminum treatment would reduce the fertility of Mystic Lake and compromise support of what was considered an outstanding mussel community prior to the 2009 die off. In addition to concern over direct toxicity, this was a second reason given for not allowing the treatment to proceed in 2010. As the purpose of the aluminum treatment is to reduce internal recycling of phosphorus and limit the potential for algal growth, this concern might seem justified. However, there is no evidence to support any contention that high abundance of algae, without consideration of quality, is beneficial to mussel populations. It was assumed that since the lake had such high quality mussel populations, any observed blooms were at least not a negative influence, and might represent a food resource. This assumes that lakes are stable environments, however, not subject to directional changes over time, which is contrary to what is known about the eutrophication process (Mattson et al. 2004, Cooke and Welch 2005).

The eutrophication of Mystic Lake appears to have shifted the algal community, as is nearly always the case, to one dominated by potentially toxic cyanobacteria. While it remains unknown if toxic cyanobacteria were actually the cause of the mussel die off, there is uncertainty over what constitutes desirable food resources for mussels (Strayer et al. 2004, Kesler et al. 2007). Again, algal quality may be at least as important as algal quantity. Abundant cyanobacteria do not appear beneficial to mussels.

Additionally, the increased abundance of cyanobacteria signals conditions potentially deleterious to mussels, conditions linked to the decline of mussel populations in North America (Strayer et al. 2004). Lakes are not static environments, and increased fertility tends to increase variability in other conditions (e.g., pH, dissolved oxygen) that can result in adverse impacts on many biological components (e.g., invertebrates, fish, birds) of aquatic systems. Surface water conditions in 2008 were as favorable as any recorded for Mystic Lake, but there was a very dense band of algae and strong anoxia in deep water; that the algae and low oxygen water remained there appears to have been a function of weather, and the mussels were fine in 2008. The algal layer that formed in 2009, however, was mixed into the upper water column by virtue of shallow stratification followed by windy conditions, along with water low in oxygen and probably high in ammonia. Something in that water killed millions of mussels, with algal toxins the most logical cause, albeit an unproven one.

Mussels may indeed be limited by food quantity in very infertile lakes, but that does not make high fertility desirable, and the assumption that whatever conditions are observed in a lake will remain indefinitely is to be avoided. While we should not seek to manage aquatic systems without adequate information, neither should we avoid action based on unjustified assumptions.

The aluminum treatment did not harm any notable biological components of Mystic Lake. It is entirely possible that damage was done to populations of midge larvae and oligochaete worms, the primary invertebrates found in anoxic sediments, but recovery has been found to be rapid in other systems (Smeltzer et al. 1999, Mattson et al. 2004, Cooke et al. 2005). Zooplankton populations are sometimes harmed by the actual flocculation process during treatment, but with such low quantities in Mystic Lake, this is not a significant issue. Direct monitoring of fish and mussels indicated no mortality and no behavioral problems that suggest adverse impacts. The mechanisms normally applied in aluminum treatments for the last decade to minimize adverse impacts on lake biota have proven effective and reliable. Low alkalinity lakes can be safely treated, although Mystic Lake is not really a low alkalinity lake, an unusual circumstance for a Cape Cod pond.

Dose estimation remains an inexact science. Dose evaluation methods have improved markedly over the last 15 years, but there is still uncertainly related to how well lab tests and simulations represent field conditions. With impact minimization precautions available, and the cost of treatment heavily dependent on permitting and labor (not just chemical cost), it is best to use the highest dose justifiable to maximize the probability of success.

Since the mortality caused by the treatment of Hamblin Pond during a spring treatment, there has been pressure to do treatments differently to avoid undesirable consequences. Deep injection of aluminum was conducted at Ashumet Pond in September 2001. The deep injection proved cumbersome and did not address phosphorus in the upper waters, and has not been attempted since. The late summer – fall treatment timing was adopted, however, for Cape Cod treatments, with the support of the MA DFW on grounds of minimizing impacts to spawning alewife, mussel reproduction, and other potential biological impacts. It has also been hinted that late summer/early fall treatments interfere less with fishing tournaments.

Long Pond was treated in fall 2007, and three treatments other than Mystic Lake were conducted in fall 2010 on Cape Cod. Since the efficiency of phosphorus stripping in upper waters is low, it may be more advantageous to treat in the spring, however, before internal recycling has released phosphorus during the summer and a portion of that phosphorus has been circulated throughout the water column. Each potential treatment case should be evaluated individually; there may be good reasons to avoid spring treatment in individual cases, but a blanket policy of requiring fall treatment is not appropriate.

Along the same lines as treatment timing, other aspects of treatment should not be subject to policies or mandates based on limited experience and conjecture about possible impacts. Dose determination and treatment timing have been addressed, and are perhaps the most critical aspects of treatment that have been subjected to questionable restrictions. However, treatment features such as depth of application, chemicals used, application mode and duration of treatment may also be subject to consideration in permitting processes. Monitoring should indeed be required before, during and after treatment, but the provisions of monitoring may vary by case and should be tailored to the circumstances. Thresholds for various mitigation actions or cessation of treatment may be warranted, but flexibility in Orders of Conditions should be maintained to the greatest degree possible.

Further Management Options

While watershed management efforts are desirable and can proceed at any time, further management action in Mystic Lake should probably be delayed until lake condition has had time to stabilize. Too many confounding influences are interacting at this time to make accurate predictions or even explain what is being observed. Continued monitoring is the most important action to be taken in the next year. Total phosphorus and total Kjeldahl nitrogen should suffice to characterize nutrient chemistry, with measurements at 2 m intervals at ML-3 in early May, late June, August and late September. Results at ML-1 and ML-2 were comparable to those at matching depths at ML-3, so a single sampling point should suffice in Mystic Lake.

Temperature and dissolved oxygen profiles should be collected at the same time as water sampling, but may be collected more often to track thermal features and development of deep water anoxia. Algal tracking should also be performed with all water quality sampling, but samples might be collected at other times as well, as visual appearance dictates. If an oxygen bulge forms at an intermediate depth, algal sampling should be conducted to determine if there is an algal layer associated with it. Zooplankton sampling in just May and September, at the same time as water quality sampling, should be sufficient in Mystic Lake to characterize that biological component of this system.

Several supplemental studies are recommended. Iron should be analyzed in the water quality samples to determine if there is any issue with iron availability in Mystic Lake. A seepage survey (Mitchell et al. 1988) should be conducted to determine if current projections of input locations and low phosphorus are correct. This would involve placing seepage meters at key locations, particularly along the northwest shore, to measure ground water entry or exit from the lake. It would also involve collection of ground water just before it enters the lake with a littoral interstitial porewater sampler, with testing for dissolved phosphorus, nitrate and ammonium, dissolved iron, and pH. Loading via ground water, including waste water inputs, can be assessed in this manner.

Re-testing of available sediment phosphorus in treatment areas B and E, the areas where a dose of >50 g/m^2 would have been recommended if not for permitting restrictions. It would be helpful to know if under-treatment is a factor in continued phosphorus loading.

Additional aluminum treatment does not appear warranted at this time. The primary problem appears to be lingering elevated phosphorus levels in surface waters, not release from bottom sediments, and aluminum treatment is not very efficient at removing phosphorus from water at the doses normally applied. In water treatment facilities, aluminum doses on the order of 20 mg/L are used to clear the water, while the dose in lake treatments is usually kept <5 mg/L to avoid possible toxicity.

Oxygenation of deep water would provide benefits on several levels, but release of phosphorus from the bottom sediments may not be a major factor in phosphorus loading at this time. Circulation systems that mix the entire water column may disrupt cyanobacteria growth, but the risk of mixing poor quality deep water does not appear justified at this time. Injection of pure oxygen into deep water, mainly in treatment area E, would improve water quality in that area, and oxygen should spread laterally to at least some other deep areas that have low oxygen. However, this is a costly approach that may not be necessary to control phosphorus loading from deep water. Still, the very strong correlation between Secchi depth and the depth at which low oxygen is encountered (Figure 10) makes it very tempting to suggest oxygen input to deep waters.

Dredging to remove undesirable sediment remains a very costly and technically difficult approach not likely to be practical at Mystic Lake. No other management action has proven success and appropriate documentation to warrant recommendation for application in Mystic Lake at this time.

Hydrilla management, while not the subject of this report, should continue. If water clarity is improved over time, expansion of submergent aquatic plant growths would be expected, and hydrilla represents a major threat to biological integrity in Mystic Lake. The current approach of monitoring with benthic barrier placement on dense patches and hand pulling for sparse growths remains appropriate.

Middle Pond monitoring should be conducted as with Mystic Lake. All the monitoring elements listed above are appropriate. Concerns that Middle Pond may experience continued algal blooms, mussel kills and deterioration of other aspects of this valued aquatic system are well founded. A transplant experiment, whereby mussels from Middle Pond would be moved to Hamblin Pond, is recommended. Any concern that Hamblin Pond may not be suitable for mussel colonization is not based on sufficient data to delay such an experiment. However, algae and zooplankton assessment of Hamblin Pond may help assuage fears about mussel survival, and would supplement ongoing PALS sampling of Hamblin Pond that has indicated generally desirable conditions in that pond.

All data for the Indian Ponds should be reviewed in early 2013 to further evaluate conditions and mechanisms in these ponds and to determine what management steps are most appropriate.

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Appendix A. Monitoring Log for Mystic Lake Treatment

Date	Time	Monitoring Activity	Lake Area (see map)	Depth (ft)	pH (SU)	Alk (mg/L)	Temp (C)	DO (mg/L)	Observations (floc, mortalilty, behavior, other) - use map as needed to indicate locations of observations
9/21/2010	11:00 AM	Pre-trtmt meeting	All						Attending: Ken Wagner, Dom Meringolo, Marie Esten, Robert Nichols, Ed Eichner, Betsy (from IPA), two other ACT staff - covered treatment plan, monitoring program, expected timeframe, permit conditions.
	12:00 noon	Pre-trtmt monitoring	A (max depth = 23 ft)	0	7.1	23			SDT = 7.3 ft
			A	20	6.8	19			
	12:15 PM	During trmt	A	0	7.0	12			Collected in floc zone
			A	20	6.8	20			Minimal floc on bottom at time of sampling
	12:25 PM	Pre-trtmt monitoring	C (max depth = 27 ft)	0	7.0	20			SDT = 7.5 ft
				20	7.2	18			
	12:50 PM	During trtmt	A	3-24					Video of bottom and floc accumulation; no problems evident, floc forming well and fairly even on bottom
	1:00 PM	During trtmt	A	0	6.8				
	2:00 PM	45 min post-trtmt	A	0	6.9	21			Floc out of upper water column, evident from 4.5 m to bottom
	2:15 PM	During trtmt	C	0	6.3	18			Sample in active floc zone, just treated
			C	24	7.1	20			No floc evident, not settled to this depth yet
	2:30 PM	Near end of trtmt	East of C	5-26					Video of bottom from deep to shallow
	2:35 PM	At end of trtmt	C	0	7.8				Sampled flushing from delivery line; apparently has more aluminate than alum, but not outside of required range.
	2:44 PM	Just after trtmt	C	0	7.0	15			
				0	7.0	18			Also took video of floc zone.
	2:45 - 3:00:00 PM	Post-trtmt visual survey	Shoreline downwind of A & C						Complete survey of all shoreline downwind of areas A and C; one dead white sucker found, partly decayed, clearly dead before treatment began.
	3:50 - 5:05 PM	Post-trtmt survey	C	0	7.0				SDT = 9.0 ft, wind moderate from SW; videos of floc zone to shore along two transects. No indication of mortality of stress. Fish observed swimming in treated area.
				24	7.0				No floc in water, all settled to bottom. Some floc as shallow as 5 ft, but no significant deposits until 17 ft depth.
			A	0	7.1				SDT = 8.7 ft, wind moderate from SW; videos of floc zone to shore along two transects. No indication of mortality of stress. Fish observed swimming in treated area.
				21	6.9				SDT = 8.7 ft, wind moderate from SW; videos of floc zone to shore along two transects. Some floc in water as shallow as 3 ft, patchy at 13 ft, even coverage from 17 ft to 23 ft depth.
	5:22 PM	Reference area check	E (46 ft)	0	7.2	23	19.3	8.4	Area at south end of lake, as far from A and C as possible. SDT = 7.9 ft
				42	6.8	18	10.4	0.3	Note: DO meter moving very slowly to 0, but recent IPA monitoring indicated no oxygen at this depth.

Date	Time	Monitoring Activity	Lake Area (see map)	Depth (ft)	pH (SU)	Alk (mg/L)	Temp (C)	DO (mg/L)	Observations (floc, mortalilty, behavior, other) - use map as needed to indicate locations of observations
9/22/2010	8:00 AM	Water quality check	E	0	7.1	23	19.2	8.4	SDT = 8.0 ft
				42	6.8	18	10.4	0.3	Z, T, DO: 0,19.2,8.4; 4,19.2,8.4; 8,18.8,6.8; 12,11.2,0.3; 14,10.4,0.3
	8:40 AM	Water quality and survey	C	0	7.0	22			No dead or stressed fish along downwind (N) shore or obsd in treatment zone by underwater video.
				24	7.0	20			Strong wind from SW; two video transects from 24 ft to 3 ft. Floc transformation evident since trtmt.
	9:20 AM	Water quality and survey	A	0	7.1	21			No dead or stressed fish along downwind (N) shore or obsd in treatment zone by underwater video.
				20	7.0	19			Strong wind from SW; two video transects from 26 ft to 5 ft. Floc transformation evident since trtmt.
	1:50 PM	Water quality check	E	0	7.3	23	19.6	7.5	Reference area for A & C, S end of lake. Attempted video, but inadequate light at 45 ft.
				42	6.8	17	10.4	0.3	Z, T, DO: 0,19.6,7.5; 2,19.3,7.3; 4,19.1,6.7; 6,18.9,6.3; 8,18.4,4.6; 10,13.6,2.1; 12,11.5,0.3; 14,10.4,0.3
	2:00 PM	WQ check and video survey	D	0	7.1				Two videos in area D, not yet treated, at 10-29 ft.
	2:30 PM	WQ check and video survey	A	0	7.1				Two videos in trtmt area 1 day post treatment, plus shoreline survey - no mortality or stress
	3:06 PM	WQ check and video survey	C	0	7.2				Two videos in trtmt area 1 day post treatment, plus shoreline survey - no mortality or stress
	1:00 - 4:00 PM	Mussel plot examination by E. Nedeau	D and A	17					Separate data to be provided: impressions from EN: plots near area E with gravel, no floc (not yet treated), almost all mussels in plot located, most open, but not all responsive to touch; plots near A with more silt, covering by floc (treated yesterday with intention to deposit floc on these plots), almost all mussels in one surveyed plot located, similar activity level and responsiveness to plots near area E. Remaining plots to be surveyed tomorrow.
9/23/2010	9:00 AM	Shoreline survey	A and C						No dead or stressed fish along shore.
	10:00 AM	Water quality check	E (47 ft)	0	7.2	19	19.7	8.5	Z,T,DO: 0,19.7,8.5; 2,19.7,8.5; 4,19.7,8.5; 6,19.7,8.5; 8,19.7,8.5; 10,12.9,1.4; 12,10.5,0.2; 14,10.3,0.2 SDT=8.1 ft
				45	6.8	62	10.3	0.2	Strong H2S smell to bottom water, gray color; beaker full stirred and left for 1 hr, turned orange from Fe.
	10:20 AM	Video scan	E	0-45					Thick suspended solids layer at 11 m; obscures all light, video camera goes dark even with lights on.
	10:30 AM	Meeting with H. Hobart and C. Thut							Discussed project progress, pond history, mussel kills of 2009 and 2010.
	11:42 AM	Water quality check	C (27 ft)	0	7.0	22	20	8.5	Z,T,DO: 0,20.0,8.5; 2,19.8,8.4; 4,19.5,8.0; 6,19.2,7.5; 8,18.7,1.7 SDT=8.7 ft
				25	7.1	18	18.7	1.7	Wind moderate from the north
	12:05 PM	Video scan	C	21-27					V1 in 21-27 ft in trtmt area, V2 at 25-32 ft from trtmt area into untreated area to south. Mussels observed in each transect at all depths to 30 ft, but few appeared live.
	12:30 PM	Water quality check	A (23 ft)	0	7.1	22	20.5	8.7	Z,T,DO: 0.20.5,8.7; 2,19.9,8.7; 4,19.5,7.8; 6,19.4,7.2; 7,19.2,2.3 SDT = 8.2 ft
				20	7.0	18	19.2	2.3	Wind moderate from the north
	12:50 PM	Video scan	A	20-23					V1 near mussel plots, some mussels observed protruding through sediment/floc, some appeared live and filtereing. V2 through middle of area A, floc gray, moving into sed, yellow perch observed.
	1:20 PM	Shoreline survey	A and C						No dead or stressed fish observed.
	5:15 PM	Shoreline survey	A and C						No dead or stressed fish observed.
	5:37 PM	WQ check/bottom survey	C	0	7.2				No dead fish observed, floc merging nicely with sediment, orginal sediment features visible.
	5:43 PM	WQ check/bottom survey	A	0	7.3				No dead fish observed, floc merging nicely with sediment, orginal sediment features visible.

Date	Time	Monitoring Activity	Lake Area (see map)	Depth (ft)	pH (SU)	Alk (mg/L)	Temp (C)	DO (mg/L)	Observations (floc, mortalilty, behavior, other) - use map as needed to indicate locations of observations
9/24/2010	8:36 AM	Water quality check	E	0	7.1	21			SDT = 9.5 ft
				45	6.8	62			
	8:49 AM	Water quality check	A	0	7.1	22			SDT = 9.9 ft
				20	7.0	22			
	8:58 AM	Video scan	A	20-23					Floc mostly gray, a little grainy, merging well with sediment. Winds strong from SW.
	9:15 AM	Water quality check	C	0	7.1	23			SDT = 9.9 ft
				25	7.0	19			
	9:20 AM	Video scan	C	20-27					Floc mostly gray, a little grainy, merging well with sediment. Winds strong from SW.
	9:35 AM	Shoreline survey	A and C						No dead or stressed fish observed. A bale of hay has been staked in the lake on the west side of the cove directly north of Area C. It was not present yesterday.
9/27/2010	8:24	Pre-treatment pH monitoring	B	22.6 ft total depth					
				0	7.25	16.9	Surface temp. 64.4 F		
				20	7.0	16.6			Secchi Depth = 2.24m
	8:48	Video monitoring	B						ACT beginning to add first load to area B; no stressed organisms observed
	8:55	Pre-treatment pH monitoring	D	35.6 ft total depth					
				0	7.25	16.7			Secchi depth = 2.38m
				34	6.75	37.9	64.4 F Surface temp.		
		Video monitoring	D						Difficult to view deep sediments, too dark
	9:25	Pre-treatment pH monitoring	Control (Area	36.4 ft total					
				0	7.25	17.1	Surface temp 66.2 F		
				35	6.5	40.0			Secchi Depth 2.19m
	9:50	During treatment pH monitorin	B	0	7.25	16.6			Weather 0-5 knots from North, overcast, foggy and damp
				20	7.0	16.9			
	10:03	Video monitoring	B						Active treatment, could see floc settling, no evidence of stressed organisms
	10:04								Treatment of Area D begins
	10:08	DO profile	Area F just outside D						Treatment progressing in Area D
							Calibrated meter		
				2			20.3	8.48	
				6			20.2	8.82	
				10			20.1	8.40	
				13			19.6	6.37	
				15			19.2	5.34	
				19			14.1	<1	
				22			12.2	<1	
				25			12.3	<1	
				28			12.3	<1	

Date	Time	Monitoring Activity	Lake Area (see map)	Depth (ft)	pH (SU)	Alk (mg/L)	Temp (C)	DO (mg/L)	Observations (floc, mortalilty, behavior, other) - use map as needed to indicate locations of observations
	10:30	Shoreline Survey	Down-wind - southern end of lake						2 dead fish; one near red barn along beach (6-12")
									Second caught against bulkhead with gray shed (6-12") most likely yellow perch
	11:24	During treatment pH monitorin	D	25 ft total					Surface sample taken in small floating foam patch
				0	7.25	16.8			
				22	6.75	16.7			
	11:40	Video monitoring	B	20 ft depth					Floc settling, no distressed organisms
	11:42								ACT loading chemical, wind calm, overcast and warm
	11:48	During treatment pH monitorin	D	30 ft total					Floc visible in surface and deep sample
				0	7.25	16.5			
				25	7.0	16.7			
	12:02	Video monitoring	D - along north western edge						No stressed organisms, Cormorant successfully fishing in treated area.
									Floc settling fairly evenly along bottom
									North east wind, <5 knots
	12:17	Video monitoring	D - along North western edge						Little floc outside treatment area
	13:13								ACT just left with new load of chemical for Area D
									3 more loads to go for today
									Wind 0-5knots from North
	13:15	During treatment pH monitorin	B	25 total depth					
				0	7.25	16.2			
				20	7.0	16.8			
	13:41	During treatment pH monitorin	D	33 ft total					
				0	7.25	16.4			
				31	6.5	21.8			
	13:52								Act goes to truck for more chemical
	14:04	Fish survey of south end of lake (downwind).							No additional dead fish. Surface pH = 7.25. Noted a lot of particles in
									water - they were there this morning before treatment as well.
									Appear to be algae as they are green
	14:08								ACT back in treatment area D - 2 more loads to go
	14:17	During treatment pH monitorin	D	35 ft total depth					Collected in area ACT already applied on to double check the
				33 ft	6.75	25.2			last alkalinity sample as it was lower than value obtained during pretreatment at bottom
									Floc in sample
	15:00								ACT leaves with another load for area D, only 1 more load for today

Date	Time	Monitoring Activity	Lake Area (see map)	Depth (ft)	pH (SU)	Alk (mg/L)	Temp (C)	DO (mg/L)	Observations (floc, mortalilty, behavior, other) - use map as needed to indicate locations of observations
	15:03	During treatment pH monitorin	B	24 ft total depth					ACT applying in Area D
				0	7.25	17.1			
				20	7.0	16.8			
	15:18	During treatment pH monitorin	D	37 ft total					Active treatment area D, collected sample in Floc zone
				0	7.25	16.4			
				35	6.75	38.7			
	15:37								ACT heads back for final load
	15:49	Post treatment pH monitoring	B	27 ft total depth					ACT loading
				0	7.25	16.7			Secchi depth = 2.03 m
				24	6.75	16.9			
	16:00								ACT leaves with last load
	16:18	Video survey	B						Floc appears to be mostly settled, seems like there is a thin layer of silt or dead algae-
									light wispy balls throughout deep treatment area that make application look patchy.
									Only 1/2 dose applied here today as well as Area D and a very small portion of Area F along
									the western edge of area D. Surveyed into 10 ft of water near downwind section B, near island,
									some floc outside treatment area along this downwind edge and mixed in with mussels
									Not able to determine status of mussels.
	16:31	Post treatment Control	F	35 ft total depth					Location: behind island (to south)
				0	7.25	15.8			Secchi depth: 2.0 m
				30	6.5	17.4			
	16:42	Post treatment pH monitoring	D	30 ft total					Secchi depth 2.0m
				0	7.25	16.3			ACT completed application for the day.
				25	7.0	16.9			
		Video survey	D						Downwind video survey in area D - can't see the bottom too dark
									No stressed organisms observed
	17:00	Visual survey downwind (southern end of pond)							No dead fish, 2 fish noticed earlier are gone.
		Downwind of area B - along N side of island							No stressed organisms
	17:30	Left site							

Date	Time	Monitoring Activity	Lake Area (see map)	Depth (ft)	pH (SU)	Alk (mg/L)	Temp (C)	DO (mg/L)	Observations (floc, mortalilty, behavior, other) - use map as needed to indicate locations of observations
9/28/2010	8:45 AM	General survey	A.B,D,E						From surface, everything looks good; no stressed or dead organisms other than long-dead mussels
		Treatment begun	E						
	9:30 AM	Water quality check	A	0	7.1	20			
			B	0	7.1	18			
			D	0	7.1	18			
			E	0	7.1	16			Away from treatment area
				42	6.8	63			SDT = 8.1 ft
			E	0	7.1	16	20.3	8.5	In treatment area
				42	6.8	55	10.4	0.3	Z,T,DO: 0,20.3,8.5; 2,20.2,8.5; 4,19.9,7.9; 6,19.7,7.1; 8,19.5,6.5; 10,15.3,0.3; 12,11.2,0.3; 14,10.4,0.3
	10:30 AM	Video survey	E	0-47					Floc layer accumulating at thermocline at about 28 ft; suspended sediment layer at 35 ft (dark). Visible as two lines on sonar.
	10:40 AM	Water quality in barge path	E	0	6.9				1 minute post-trtmt
				0	7.1				3 minutes post-trtmt
	10:50 AM	Video survey near barge	E	0-42					Some northward drift, about 100 yard to N of discharge, due to wind from SW. Floc up to 3 inches thick, snowy, forming very nicely.
	11:00 AM	Shoreline survey	E, D west side	0.3					No floc in water <24 ft deep, no stressed or dead fish. Wind keeping floc away from shore.
			E,D east side	0-32					No floc in water <20 ft deep, no stressed or dead fish. Wind keeping floc away from shore.
	11:40 AM	Water quality check	E	0	7.1				
	11:42 AM	Video survey	E, south side	0-42					No floc in <17 ft of water, light, patchy accumulation from 17-20 ft, layer from 20 ft deeper, substantial at 24 ft.
	11:55 AM	Water quality check	A	0	7.1				
	Noon	Video survey	A	0-23					Floc blended with sediment, almost indistinguishable
	1:45 PM	Video survey	C	0-27					Floc blended with sediment, almost indistinguishable
	1:50 PM	Shoreline survey	C						No stressed or dead organisms related to treatment
	2:05 PM	Water quality check	C	0	7.3				
	2:10 PM	Video survey	F	20-37					Untreated as of yet, too dark to see bottom >30 ft,soft sediment with limited features
	2:15 PM	Water quality check	F	0	7.2	21			SDT = 8.1 ft
				35	6.6	20			
	2:20 PM	Water quality check	D-E border	0	7.1				
	2:25 PM	Went for gas, weather check							
	2:50 PM	Video survey	E	0-47					Done treatment about 2:20 PM; No floc in water <20 ft deep, most unsettled floc at >30 ft water depth. Dark below 33 ft.
	3:00 PM	Water quality check	E	0	7.1				SDT = 8.0 ft
				45	6.7				
	3:10 PM	Shoreline and deeper survey	E	0-30					No stressed or dead organisms related to treatment.
	3:29 PM	Water quality check	F	0	7.0				2 minutes post-treatment
				0	7.0				3 minutes post-treatment
	3:37 PM	Video survey	F	0-32					Floc forming well in treatment zone, drift to north, some landing in shallow areas (7-20 ft on slope leading to island) as a consequence of increasing wind. Island is only about 100 yds from edge of trtmt zone
	3:48 PM	Water quality check	F	0	7.2				30 minutes post-treatment; floc forming well but settling slower than other days, due to wind.
	4:30 PM	Video survey	E	0-47					Still some light floc in water 10-20 ft deep, most at >25 ft.
	4:40 PM	Water quality check	E	0	7.1				Almost 2 hr post-treatment, still moderate wind from south
				45	6.8				
	5:00 PM	Complete shoreline survey	All	0-8					No dead of stressed organisms observed. Windblown bluegreen accumulation noted in Thut (eastern) cove. Also large cloud of filamentous green algae in Thut cove. Light floc at 13 ft just S of island, near edge of Area F treatment zone, but no floc visible at <10 ft. Cut across lake in several places - no fish at surface.
	6:02 PM	Water quality check	F	0	7.1				1 hr post-treatment

Date	Time	Monitoring Activity	Lake Area (see map)	Depth (ft)	pH (SU)	Alk (mg/L)	Temp (C)	DO (mg/L)	Observations (floc, mortalilty, behavior, other) - use map as needed to indicate locations of observations
9/29/2010	8:10 AM	Shoreline survey	A						One dead yellow perch (seen by crew yesterday out of treatment area), one adult alewife with cut and hook marks (used as bait); no adult alewife should be in lake at this time. Conductivity = 33 umhos/cm
	8:10 AM	Water quality check	D	0	7.1	14			SDT = 8.7 ft.Treating Area D starting at 8:30 AM.
				24	7.1	15			
			E	0	7.1	18	20.7	8.6	SDT = 8.9 ft
				45	6.8	48	10.3	0.3	Z,T,DO: 0,20.7,8.6; 2,20.5,8.6; 4,20.3,8.3; 6,20.3,7.7; 8,19.1,6.0; 10,16.1,0.4; 12,11.9,0.3; 14,10.3,0.3
	9:00 AM	Video survey	E						No floc in water until dark zone; not discernible there. Light wind from SW. No stressed/dead fish.
	9:20 AM	Shoreline survey	E-west						No stressed/dead fish
			E-east						No stressed/dead fish
	9:30 AM	Video survey	E into D						No floc evident until in Area D, being treated. Wind from SW. Light floc at surface near border, much more at greater depths (25-34 ft).
	9:40 AM	Water quality check	D	0	7.0				
				0	7.0				
				0	7.0				
				30	7.1				
	9:50 AM	Shoreline survey	A, B, C						No stressed/dead fish other than those observed first thing in the morning.
	10:02 AM	Water quality check	A	0	7.2				
	10:10 AM	Video survey	A						Area treated 8 days prior; no evidence of floc.
	12:30 PM	Water quality check	A	0	7.2				
			D	0	7.1				
			E	0	7.1				
	12:40 PM	Video check	E	12 - 35					18 - 35 ft , then 12 - 34 ft
	1:05 PM	Video check	D	10-30					Vertical profile about 40 min post-trtmt, outside treatment area to W in <12 ft of water, then in trtmt zone on W edge at 25-28 ft, then out of trtmt zone to W at 14-18 ft,then outside treated area of D at 24-29 ft. Video coming out of D onto submerged spit from island at 10 ft, again from 24 to 10 ft, light floc evident. Video of floc formation.
	2:30 PM	Mussel collection	Middle Pond						Collected 4 spp of mussel for toxin testing
	4:01 PM	Water quality check	E	0	7.3				No trtmt; SDT = 8.0 ft.
			D	0	7.2				2 hr post-trtmt; SDT = 7.6 ft.
			A	0	7.4				No trtmt; SDT = 7.5 ft.
			F	0	7.3				Being treated
			F	0	7.1				Being treated
			F	0	7.1				Being treated
	4:20 PM	Video survey	F	10 - 30					floc formation and depostion, some accumulation in shallow water to NE due to moderate wind.
	5:00 PM	Shoreline survey	A, B, C	0 - 6					No stressed/dead fish

Date	Time	Monitoring Activity	Lake Area (see map)	Depth (ft)	pH (SU)	Alk (mg/L)	Temp (C)	DO (mg/L)	Observations (floc, mortalilty, behavior, other) - use map as needed to indicate locations of observations
9/30/2010	7:20 AM	Water quality check	F	0	7.2	18			Z/T: 0/21.4; 2,21.2; 4,20.9; 6,20.5; 8,19.3; 10,18.3; 11,11.8
				35	6.6	20			SDT = 8.1 ft
			E	0	7.2	18			Z,T,DO: 0,21.1,9.1; 2,20.9,9.1; 4,20.7,8.9; 6,20.3,8.3; 8,19.6,6.4; 10,15.2,0.6; 12,10.6,0.4; 14,10.2,0.3
				45	6.6	32			SDT = 8.1 ft
			F	0	7.1				In plume behind barge while treating.
	8:00 AM	Shoreline survey	All	0-8					1 recently dead WS near Linxholm launch; YOY alewife observed swimming near shore, esp. S end; no hydrilla observed anywhere.
	8:41 AM	Water quality check	F	0	7.2				
			E	0	7.3				
	8:45 AM	Video survey	E	0-35					Floc forming well, dark at bottom (35-39 ft), light wind from SW.
	9:20 AM	Tour with Rob Gatewood	Mostly E						Visual and video survey
	9:40 AM	Video survey	D into B	7-28					Groups of WS near pinch point between B and D. No appreciable floc on sediment at >17 ft
	10:20 AM	Water quality check	F	0	7.1				Portion of Area F treated 30 min before measurement
				10	7.2				Portion of Area F treated 30 min before measurement
				0	7.0				Treated <5 min before measurement
	11:10 AM	Water quality check	F	0	7.2				Finished treating Area F 10 min prior to measurement
				0	7.3				No floc evident.
				0	7.1				Last portion of Area F treated.
				30	6.9				
	11:20 AM	Video survey	F						Some floc up to 100 yards downwind of treatment area; S side of island very steep, goes from 0 to 35 ft in <100 yds, floc accumulating in water >17 ft deep, some evidence of floc in water as shallow as 7 ft. Floc forming and settling well in treatment area, but wind is blowing some toward island.
	11:30 AM	Visual survey	All						Toured entire lake, shoreline out about 200 ft; 1 dead WS floating near spit under water off island to W; otherwise no evidence of stress or mortality, and that WS appeared to have been dead for several days. Crew now treating in Area E, last load of day. Will avoid windy weather later today and tomorrow, finish Monday.
	12:30 PM	Water quality check	E	0	7.1				Just finished treatment in this area, will resume Monday, Oct 4. Wind not gusting to near 20 mph, stopping treatment.
10/1/2010									Wind predicted, treatment cancelled for the day
10/4/2010									Too windy to work
10/5/2010	7:40 AM	Water quality check	E	0	7.1	18			SDT = 7.0 ft. Winds from N/NE
				8	7.0	22			Z,T,DO: 0,18.4,8.7; 2,18.5,8.6; 4,18.5,8.6; 6,18.5,8.6; 8,18.5,8.6; 10,18.5,8.6; 12,11.3,0.3; 14,10.8,0.3
				14	6.8	62			
			D	0	7.1	19			SDT = 7.0 ft
				10	7.0	20			Z,T,DO: 0,18.4,9.0; 2,18.5,8.8; 4,18.5,8.7; 6,18.5,8.7; 8,18.5,8.7; 10,18.4,5.7
			A	0	7.0	20			SDT = 6.9 ft
				6	7.0	16			Z,T,DO: 0,18.5,9.0; 2,18.5,8.6; 4,18.5,8.6; 6,18.5,8.6; 8,18.6,2.4
	8:00 AM	Shoreline survey	E & D-West						1 dead white sucker in D-West, washed on shore, not recently dead. No dead fish along south shore, despite 3 days of wind from north.
	8:45 AM	Water quality check	E	0	7.0				In treated area about 10 min after treatment
	9:05 AM	Video check	E	0-47					Floc in water, forming well, not as dark as bottom as last week (fewer particles in water); no stressed fish observed.
	9:50 AM	Water quality check	E	0	6.9				In floc 1 min after treatment.
	10:10 AM	Water quality sampling	A, D/F, E	0-bottom					Collected samples for lab analysis. Areas A (ML-1) and D/F (ML-2) not treated since last week. Area F (ML-3) still being treated, likely to have gotten some floc in at least deeper samples. Collected 2 extra samples at surface and at 10 m, right in treatment area, floc observable in samples.
	11:25 AM	Visual survey	All						Done treating Area E, doing small area on western edge of Area D not well treated previously. No dead fish, no stressed organisms observed all around lake.
	1:05 PM	All treatment concluded							

Appendix B. WRS Water Quality Results and Other Selected Water Quality Data.

Temperature (C) and Dissolved Oxygen (mg/L)																											
	8/18/2010			9/13/2010			10/5/2010			10/25/2010			2/16/2011			5/23/2011			6/26/2011			7/19/2011			9/22/2011		
Station	Temp (C)	DO (mg/L)	DO (% sat)	Temp (C)	DO (mg/L)	DO (% sat)	Temp (C)	DO (mg/L)	DO (% sat)	Temp (C)	DO (mg/L)	DO (% sat)	Temp (C)	DO (mg/L)	DO (% sat)	Temp (C)	DO (mg/L)	DO (% sat)	Temp (C)	DO (mg/L)	DO (% sat)	Temp (C)	DO (mg/L)	DO (% sat)	Temp (C)	DO (mg/L)	DO (% sat)
ML-1.0	27.8	8.6		21.2	8.6	97	18.5	9.0	95	13.8	9.5	92	4.0	16.4	126.5	15.9	10.4	106	21.6	10.6	120	26.0	9.64		21.4	9.8	112.3
ML-1.2	27.8	8.6		21.0	8.3	93	18.5	8.6	91	13.7	9.5	92	4.2	16.1	124.9	15.9	10.4	107	21.5	10.7	121	25.5	9.47		20.9	9.7	110.1
ML-1.4	27.6	7.3		21.0	8.0	89	18.5	8.6	91	13.6	9.8	94	4.1	15.7	121.2	15.9	10.3	106	21.4	10.6	120	25.1	9.86		20.2	9.6	107.7
ML-1.6	26.8	0.7		20.9	7.8	88	18.5	8.6	91	13.6	9.9	94	4.1	14.8	114.8	15.3	9.4	95	19.1	8.9	96	20.8	11.17		20.1	9.0	100.6
ML-1.8	17.5	0.2		18.8	0.4	2	18.6	2.4	25	13.6	9.9	94	4.7	5.2	40.8	14.2	7.0	69	15.2	0.4	4	15.9	0.04		19.7	5.7	62.7
ML-2.0				21.3	8.3	94	18.4	9.0	95	13.9	9.4	91	3.8	16.6	127.7	15.9	10.3	106	21.9	10.5	120				21.1	9.9	112.8
ML-2.2				21.2	8.2	92	18.5	8.8	93	13.9	9.4	91	4.3	16.6	129.1	15.9	10.4	107	21.6	10.7	121				20.6	9.8	110.2
ML-2.4				21.1	8.1	91	18.5	8.7	92	13.8	9.4	90	4.0	16.7	128.8	15.8	10.4	106	21.2	10.6	119				20.2	9.6	107.3
ML-2.6				21.0	7.8	87	18.5	8.7	92	13.7	9.3	89	3.9	16.6	128.3	14.6	10.5	104	19.0	9.3	100				20.1	8.7	97.0
ML-2.8				19.5	0.5	5	18.5	8.7	92	13.6	9.3	89	3.8	15.7	120.8	14.1	8.6	85	15.2	3.8	38				19.9	7.5	83.7
ML-2.10				12.5	0.2	2	18.4	5.7	60	13.6	9.3	89	3.7	11.3	86.9	13.8	7.2	70	13.7	0.3	3				15.7	0.4	4.3
ML-3.0	26.6	8.3		21.4	8.7	98	18.4	8.7	92	14.2	9.2	89	1.6	15.9	115.0	15.9	10.5	108	22.0	10.5	120	26.4	9.2		20.7	9.9	112.4
ML-3.2	26.7	8.3		21.2	8.6	97	18.5	8.6	91	13.7	9.0	86	4.2	15.9	123.9	15.2	11.1	112	21.5	10.7	122	26.0	9.3		20.5	9.9	111.6
ML-3.4	26.6	8.1		21.1	8.4	95	18.5	8.6	91	13.6	8.8	85	3.9	16.2	125.2	14.8	10.9	109	21.2	10.8	121	25.8	9.4		20.2	9.5	105.9
ML-3.6	24.0	2.8		21.0	8.2	92	18.5	8.6	91	13.6	8.8	84	3.9	16.2	124.5	14.4	10.5	104	19.2	9.7	104	20.1	11.3		20.1	8.6	95.8
ML-3.8	16.7	0.2		19.6	2.6	29	18.5	8.6	91	13.6	8.8	84	3.8	16.2	124.5	14.1	9.7	96	15.0	4.6	46	15.7	3.1		20.0	7.8	86.6
ML-3.10	13.4	0.2		12.6	0.3	3	18.5	8.6	91	13.6	8.7	83	3.8	12.6	96.8	13.8	8.1	79	13.6	0.3	3	13.5	0.0		14.5	0.4	3.5
ML-3.12	12.2	0.3		10.5	0.1	1	11.3	0.3	3	13.5	8.5	81	4.1	4.4	34.0	13.2	3.4	33	12.6	0.0	0	12.5	0.0		12.3	0.0	0.0
ML-3.14	10.3	0.3		10.1	0.1	1	10.8	0.3	3	10.4	0.5	4	4.5	0.6	5.0	12.5	0.0	0	12.3	0.0	0	11.9	0.0		11.8	0.0	0.0

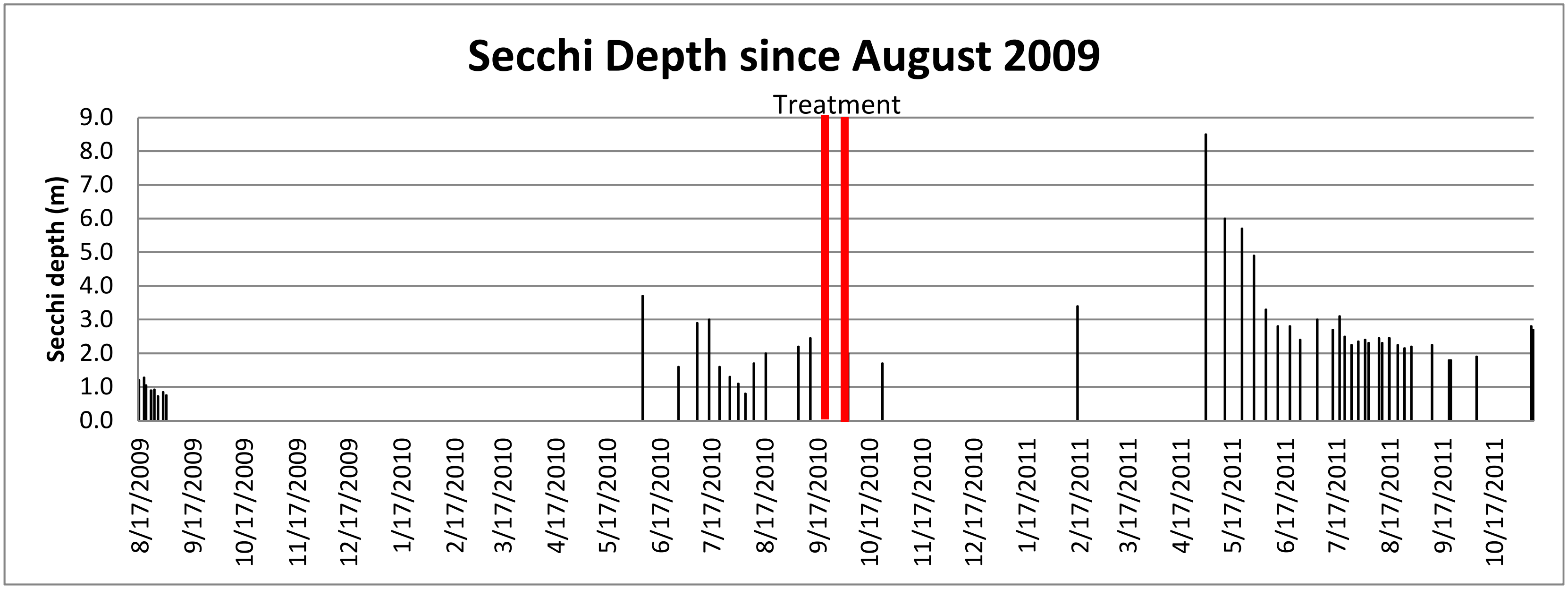
pH																				
Station	8/16/2001	9/5/2002	9/9/2003	Mean 2004	8/24/2004	9/13/2005	8/31/2006	8/21/2007	8/19/2008	8/24/2009	8/18/2010	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	8/22/2011	9/22/2011
ML-1.0	6.4											7.2	7.0	7.0	7.3	7.3	8.6			8.1
ML-1.2												7.2	7.0	7.0	7.3	7.4	8.5			8.1
ML-1.4												7.1	7.0	7.0	7.3	7.3	8.5			8.0
ML-1.6												7.0	7.0	7.0	7.2	7.1	7.4			7.8
ML-1.8												6.5	7.0	7.0	6.6	6.7	6.3			7.1
ML-2.0												7.2	7.1	7.0	7.4	7.4	8.7			8.3
ML-2.2												7.1	7.0	7.0	7.6	7.4	8.8			8.2
ML-2.4												7.0	7.0	7.0	7.8	7.4	8.6			8.0
ML-2.6												7.0	7.0	7.0	7.8	7.3	7.4			7.7
ML-2.8												6.7	7.0	7.0	7.6	7.0	7.0			7.5
ML-2.10												6.4	7.0	7.0	7.0	6.6	6.3			6.9
ML-3.0	6.4	6.5	6.8	7.0	6.8	6.9	6.9	6.4	6.7	9.4	9.3	7.2	7.1	7.0	7.5	7.6	8.5	8.5	7.8	8.3
ML-3.1																				
ML-3.2												7.1	7.1	7.0	7.6	8.0	8.5			8.2
ML-3.3	6.8	6.6	6.8	6.9	6.6	6.9	6.5	6.7	6.9	9.4	9.3								8.0	
ML-3.4												7.0	7.1	7.0	7.6	7.7	8.4			7.9
ML-3.5																				
ML-3.6										6.6		7.0	7.1	7.0	7.6	7.5	7.4			7.7
ML-3.7																				
ML-3.8												6.8	7.0	7.0	7.6	7.3	6.7			7.5
ML-3.9	6.8	6.4	6.4	6.5	6.6	6.3		6.2	6.4	6.3	6.6								6.9	
ML-3.10												6.4	7.1	7.0	7.1	7.0	6.4			6.9
ML-3.11								6.1		6.5										
ML-3.12												6.7	6.7	7.0	6.7	6.5	6.3			6.9
ML-3.13	6.4	6.3	6.5	6.6	6.6				6.5		6.7								6.8	
ML-3.14						6.4						6.6	6.6	7.0	6.7	6.0	6.4			6.9
Station	8/16/2001	9/5/2002	9/9/2003	Mean 2004	8/24/2004	9/13/2005	8/31/2006	8/21/2007	8/19/2008	8/24/2009	8/18/2010	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	8/22/2011	9/22/2011
Avg 0-6 m	6.5	6.6	6.8	7.0	6.7	6.9	6.7	6.5	6.8	8.5	9.3	7.1	7.0	7.0	7.5	7.4	8.3	8.5	7.9	8.0
Avg 7-11 m	6.8	6.4	6.4	6.5	6.6	6.3		6.2	6.4	6.4	6.6	6.6	7.0	7.0	7.2	6.9	6.5		6.9	7.1
Avg >12 m	6.4	6.3	6.5	6.6	6.6	6.4			6.5		6.7	6.7	6.7	7.0	6.7	6.3	6.4		6.8	6.9

Alkalinity (mg/L)																			
Station	8/16/2001	9/5/2002	9/9/2003	J-S/2004	9/13/2005	8/31/2006	8/21/2007	8/19/2008	8/24/2009	8/18/2010	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	8/22/2011	9/22/2011
ML-1.0											20	20	18	15	12	16			10
ML-1.2											24	18	18			14			
ML-1.4											21	18	16			14			
ML-1.6											21	18	17			13			
ML-1.8											13	16	16			12			12
ML-2.0											24	19	16	16	12	14			14
ML-2.2											23	18	16			14			
ML-2.4											20	18	16			14			
ML-2.6											20	18	17			14			
ML-2.8											20	19	16			13			
ML-2.10											18	20	18			13			10
ML-3.0	5	10					12	14	21	17	28	18	14	15	12	16		18	13
ML-3.1																			
ML-3.2											16	16	15			15			
ML-3.3	6	10					12	14	19	17								18	
ML-3.4											18	16	15			15			
ML-3.5																			
ML-3.6									20		16	17	16			15			
ML-3.7																			
ML-3.8											17	17	16			12			
ML-3.9	9	10					13	16	22	24								23	
ML-3.10											26	18	16			12			14
ML-3.11							13		66										
ML-3.12											50	48	15			19			42
ML-3.13	34	26						50		51								52	
ML-3.14											55	64	19	19	15	33			48
Station	8/16/2001	9/5/2002	9/9/2003	J-S/2004	9/13/2005	8/31/2006	8/21/2007	8/19/2008	8/24/2009	8/18/2010	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	8/22/2011	9/22/2011
Avg 0-6 m	6	10					12	14	20	17	21	18	16	15	12	15		18	19
Avg 7-11 m	9	10					13	16	44	24	19	18	16			12		23	25
Avg >12 m	34	26						50		51	53	56	17	19	15	26		52	45

Conductivity (umhos/cm)							
Station	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	9/22/2011
ML-1.0	119	124	120	111	88	111	91
ML-1.2	119	117	113	111	88	112	91
ML-1.4	119	115	120	113	88	111	91
ML-1.6	119	117	117	114	88	110	91
ML-1.8	121	124	121	135	90	143	93
ML-2.0	118	128	120	110	88	112	91
ML-2.2	118	113	120	114	88	112	91
ML-2.4	119	111	118	115	88	111	91
ML-2.6	119	125	126	115	88	109	91
ML-2.8	118	117	119	115	89	110	91
ML-2.10	124	127	127	116	89	116	139
ML-3.0	110	110	118	114	88	112	91
ML-3.2	113	122	120	115	88	112	91
ML-3.4	112	115	113	114	88	111	91
ML-3.6	112	118	124	114	88	110	91
ML-3.8	112	119	122	114	88	110	91
ML-3.10	117	127	127	116	89	116	118
ML-3.12	134	138	129	122	91	139	149
ML-3.14	159	168	134	165	104	146	162
ML-3.0 in floc		119					
ML-3.10 in floc		130					

Dissolved Aluminum (mg/L)			
Station	9/13/2010	10/5/2010	10/25/2010
ML-1.0	0.094	0.086	<0.050
ML-1.2	0.075	0.085	<0.050
ML-1.4	0.073	0.092	<0.050
ML-1.6	0.088	0.082	<0.050
ML-1.8	0.130	0.096	<0.050
ML-2.0	0.140	0.170	<0.050
ML-2.2	0.160	0.220	<0.050
ML-2.4	0.110	0.270	<0.050
ML-2.6	0.110	0.170	<0.050
ML-2.8	0.099	0.130	0.090
ML-2.10	0.080	0.094	0.054
ML-3.0	0.090	0.120	0.170
ML-3.2	0.170	0.180	0.052
ML-3.4	0.180	0.150	<0.050
ML-3.6	0.160	0.200	<0.050
ML-3.8	0.170	0.290	0.060
ML-3.10	0.180	0.190	<0.050
ML-3.12	0.280	0.240	0.078
ML-3.14	0.230	0.850	<0.050
ML-3.0 in floc		0.530	
ML-3.10 in floc		0.750	

Recent Secchi Disk Transparency (m)																						
Station	8/17/2009	8/20/2009	8/21/2009	8/24/2009	8/26/2009	8/28/2009	8/31/2009	9/2/2009	6/7/2010	6/28/2010	7/9/2010	7/16/2010	7/22/2010	7/28/2010	8/2/2010	8/6/2010	8/11/2010	8/18/2010	9/6/2010	9/13/2010	10/5/2010	10/25/2010
ML-1											2.7		1.6		1.1	0.8			2.2	2.2	2.1	1.7
ML-2																				2.4	2.1	1.8
ML-3	1.2	1.3	1.1	0.9	0.9	0.7	0.9	0.8	3.7	1.6	2.9	3.0	1.6	1.3	1.1	0.8	1.7	2.0	2.2	2.5	2.0	1.7
Station	2/16/2011	4/24/2011	5/2/2011	5/7/2011	5/13/2011	5/15/2011	5/21/2011	5/23/2011	5/27/2011	5/30/2011	6/6/2011	6/13/2011	6/16/2011	6/20/2011	6/26/2011	7/6/2011	7/7/2011	7/10/2011	7/15/2011	7/19/2011	7/22/2011	7/26/2011
ML-1	2.3	4.4	8.1	9.7	6.0	7.4	7.2	6.2	4.8	4.7	3.1	2.8	3.5	2.8	2.4	3.2	3.5	3.8	2.6			
ML-2	2.9							6.2							2.4							
ML-3	3.4		8.5		6.0			5.7		4.9	3.3	2.8		2.8	2.4	3.0			2.7	3.1	2.5	2.3
Station	7/30/2011	8/3/2011	8/5/2011	8/11/2011	8/13/2011	8/17/2011	8/22/2011	8/26/2011	8/30/2011	9/11/2011	9/21/2011	9/22/2011	10/7/2011	11/8/2011	11/9/2011							
ML-1												1.7										
ML-2												1.7										
ML-3	2.4	2.4	2.3	2.5	2.3	2.5	2.3	2.2	2.2	2.3	1.8	1.8	1.9	2.8	2.7							



Chlorophyll a (ug/L) - with phaeophytin (which is substantial in some cases, mainly in deep water)												
Station	8/31/1998	8/16/2001	9/5/2002	9/9/2003	J-S/2004	9/13/2005	8/31/2006	8/21/2007	8/19/2008	8/24/2009	8/18/2010	8/22/2011
ML-1.0												
ML-1.2												
ML-1.4												
ML-1.6												
ML-1.8												
ML-2.0												
ML-2.2												
ML-2.4												
ML-2.6												
ML-2.8												
ML-2.10												
ML-3.0	7.5	5.0	5.4	5.5	5.4	6.2	3.8	2.5	3.1	31.8	8.5	3.5
ML-3.1												
ML-3.2												
ML-3.3		4.6	7.2	5.4	5.4	8.6	3.7	2.3	1.9	40.6	8.1	3.6
ML-3.4												
ML-3.5												
ML-3.6										21.0		
ML-3.7												
ML-3.8												
ML-3.9		9.0	7.8	8.0	12.6	22.8		20.8	147.8	28.4	11.3	15.8
ML-3.10												
ML-3.11								33.5		19.4		
ML-3.12												
ML-3.13		38.3	16.8		31.6	89.8			20.8		37.1	11.7
ML-3.14				40.0								
Avg 0-6 m	7.5	4.8	6.3	5.4	5.4	7.4	3.7	2.4	2.5	31.1	8.3	3.5
Avg 7-11 m		9.0	7.8	8.0	12.6	22.8		27.1	147.8	23.9	11.3	15.8
Avg >12 m		38.3	16.8	40.0	31.6	89.8			20.8		37.1	11.7

Dissolved Phosphorus (mg/L)								
Station	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	9/22/2011
ML-1.0	0.023	0.023	0.020	0.013	0.022	0.020		0.021
ML-1.2	0.016	0.017	0.013	0.012		0.024		0.025
ML-1.4	0.017	0.013	0.012	0.013		0.026		0.030
ML-1.6	0.017	0.012	0.018	0.012		0.022		0.033
ML-1.8	0.026	0.017	0.013	0.013		0.037		0.028
ML-2.0	0.017	0.016	0.012	0.012	0.021	0.017		0.024
ML-2.2	0.021	0.017	0.008	0.012		0.021		0.025
ML-2.4	0.026	0.017	0.018	0.013		0.025		0.029
ML-2.6	0.017	0.018	0.016	0.013		0.025		0.033
ML-2.8	0.026	0.019	0.019	0.014		0.026		0.031
ML-2.10	0.019	0.015	0.014	0.015	0.021	0.019		0.033
ML-3.0	0.021	0.012	0.013	0.015	0.022	0.016	0.019	0.021
ML-3.2	0.027	0.023	0.020	0.016		0.023		0.021
ML-3.4	0.019	0.018	0.019	0.018		0.021		0.024
ML-3.6	0.020	0.018	0.017	0.019		0.033	0.027	0.023
ML-3.8	0.027	0.018	0.021	0.024		0.030		0.024
ML-3.10	0.034	0.017	0.017	0.022		0.034		0.023
ML-3.12	0.109	0.061	0.025	0.023		0.021		0.064
ML-3.14	0.120	0.107	0.016	0.017	0.053	0.056	0.050	0.082
ML-3.0 in floc		0.018						
ML-3.10 in floc		0.022						

Total Phosphorus (mg/L)								
Station	9/13/2010	10/5/2010	10/25/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	9/22/2011
ML-1.0	0.027	0.031	0.025	0.022	0.028	0.029		0.024
ML-1.2	0.025	0.020	0.019	0.019		0.033		0.027
ML-1.4	0.031	0.016	0.017	0.018		0.036		0.035
ML-1.6	0.023	0.015	0.023	0.025		0.030		0.035
ML-1.8	0.052	0.019	0.019	0.026		0.055		0.030
ML-2.0	0.029	0.017	0.017	0.022	0.025	0.023		0.030
ML-2.2	0.026	0.020	0.011	0.017		0.030		0.031
ML-2.4	0.036	0.019	0.020	0.020		0.031		0.031
ML-2.6	0.029	0.019	0.019	0.017		0.034		0.036
ML-2.8	0.039	0.021	0.022	0.023		0.035		0.035
ML-2.10	0.070	0.017	0.022	0.019	0.022	0.028		0.042
ML-3.0	0.026	0.013	0.016	0.016	0.023	0.018	0.023	0.027
ML-3.2	0.031	0.027	0.022	0.019		0.028		0.028
ML-3.4	0.029	0.022	0.023	0.026		0.028		0.030
ML-3.6	0.031	0.021	0.023	0.027		0.039	0.036	0.030
ML-3.8	0.030	0.023	0.024	0.027		0.033		0.030
ML-3.10	0.039	0.018	0.023	0.023		0.038		0.030
ML-3.12	0.363	0.103	0.031	0.023		0.028		0.077
ML-3.14	0.768	0.660	0.028	0.017	0.068	0.074	0.089	0.122
ML-3.0 in floc		0.022						
ML-3.10 in floc		0.027						

Total Phosphorus (ug/L)

	8/16/2001	9/5/2002	9/9/2003	2004 Mean	8/24/2004	9/13/2005	8/31/2006	8/21/2007	8/19/2008	8/24/2009	8/18/2010	9/13/2010	9/13/2010	9/13/2010	10/5/2010	10/5/2010	10/5/2010	10/25/2010	10/25/2010	10/25/2010	2/16/2011	2/16/2011	2/16/2011	5/23/2011	5/23/2011	5/23/2011	6/26/2011	6/26/2011	6/26/2011	7/19/2011	8/22/2011	9/22/2011	9/22/2011	9/22/2011
Depth (m)	ML-3	ML-3	ML-3	ML-3	ML-3	ML-3	ML-3	ML-3	ML-3	ML-3	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-3	ML-3	ML-1	ML-2	ML-3
0												27	29	26	31	17	13	25	17	16	22	22	16	28	25	23	29	23	18	23	19	24	30	27
0.5	12	19	16	18	12	13	13	16	7	23	23																							
1																																		
2												25	26	31	20	20	27	19	11	22	19	17	19				33	30	28			27	31	28
3	15		9	15	15	15	19	16	15	34	39																				31			
4												31	36	29	16	19	22	17	20	23	18	20	26				36	31	28			35	31	30
5																																		
6										25		23	29	31	15	19	21	23	19	23	25	17	27				30	34	39	36		35	36	30
7																																		
8												52	70	30	19	21	23	19	22	24	26	23	27				55	35	33			30	35	30
9	15	12	22	46	22	19		96	56	24	31																				25			
10														39		17	18		22	23		19	23		22			28	38				42	30
11								87		41																								
12														363			103			31			23						28					77
13	296	280		492	731				448		901																				96			
14			372			1083								768			660			28			17			68			74	89				122
Avg 0-6 m	13	19	12	16	14	14	16	16	11	27	31	27	30	29	21	19	21	21	17	21	21	19	22	28	25	23	32	30	28	30	25	30	32	29
Avg 7-11 m	15	12	22	46	22	19		91	56	33	31	52	70	35	19	19	21	19	22	24	26	21	25		22		55	32	36		25	30	39	30
Avg >12 m	296	280	372	492	731	1083			448		901			566			382			30			20			68			51	89	96			100

Nitrate Nitrogen (mg/L)

Station	2/16/2011	5/23/2011	6/26/2011	7/19/2011	9/22/2011
ML-1.0	0.21	0.17			0.010
ML-1.2	0.20				0.010
ML-1.4	0.22				0.010
ML-1.6	0.23				0.010
ML-1.8	0.23				0.010
ML-2.0	0.22	0.17			0.010
ML-2.2	0.22				0.010
ML-2.4	0.22				0.010
ML-2.6	0.22				0.010
ML-2.8	0.22				0.010
ML-2.10	0.27	0.15			0.010
ML-3.0	0.24	0.16	0.01	0.010	0.010
ML-3.2	0.24		0.01		0.010
ML-3.4	0.23		0.01		0.010
ML-3.6	0.23		0.01	0.010	0.010
ML-3.8	0.24		0.08		0.010
ML-3.10	0.27		0.07		0.010
ML-3.12	0.30		0.01		0.010
ML-3.14	0.17	0.01	0.01	0.010	0.010

Ammonium Nitrogen (mg/L)

Station	2/16/2011	6/26/2011	7/19/2011	9/22/2011
ML-1.0	0.01			0.01
ML-1.2	0.01			0.01
ML-1.4	0.01			0.01
ML-1.6	0.01			0.01
ML-1.8	0.01			0.01
ML-2.0	0.01			0.01
ML-2.2	0.01			0.01
ML-2.4	0.01			0.01
ML-2.6	0.01			0.01
ML-2.8	0.01			0.01
ML-2.10	0.01			0.01
ML-3.0	0.01	0.20	0.01	0.01
ML-3.2	0.01	0.17		0.01
ML-3.4	0.01	0.15		0.01
ML-3.6	0.01	0.22	0.01	0.01
ML-3.8	0.01	0.28		0.01
ML-3.10	0.01	0.32		0.01
ML-3.12	0.01	0.50		0.01
ML-3.14	0.46	0.80	1.50	0.01

Total Kjeldahl Nitrogen (mg/L)

Station	2/16/2011	5/23/2011	6/26/2011	7/19/2011	9/22/2011
ML-1.0	0.47	0.52			0.50
ML-1.2	0.44				0.47
ML-1.4	0.40				0.48
ML-1.6	0.39				0.44
ML-1.8	0.42				0.52
ML-2.0	0.25	0.48			0.38
ML-2.2	0.27				0.35
ML-2.4	0.27				0.44
ML-2.6	0.24				0.44
ML-2.8	0.24				0.48
ML-2.10	0.22	0.42			0.48
ML-3.0	0.44	0.52	0.48	0.68	0.58
ML-3.2	0.35		0.52		0.54
ML-3.4	0.27		0.45		0.58
ML-3.6	0.38		0.68		0.48
ML-3.8	0.54		0.76		0.54
ML-3.10	0.52		0.86		0.54
ML-3.12	0.52		1.10		1.20
ML-3.14	0.62	1.20	1.30		1.90

Total Nitrogen (mg/L)																
Station	8/16/2001	9/5/2002	9/9/2003	J-S/2004	9/13/2005	8/31/2006	8/21/2007	8/19/2008	8/24/2009	8/18/2010	2/16/2011	5/23/2011	6/26/2011	7/19/2011	8/22/2011	9/22/2011
ML-1.0											0.680	0.690				0.51
ML-1.2											0.640					0.48
ML-1.4											0.620					0.49
ML-1.6											0.620					0.45
ML-1.8											0.650					0.53
ML-2.0											0.470	0.650				0.39
ML-2.2											0.490					0.36
ML-2.4											0.490					0.45
ML-2.6											0.460					0.45
ML-2.8											0.460					0.49
ML-2.10											0.490	0.570				0.49
ML-3.0	0.26	0.30	0.37	0.31	0.26	0.31	0.346	0.259	0.475	0.583	0.680	0.680	0.49	0.69	0.39	0.59
ML-3.1																
ML-3.2											0.590		0.53			0.55
ML-3.3	0.28	0.30	0.26	0.32	0.31	0.52	0.381	0.266	0.832	0.534					0.39	
ML-3.4											0.500		0.46			0.59
ML-3.5																
ML-3.6									0.520		0.610		0.69			0.49
ML-3.7																
ML-3.8											0.780		0.84			0.55
ML-3.9	0.29	0.31	0.26	0.45	0.37		0.868	1.283	0.384	0.751					0.42	
ML-3.10											0.790		0.93			0.55
ML-3.11							0.754		4.971							
ML-3.12											0.820		1.11			1.21
ML-3.13	1.74	1.85		2.02				3.108		4.292					4.27	
ML-3.14			2.29		4.98						0.790	1.210	1.31			1.91
Avg 0-6 m	0.27	0.30	0.32	0.31	0.29	0.41	0.36	0.26	0.61	0.56	0.57	0.67	0.54	0.69	0.39	0.48
Avg 7-11 m	0.29	0.31	0.26	0.45	0.37		0.81	1.28	2.68	0.75	0.63	0.57	0.89		0.42	0.52
Avg >12 m	1.74	1.85	2.29	2.02	4.98			3.11		4.29	0.81	1.21	1.21		4.27	1.56

Appendix C. Plankton Data

Appendix C. Phytoplankton Data for Mystic Lake – cell counts

	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic
	ML-2	ML-2	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3
TAXON	07/22/10	08/16/10	09/13/10	09/13/10	10/04/10	10/04/10	10/04/10	10/04/10	10/25/10	10/25/10	10/25/10
BACILLARIOPHYTA											
Centric Diatoms											
Acanthoceras	0	0	0	0	0	16	15	15	0	18	0
Aulacoseira	0	0	157	211	94	65	0	15	60	55	120
Cyclotella	0	0	0	0	0	0	15	0	0	0	0
Stephanodiscus	0	0	47	16	16	16	0	29	30	18	30
Urosolenia	0	0	0	0	0	16	15	15	270	400	225
Araphid Pennate Diatoms											
Asterionella	0	0	0	0	0	16	15	0	15	36	105
Fragilaria/related taxa	60	0	157	130	63	0	0	0	0	73	0
Synedra	110	63	126	194	126	162	30	44	15	18	15
Tabellaria	0	0	0	0	0	0	15	15	0	0	0
Monoraphid Pennate Diatoms											
Biraphid Pennate Diatoms											
Amphora	0	0	0	0	0	0	0	0	0	18	0
Cymbella/related taxa	10	0	0	0	0	0	0	0	0	0	0
Epithemia	0	0	0	0	0	0	0	0	15	18	0
Eunotia	0	0	0	0	0	0	0	0	0	0	0
Navicula/related taxa	0	0	0	0	0	0	0	0	0	0	0
Nitzschia	0	21	31	65	47	0	0	0	0	0	0
CHLOROPHYTA											
Flagellated Chlorophytes											
Chlamydomonas	0	0	0	0	0	0	0	0	0	0	0
Pandorina	0	0	0	0	0	0	0	0	0	0	0
Coccoid/Colonial Chlorophytes											
Ankistrodesmus	0	0	0	0	0	0	0	0	0	18	15
Chlorococcum	0	0	220	162	204	0	0	0	30	0	15
Closteriopsis	0	0	63	16	16	0	0	0	0	0	0
Coelastrum	0	336	0	0	0	0	0	0	0	0	0
Elakatothrix	0	0	0	0	0	0	0	0	0	0	0
Gloeocystis	3510	0	0	0	0	0	0	0	0	0	0
Golenkinia	0	2016	47	49	79	0	0	15	0	0	0
Kirchneriella	0	84	31	130	31	32	30	44	0	0	0
Oocystis	0	336	126	65	63	0	0	0	60	36	30
Paulschulzia	0	0	0	0	0	0	0	0	0	0	0
Pediastrum	0	0	0	0	0	389	0	0	0	0	0
Scenedesmus	160	1008	879	518	188	518	360	350	840	437	180
Schroederia	10	0	0	0	0	0	0	0	0	0	0
Selenastrum	0	0	0	0	0	0	0	0	0	0	0
Tetraedron	0	945	2512	2576	1931	6755	4935	5402	7500	7280	4575
Filamentous Chlorophytes											
Desmids											
Closterium	0	0	0	0	0	0	0	0	0	0	0
Cosmarium	0	0	0	0	0	0	0	0	0	0	0
Staurastrum	0	0	0	0	0	0	15	15	0	0	0
Teilingia/related taxa	0	0	0	0	0	0	0	0	0	0	0
CHRYSOPHYTA											
Flagellated Classic Chrysophytes											
Dinobryon	0	0	0	0	0	0	0	0	60	73	60
Mallomonas	0	0	16	16	16	16	15	15	15	18	30
Ochromonas	0	294	0	0	0	0	0	0	0	0	0
Non-Motile Classic Chrysophytes											
Haptophytes											
Tribophytes/Eustigmatophytes											
Raphidophytes											
CRYPTOPHYTA											
Cryptomonas	0	63	31	65	31	0	0	0	150	109	45
CYANOPHYTA											
Unicellular and Colonial Forms											
Aphanocapsa	1000	4200	942	648	628	0	0	0	0	0	0
Aphanothece	0	0	0	0	0	0	0	0	0	0	0
Chroococcus	0	0	0	0	0	0	0	0	0	0	0
Dactylococcopsis	0	63	79	49	79	32	0	0	0	0	0
Microcystis	0	3780	3140	2916	2198	4536	3000	2920	1500	1820	1500
Filamentous Nitrogen Fixers											
Anabaena	0	1260	0	0	0	0	0	0	0	0	0
Aphanizomenon	15900	0	0	0	0	0	0	0	0	0	0
Filamentous Non-Nitrogen Fixers											
Planktolyngbya	0	6300	2355	1296	314	6480	3600	4672	0	0	0
Planktothrix	0	840	0	0	0	0	0	0	0	0	0
Pseudanabaena	0	23940	21980	22680	17898	18792	15600	17228	900	910	1050
EUGLENOPHYTA											
Trachelomonas	10	42	47	81	79	162	120	117	30	36	30
PYRRHOPHYTA											
Ceratium	30	0	16	16	16	16	15	15	0	0	0
Peridinium	30	42	31	32	31	32	15	15	15	18	15

	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic
	ML-1	ML-2	ML-3	ML-3	ML-1	ML-2	ML-3	ML-3.8.5	Dock	ML-3.4	ML-3.0
TAXON	02/16/11	02/16/11	02/16/11	05/23/11	06/26/11	06/26/11	06/26/11	06/26/11	06/26/11	07/07/11	07/19/11
BACILLARIOPHYTA											
Centric Diatoms											
Acanthoceras	0	0	0	0	0	0	0	0	0	0	0
Aulacoseira	0	0	0	0	0	0	0	0	0	0	0
Cyclotella	0	0	0	0	0	0	0	0	0	0	0
Stephanodiscus	14	13	0	0	0	0	0	0	0	0	0
Urosolenia	419	370	313	0	0	0	44	0	150	0	0
Araphid Pennate Diatoms											
Asterionella	8667	5148	5475	391	360	287	308	1804	900	60	28
Fragilaria/related taxa	959	1056	1125	340	900	820	1232	3280	600	2490	2016
Synedra	0	0	13	0	648	738	748	861	1350	2040	84
Tabellaria	0	0	0	0	0	0	0	0	0	60	0
Monoraphid Pennate Diatoms											
Biraphid Pennate Diatoms											
Amphora	0	0	0	0	0	0	0	0	0	0	0
Cymbella/related taxa	0	0	0	0	0	0	0	0	0	0	0
Epithemia	0	0	0	0	0	0	0	0	0	0	0
Eunotia	0	0	0	0	0	0	0	0	0	0	0
Navicula/related taxa	0	0	0	0	0	0	0	0	0	0	28
Nitzschia	0	0	0	17	0	0	0	41	0	0	0
CHLOROPHYTA											
Flagellated Chlorophytes											
Chlamydomonas	0	0	0	0	0	0	0	0	0	0	0
Pandorina	0	0	0	0	0	0	0	0	4800	0	0
Coccoid/Colonial Chlorophytes											
Ankistrodesmus	0	0	13	0	0	0	0	0	0	0	0
Chlorococcum	0	0	0	0	0	0	0	0	0	0	0
Closteriopsis	0	0	0	34	0	0	0	0	0	0	0
Coelastrum	0	0	0	136	0	0	0	0	0	0	0
Elakatothrix	81	53	25	68	0	0	0	0	0	0	0
Gloeocystis	0	0	0	0	0	0	0	0	0	0	0
Golenkinia	0	0	0	0	36	41	44	0	600	90	56
Kirchneriella	0	0	0	0	0	0	0	0	0	0	0
Oocystis	54	0	0	0	0	0	0	0	0	0	0
Paulschulzia	0	0	0	0	72	0	0	0	0	0	0
Pediastrum	0	0	0	0	0	0	0	0	0	0	0
Scenedesmus	54	53	0	0	720	984	1232	820	300	480	1792
Schroederia	0	0	0	0	0	0	0	0	0	0	0
Selenastrum	0	0	0	0	36	82	660	656	300	60	0
Tetraedron	257	317	125	17	1404	1230	704	697	450	1350	2800
Filamentous Chlorophytes											
Desmids											
Closterium	0	0	0	0	0	0	0	0	0	0	0
Cosmarium	14	0	0	0	0	41	0	0	0	30	0
Staurastrum	0	13	0	0	36	41	0	41	150	30	224
Teilingia/related taxa	0	26	0	0	72	0	0	0	0	0	0
CHRYSOPHYTA											
Flagellated Classic Chrysophytes											
Dinobryon	54	26	63	153	0	0	88	82	0	0	0
Mallomonas	95	66	38	68	36	41	0	0	150	0	28
Ochromonas	0	0	0	0	0	0	0	0	0	0	0
Non-Motile Classic Chrysophytes											
Haptophytes											
Tribophytes/Eustigmatophytes											
Raphidophytes											
CRYPTOPHYTA											
Cryptomonas	0	0	0	0	144	82	88	82	600	0	0
CYANOPHYTA											
Unicellular and Colonial Forms											
Aphanocapsa	0	0	0	0	0	0	0	0	0	0	0
Aphanothece	0	0	0	0	2160	0	0	0	0	0	0
Chroococcus	0	0	0	0	0	0	0	0	0	0	0
Dactylococcopsis	0	0	0	0	0	0	0	0	0	0	0
Microcystis	0	0	0	0	2160	3280	3080	1640	127500	1800	2240
Filamentous Nitrogen Fixers											
Anabaena	0	0	0	0	0	0	0	0	0	0	0
Aphanizomenon	0	0	0	0	0	0	0	0	0	0	420
Filamentous Non-Nitrogen Fixers											
Planktolyngbya	0	0	0	0	0	0	0	0	0	0	0
Planktothrix	0	0	0	0	0	0	0	0	0	0	2520
Pseudanabaena	0	0	0	0	0	0	0	0	0	0	0
EUGLENOPHYTA											
Trachelomonas	27	13	13	0	0	0	0	82	0	30	0
PYRRHOPHYTA											
Ceratium	0	0	0	9	0	41	0	0	0	30	0
Peridinium	0	0	0	0	72	287	88	41	150	30	0

	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic
	ML-3.0	ML-3.6	ML-1.0	ML-3.0	ML-3.0	ML-1.0	ML-2.0	ML-3.0	ML-2.0
TAXON	07/19/11	07/19/11	08/06/11	08/06/11	08/30/11	09/22/11	09/22/11	09/22/11	10/14/11
BACILLARIOPHYTA									
Centric Diatoms									
Acanthoceras	0	0	0	0	0	0	0	0	0
Aulacoseira	0	0	0	0	0	12	15	12	0
Cyclotella	0	0	0	0	30	24	90	85	0
Stephanodiscus	0	0	0	0	0	0	0	12	0
Urosolenia	0	0	0	0	1110	120	90	85	192
Araphid Pennate Diatoms									
Asterionella	28	32	0	0	15	0	0	0	0
Fragilaria/related taxa	2016	1072	0	0	300	0	0	122	0
Synedra	84	48	270	192	0	24	15	12	756
Tabellaria	0	32	0	0	0	0	0	0	0
Monoraphid Pennate Diatoms									
Biraphid Pennate Diatoms									
Amphora	0	0	0	0	0	0	0	0	0
Cymbella/related taxa	0	0	0	0	0	0	0	0	0
Epithemia	0	0	0	0	0	0	0	0	0
Eunotia	0	0	23	0	0	0	0	0	0
Navicula/related taxa	28	0	0	0	0	0	0	0	12
Nitzschia	0	0	0	0	0	0	15	12	0
CHLOROPHYTA									
Flagellated Chlorophytes									
Chlamydomonas	0	0	90	24	0	0	0	0	0
Pandorina	0	0	0	0	120	96	0	0	0
Coccoid/Colonial Chlorophytes									
Ankistrodesmus	0	0	0	0	30	0	15	12	0
Chlorococcum	0	0	0	0	0	0	0	0	0
Closteriopsis	0	0	0	0	0	0	0	0	0
Coelastrum	0	0	0	0	240	96	0	0	0
Elakatothrix	0	0	0	0	60	24	30	24	0
Gloeocystis	0	0	0	0	0	0	0	0	0
Golenkinia	56	32	0	0	0	0	15	0	0
Kirchneriella	0	0	0	0	0	0	0	0	0
Oocystis	0	0	0	0	0	0	0	0	0
Paulschulzia	0	0	0	0	0	0	180	49	0
Pediastrum	0	0	0	0	0	0	0	0	0
Scenedesmus	1792	1088	180	96	0	120	90	73	96
Schroederia	0	0	0	0	0	0	0	0	0
Selenastrum	0	0	0	0	0	0	0	0	0
Tetraedron	2800	3456	1958	1992	990	5328	5100	5441	2412
Filamentous Chlorophytes									
Desmids									
Closterium	0	0	23	0	0	12	0	0	0
Cosmarium	0	0	0	0	0	0	0	0	0
Staurastrum	224	208	113	168	90	36	15	12	0
Teilingia/related taxa	0	0	0	0	0	0	0	0	0
CHRYSOPHYTA									
Flagellated Classic Chrysophytes									
Dinobryon	0	16	0	0	0	12	15	12	300
Mallomonas	28	0	68	24	30	120	120	98	24
Ochromonas	0	0	0	0	0	0	0	0	0
Non-Motile Classic Chrysophytes									
Haptophytes									
Tribophytes/Eustigmatophytes									
Raphidophytes									
CRYPTOPHYTA									
Cryptomonas	0	64	180	72	0	0	0	0	0
CYANOPHYTA									
Unicellular and Colonial Forms									
Aphanocapsa	0	0	1800	0	0	0	0	0	0
Aphanothece	0	1280	0	0	0	0	0	0	0
Chroococcus	0	0	0	720	0	0	0	0	0
Dactylococcopsis	0	0	0	0	0	0	0	0	0
Microcystis	2240	1920	0	0	0	240	300	244	0
Filamentous Nitrogen Fixers									
Anabaena	0	0	0	0	0	0	0	0	0
Aphanizomenon	420	480	11250	24480	9600	0	0	0	0
Filamentous Non-Nitrogen Fixers									
Planktolyngbya	0	0	0	0	0	0	0	0	0
Planktothrix	2520	1600	0	0	0	0	0	0	0
Pseudanabaena	0	0	3150	3600	6750	0	0	0	0
EUGLENOPHYTA									
Trachelomonas	0	32	23	24	15	12	15	12	24
PYRRHOPHYTA									
Ceratium	0	0	0	0	0	12	0	0	0
Peridinium	0	32	0	24	45	12	15	12	24

Appendix C. Phytoplankton Data for Mystic Lake – biomass estimates

	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic
	ML-2	ML-2	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3
TAXON	07/22/10	08/16/10	09/13/10	09/13/10	10/04/10	10/04/10	10/04/10	10/04/10	10/25/10	10/25/10	10/25/10
BACILLARIOPHYTA											
Centric Diatoms											
Acanthoceras	0.0	0.0	0.0	0.0	0.0	19.4	18.0	17.5	0.0	21.8	0.0
Aulacoseira	0.0	0.0	47.1	63.2	28.3	19.4	0.0	4.4	18.0	16.4	36.0
Cyclotella	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0
Stephanodiscus	0.0	0.0	117.8	40.5	39.3	40.5	0.0	153.3	39.0	45.5	157.5
Urosolenia	0.0	0.0	0.0	0.0	0.0	19.4	18.0	17.5	324.0	480.5	270.0
Araphid Pennate Diatoms											
Asterionella	0.0	0.0	0.0	0.0	0.0	3.2	3.0	0.0	3.0	7.3	21.0
Fragilaria/related taxa	18.0	0.0	47.1	38.9	18.8	0.0	0.0	0.0	0.0	21.8	0.0
Synedra	160.0	50.4	100.5	155.5	100.5	596.2	24.0	35.0	12.0	14.6	12.0
Tabellaria	0.0	0.0	0.0	0.0	0.0	0.0	12.0	11.7	0.0	0.0	0.0
Monoraphid Pennate Diatoms											
Biraphid Pennate Diatoms											
Amphora	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	25.5	0.0
Cymbella/related taxa	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	72.0	87.4	0.0
Eunotia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula/related taxa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nitzschia	0.0	16.8	25.1	51.8	37.7	0.0	0.0	0.0	0.0	0.0	0.0
CHLOROPHYTA											
Flagellated Chlorophytes											
Chlamydomonas	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pandorina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coccoid/Colonial Chlorophytes											
Ankistrodesmus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	1.5
Chlorococcum	0.0	0.0	22.0	16.2	20.4	0.0	0.0	0.0	3.0	0.0	1.5
Closteriopsis	0.0	0.0	31.4	8.1	7.9	0.0	0.0	0.0	0.0	0.0	0.0
Coelastrum	0.0	67.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Elakatothrix	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gloeocystis	702.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Golenkinia	0.0	403.2	9.4	9.7	15.7	0.0	0.0	2.9	0.0	0.0	0.0
Kirchneriella	0.0	8.4	3.1	13.0	3.1	3.2	3.0	4.4	0.0	0.0	0.0
Oocystis	0.0	134.4	50.2	25.9	25.1	0.0	0.0	0.0	24.0	14.6	12.0
Paulschulzia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pediastrum	0.0	0.0	0.0	0.0	0.0	77.8	0.0	0.0	0.0	0.0	0.0
Scenedesmus	16.0	100.8	87.9	51.8	18.8	51.8	36.0	35.0	84.0	43.7	18.0
Schroederia	25.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Selenastrum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tetraedron	0.0	567.0	1507.2	1545.5	1158.7	4053.2	2961.0	3241.2	4500.0	4368.0	2745.0
Filamentous Chlorophytes											
Desmids											
Closterium	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cosmarium	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum	0.0	0.0	0.0	0.0	0.0	0.0	12.0	11.7	0.0	0.0	0.0
Teilingia/related taxa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CHRYSOPHYTA											
Flagellated Classic Chrysophytes											
Dinobryon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	180.0	218.4	180.0
Mallomonas	0.0	0.0	7.9	8.1	7.9	8.1	7.5	7.3	7.5	9.1	67.5
Ochromonas	0.0	29.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Non-Motile Classic Chrysophytes											
Haptophytes											
Tribophytes/Eustigmatophytes											
Raphidophytes											
CRYPTOPHYTA											
Cryptomonas	0.0	12.6	6.3	13.0	6.3	0.0	0.0	0.0	30.0	21.8	9.0
CYANOPHYTA											
Unicellular and Colonial Forms											
Aphanocapsa	10.0	42.0	9.4	6.5	6.3	0.0	0.0	0.0	0.0	0.0	0.0
Aphanothece	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chroococcus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dactylococcopsis	0.0	25.2	31.4	0.5	0.8	0.3	0.0	0.0	0.0	0.0	0.0
Microcystis	0.0	37.8	31.4	29.2	22.0	45.4	30.0	29.2	15.0	18.2	15.0
Filamentous Nitrogen Fixers											
Anabaena	0.0	252.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aphanizomenon	2067.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Filamentous Non-Nitrogen Fixers											
Planktolyngbya	0.0	63.0	23.6	13.0	3.1	64.8	36.0	46.7	0.0	0.0	0.0
Planktothrix	0.0	8.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudanabaena	0.0	239.4	219.8	226.8	179.0	187.9	156.0	172.3	9.0	9.1	10.5
EUGLENOPHYTA											
Trachelomonas	10.0	42.0	47.1	81.0	78.5	440.6	442.5	367.9	30.0	36.4	30.0
PYRRHOPHYTA											
Ceratium	522.0	0.0	273.2	281.9	273.2	281.9	261.0	254.0	0.0	0.0	0.0
Peridinium	63.0	88.2	65.9	68.0	65.9	68.0	31.5	30.7	31.5	38.2	31.5

	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic
	ML-1	ML-2	ML-3	ML-3	ML-1	ML-2	ML-3	ML-3.8.5	Dock
TAXON	02/16/11	02/16/11	02/16/11	05/23/11	06/26/11	06/26/11	06/26/11	06/26/11	06/26/11
BACILLARIOPHYTA									
Centric Diatoms									
Acanthoceras	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aulacoseira	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cyclotella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stephanodiscus	33.8	33.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Urosolenia	502.2	443.5	375.0	0.0	0.0	0.0	52.8	0.0	180.0
Araphid Pennate Diatoms									
Asterionella	1733.4	1029.6	1095.0	78.2	72.0	57.4	61.6	360.8	180.0
Fragilaria/related taxa	287.6	316.8	337.5	102.0	270.0	246.0	369.6	984.0	180.0
Synedra	0.0	0.0	10.0	0.0	518.4	590.4	598.4	688.8	1080.0
Tabellaria	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Monoraphid Pennate Diatoms									
Biraphid Pennate Diatoms									
Amphora	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cymbella/related taxa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eunotia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula/related taxa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nitzschia	0.0	0.0	0.0	13.6	0.0	0.0	0.0	32.8	0.0
CHLOROPHYTA									
Flagellated Chlorophytes									
Chlamydomonas	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pandorina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	480.0
Coccoid/Colonial Chlorophytes									
Ankistrodesmus	0.0	0.0	6.3	0.0	0.0	0.0	0.0	0.0	0.0
Chlorococcum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Closteriopsis	0.0	0.0	0.0	17.0	0.0	0.0	0.0	0.0	0.0
Coelastrum	0.0	0.0	0.0	27.2	0.0	0.0	0.0	0.0	0.0
Elakatothrix	8.1	5.3	2.5	6.8	0.0	0.0	0.0	0.0	0.0
Gloeocystis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Golenkinia	0.0	0.0	0.0	0.0	7.2	8.2	8.8	0.0	120.0
Kirchneriella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oocystis	21.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paulschulzia	0.0	0.0	0.0	0.0	28.8	0.0	0.0	0.0	0.0
Pediastrum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scenedesmus	5.4	5.3	0.0	0.0	72.0	98.4	123.2	82.0	30.0
Schroederia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Selenastrum	0.0	0.0	0.0	0.0	3.6	8.2	66.0	65.6	30.0
Tetraedron	153.9	190.1	75.0	10.2	842.4	738.0	422.4	418.2	270.0
Filamentous Chlorophytes									
Desmids									
Closterium	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cosmarium	10.8	0.0	0.0	0.0	0.0	32.8	0.0	0.0	0.0
Staurastrum	0.0	10.6	0.0	0.0	28.8	32.8	0.0	32.8	120.0
Teilingia/related taxa	0.0	52.8	0.0	0.0	144.0	0.0	0.0	0.0	0.0
CHRYSOPHYTA									
Flagellated Classic Chrysophytes									
Dinobryon	162.0	79.2	187.5	459.0	0.0	0.0	264.0	246.0	0.0
Mallomonas	47.3	33.0	18.8	34.0	18.0	20.5	0.0	0.0	75.0
Ochromonas	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Non-Motile Classic Chrysophytes									
Haptophytes									
Tribophytes/Eustigmatophytes									
Raphidophytes									
CRYPTOPHYTA									
Cryptomonas	0.0	0.0	0.0	0.0	28.8	16.4	17.6	16.4	120.0
CYANOPHYTA									
Unicellular and Colonial Forms									
Aphanocapsa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aphanothece	0.0	0.0	0.0	0.0	21.6	0.0	0.0	0.0	0.0
Chroococcus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dactylococcopsis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Microcystis	0.0	0.0	0.0	0.0	21.6	32.8	30.8	16.4	1275.0
Filamentous Nitrogen Fixers									
Anabaena	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aphanizomenon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Filamentous Non-Nitrogen Fixers									
Planktolyngbya	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planktothrix	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudanabaena	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
EUGLENOPHYTA									
Trachelomonas	27.0	13.2	12.5	0.0	0.0	0.0	0.0	82.0	0.0
PYRRHOPHYTA									
Ceratium	0.0	0.0	0.0	147.9	0.0	713.4	0.0	0.0	0.0
Peridinium	0.0	0.0	0.0	0.0	151.2	602.7	184.8	86.1	315.0

	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic
	ML-3.4	ML-3.0	ML-3.6	ML-1.0	ML-3.0	ML-3.0	ML-1.0	ML-2.0	ML-3.0	ML-2.0
TAXON	07/07/11	07/19/11	07/19/11	08/06/11	08/06/11	08/30/11	09/22/11	09/22/11	09/22/11	10/14/11
BACILLARIOPHYTA										
Centric Diatoms										
Acanthoceras	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aulacoseira	0.0	0.0	0.0	0.0	0.0	0.0	3.6	4.5	3.7	0.0
Cyclotella	0.0	0.0	0.0	0.0	0.0	3.0	2.4	45.0	37.8	0.0
Stephanodiscus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	30.5	0.0
Urosolenia	0.0	0.0	0.0	0.0	0.0	1332.0	144.0	108.0	102.5	230.4
Araphid Pennate Diatoms										
Asterionella	12.0	5.6	6.4	0.0	0.0	3.0	0.0	0.0	0.0	0.0
Fragilaria/related taxa	747.0	604.8	321.6	0.0	0.0	90.0	0.0	0.0	36.6	0.0
Synedra	1632.0	67.2	38.4	216.0	153.6	0.0	19.2	12.0	9.8	604.8
Tabellaria	48.0	0.0	25.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Monoraphid Pennate Diatoms										
Biraphid Pennate Diatoms										
Amphora	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cymbella/related taxa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epithemia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eunotia	0.0	0.0	0.0	22.5	0.0	0.0	0.0	0.0	0.0	0.0
Navicula/related taxa	0.0	14.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.0
Nitzschia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.0	9.8	0.0
CHLOROPHYTA										
Flagellated Chlorophytes										
Chlamydomonas	0.0	0.0	0.0	36.0	9.6	0.0	0.0	0.0	0.0	0.0
Pandorina	0.0	0.0	0.0	0.0	0.0	12.0	9.6	0.0	0.0	0.0
Coccoid/Colonial Chlorophytes										
Ankistrodesmus	0.0	0.0	0.0	0.0	0.0	3.0	0.0	1.5	1.2	0.0
Chlorococcum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Closteriopsis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coelastrum	0.0	0.0	0.0	0.0	0.0	48.0	19.2	0.0	0.0	0.0
Elakatothrix	0.0	0.0	0.0	0.0	0.0	6.0	2.4	3.0	2.4	0.0
Gloeocystis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Golenkinia	48.0	11.2	6.4	0.0	0.0	0.0	0.0	3.0	0.0	0.0
Kirchneriella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oocystis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paulschulzia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	72.0	19.5	0.0
Pediastrum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scenedesmus	48.0	179.2	108.8	18.0	9.6	0.0	12.0	9.0	7.3	9.6
Schroederia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Selenastrum	6.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tetraedron	810.0	1680.0	2073.6	1174.5	1195.2	594.0	3196.8	3060.0	3264.7	1447.2
Filamentous Chlorophytes										
Desmids										
Closterium	0.0	0.0	0.0	90.0	0.0	0.0	48.0	0.0	0.0	0.0
Cosmarium	24.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum	24.0	179.2	166.4	90.0	134.4	72.0	28.8	12.0	9.8	0.0
Teilingia/related taxa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CHRYSOPHYTA										
Flagellated Classic Chrysophytes										
Dinobryon	0.0	0.0	48.0	0.0	0.0	0.0	36.0	45.0	36.6	900.0
Mallomonas	0.0	14.0	0.0	33.8	12.0	15.0	60.0	60.0	48.8	12.0
Ochromonas	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Non-Motile Classic Chrysophytes										
Haptophytes										
Tribophytes/Eustigmatophytes										
Raphidophytes										
CRYPTOPHYTA										
Cryptomonas	0.0	0.0	35.2	36.0	14.4	0.0	0.0	0.0	0.0	0.0
CYANOPHYTA										
Unicellular and Colonial Forms										
Aphanocapsa	0.0	0.0	0.0	18.0	0.0	0.0	0.0	0.0	0.0	0.0
Aphanothece	0.0	0.0	12.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chroococcus	0.0	0.0	0.0	0.0	288.0	0.0	0.0	0.0	0.0	0.0
Dactylococcopsis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Microcystis	54.0	22.4	19.2	0.0	0.0	0.0	2.4	3.0	2.4	0.0
Filamentous Nitrogen Fixers										
Anabaena	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aphanizomenon	0.0	54.6	62.4	1462.5	3182.4	1248.0	0.0	0.0	0.0	0.0
Filamentous Non-Nitrogen Fixers										
Planktolyngbya	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Planktothrix	0.0	25.2	16.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudanabaena	0.0	0.0	0.0	31.5	36.0	67.5	0.0	0.0	0.0	0.0
EUGLENOPHYTA										
Trachelomonas	30.0	0.0	32.0	119.3	24.0	79.5	12.0	15.0	12.2	24.0
PYRRHOPHYTA										
Ceratium	522.0	0.0	0.0	0.0	0.0	0.0	208.8	0.0	0.0	0.0
Peridinium	63.0	0.0	67.2	0.0	50.4	94.5	25.2	31.5	25.6	50.4

Appendix C – Zooplankton Data for Mystic Lake – #/L

	ZOOPLANKTON DENSITY (#/L)																
	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic
	ML-2	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-3	ML-3	ML-3	ML-3
TAXON	8/16/10	9/13/10	9/13/10	9/13/10	10/4/10	10/4/10	10/4/10	10/25/10	10/25/10	10/25/10	2/16/11	2/16/11	2/16/11	5/29/11	6/26/11	7/19/11	9/22/11
PROTOZOA																	
Ciliophora	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mastigophora	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sarcodina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ROTIFERA																	
Asplanchna	0.0	0.0	0.0	0.0	4.7	2.8	7.2	0.8	1.2	1.1	3.0	4.6	4.4	0.0	0.0	5.5	0.0
Conochilus	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	1.1
Hexarthra	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Kellicottia	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.6	0.0	0.0	1.0	0.1
Keratella	0.4	0.0	0.0	0.0	0.0	0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.1	0.3
Polyarthra	0.4	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.2
Trichocerca	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
COPEPODA																	
Copepoda-Cyclopoida																	
Cyclops	0.0	0.0	0.1	0.1	0.0	0.0	0.5	0.3	0.4	0.2	6.2	5.3	6.3	0.0	0.1	0.0	0.0
Mesocyclops	0.0	0.1	0.1	0.4	0.1	0.6	1.1	0.4	0.1	0.4	0.5	1.1	1.0	0.3	0.6	0.0	0.5
Copepoda-Calanoida																	
Diaptomus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0
Other Copepoda-Nauplii	0.0	0.1	0.1	0.1	0.1	0.1	0.2	0.1	0.1	0.0	1.6	1.9	1.5	0.3	0.1	0.1	0.5
CLADOCERA																	
Bosmina	0.4	0.5	0.7	1.6	14.6	17.3	43.0	28.8	13.2	86.8	10.9	10.1	12.2	0.0	0.0	1.3	6.1
Ceriodaphnia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Chydorus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.3	0.2	0.0	0.0	0.0	0.0
Daphnia ambigua	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	5.4	2.4	2.9	0.0	0.1	0.1	0.0
Daphnia pulex	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.1	0.0	0.0	0.0
Diaphanosoma	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Holopedium	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
OTHER ZOOPLANKTON																	
Hydracarina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
SUMMARY STATISTICS																	
DENSITY																	
PROTOZOA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ROTIFERA	0.8	0.0	0.0	0.1	4.9	2.9	7.4	0.8	1.2	1.1	3.4	5.0	4.9	1.4	0.1	6.8	1.8
COPEPODA	0.0	0.2	0.3	0.6	0.2	0.7	1.8	0.8	0.7	0.5	8.3	8.3	8.7	0.7	0.8	0.5	1.0
CLADOCERA	0.4	0.5	0.7	1.6	14.6	17.3	43.0	28.8	13.2	86.9	16.8	12.8	15.2	5.1	0.1	1.5	6.2
OTHER ZOOPLANKTON	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
TOTAL ZOOPLANKTON	1.2	0.7	1.0	2.2	19.8	20.9	52.2	30.5	15.1	88.6	28.5	26.1	28.9	7.1	1.0	8.9	9.0
TAXONOMIC RICHNESS																	
PROTOZOA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
ROTIFERA	4	0	0	2	3	2	2	1	1	1	2	2	2	2	1	4	4
COPEPODA	0	2	3	3	2	2	3	3	4	2	3	3	3	2	3	2	2
CLADOCERA	1	1	1	1	1	1	1	1	1	2	3	3	3	1	1	3	3
OTHER ZOOPLANKTON	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	1	0
TOTAL ZOOPLANKTON	5	3	4	6	6	5	7	6	6	5	8	8	8	5	5	10	9
S-W DIVERSITY INDEX	0.61	0.35	0.37	0.43	0.30	0.25	0.26	0.12	0.22	0.05	0.70	0.72	0.69	0.42	0.49	0.56	0.51
EVENNESS INDEX	0.88	0.72	0.62	0.55	0.38	0.36	0.31	0.16	0.29	0.08	0.77	0.79	0.76	0.61	0.70	0.56	0.54
MEAN LENGTH (mm): ALL FORMS	0.16	0.33	0.34	0.34	0.32	0.32	0.32	0.31	0.32	0.30	0.56	0.50	0.51	0.95	0.90	0.32	0.28
MEAN LENGTH: CRUSTACEANS	0.30	0.33	0.34	0.35	0.30	0.31	0.31	0.31	0.32	0.30	0.58	0.54	0.55	1.15	0.95	0.42	0.32

Appendix C – Zooplankton Data for Mystic Lake – Biomass Estimates

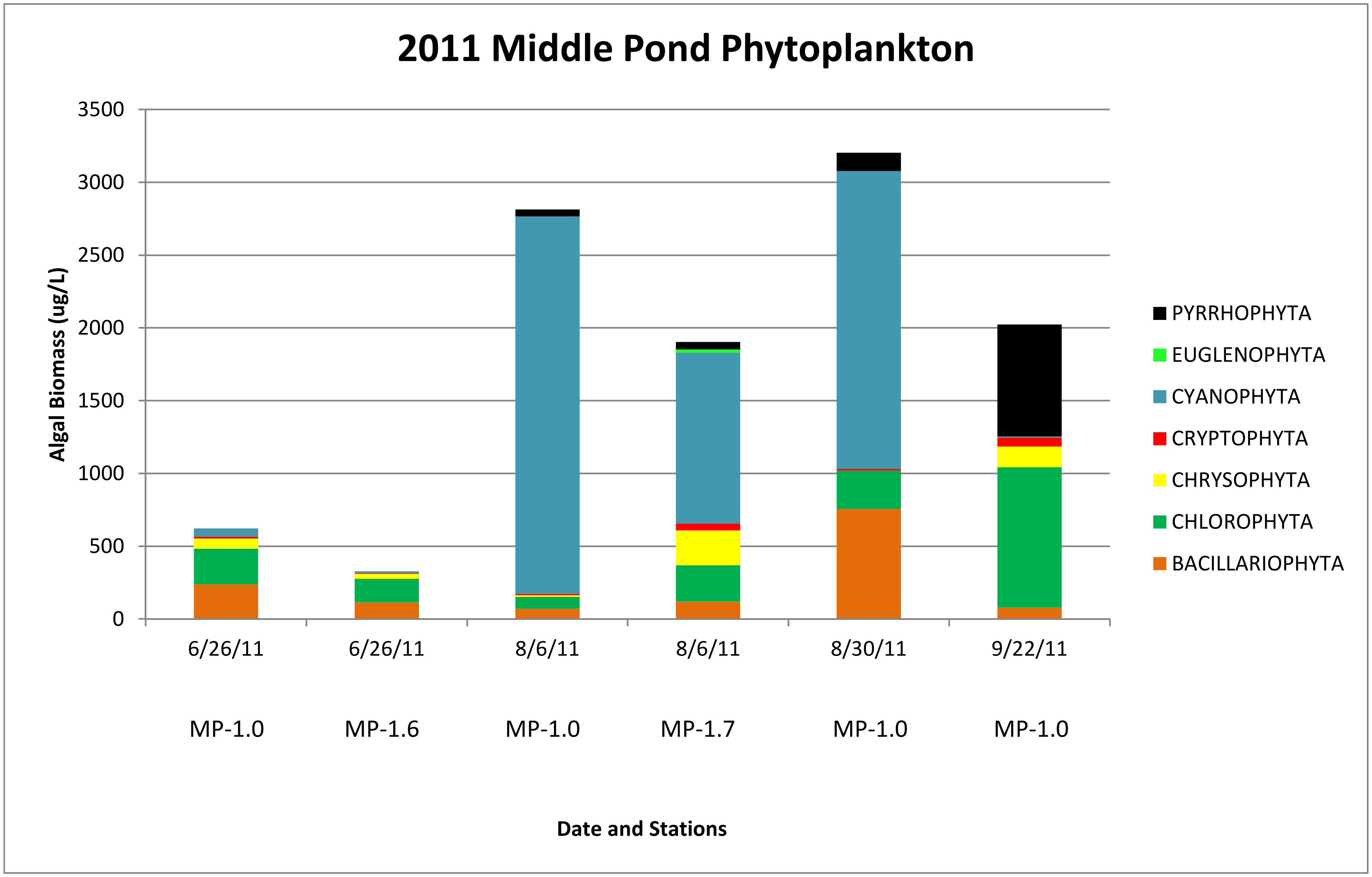
	ZOOPLANKTON BIOMASS (UG/L)																
	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic	Mystic
	ML-2	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-1	ML-2	ML-3	ML-3	ML-3	ML-3	ML-3
TAXON	8/16/10	9/13/10	9/13/10	9/13/10	10/4/10	10/4/10	10/4/10	10/25/10	10/25/10	10/25/10	2/16/11	2/16/11	2/16/11	5/29/11	6/26/11	7/19/11	9/22/11
PROTOZOA																	
Ciliophora	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mastigophora	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sarcodina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ROTIFERA																	
Asplanchna	0.0	0.0	0.0	0.0	7.0	4.3	10.4	1.3	1.8	1.6	4.5	6.2	6.3	0.0	0.0	5.5	0.0
Conochilus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hexarthra	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Kellicottia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Keratella	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
Polyarthra	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trichocerca	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
COPEPODA																	
Copepoda-Cyclopoida																	
Cyclops	0.0	0.0	0.1	0.2	0.0	0.0	1.2	0.7	1.0	0.4	23.7	18.7	22.2	0.0	0.3	0.0	0.0
Mesocyclops	0.0	0.1	0.1	0.4	0.2	0.8	1.4	0.5	0.1	0.5	0.6	1.4	1.2	1.1	4.6	0.0	0.6
Copepoda-Calanoida																	
Diaptomus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
Other Copepoda-Nauplii	0.0	0.3	0.3	0.3	0.3	0.3	0.4	0.4	0.3	0.0	4.2	5.1	4.0	0.9	0.2	0.3	1.3
CLADOCERA																	
Bosmina	0.3	0.5	0.7	1.5	14.3	16.9	42.2	28.3	12.9	85.0	21.5	19.5	24.1	0.0	0.0	1.3	6.0
Ceriodaphnia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2
Chydorus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.3	0.2	0.0	0.0	0.0	0.0
Daphnia ambigua	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	20.2	8.6	10.2	0.0	0.1	0.2	0.0
Daphnia pulex	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	56.6	0.0	0.0	0.0
Diaphanosoma	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Holopedium	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0
OTHER ZOOPLANKTON																	
Hydracarina	0.0	0.0	0.0	0.0	0.0	0.0	15.2	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	95.0	0.0
SUMMARY STATISTICS																	
BIOMASS																	
PROTOZOA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ROTIFERA	0.1	0.0	0.0	0.0	7.0	4.3	10.4	1.3	1.8	1.6	4.5	6.3	6.3	0.1	0.0	5.6	0.1
COPEPODA	0.0	0.4	0.5	0.9	0.5	1.1	3.0	1.6	1.4	0.9	28.6	25.2	27.4	2.0	5.2	0.5	1.9
CLADOCERA	0.3	0.5	0.7	1.5	14.3	16.9	42.2	28.3	12.9	85.3	42.2	28.4	34.4	56.6	0.1	2.3	6.2
OTHER ZOOPLANKTON	0.0	0.0	0.0	0.0	0.0	0.0	15.2	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	95.0	0.0
TOTAL ZOOPLANKTON	0.4	0.9	1.2	2.5	21.8	22.3	70.8	44.4	16.2	87.8	75.2	59.8	68.1	58.7	5.3	103.3	8.2

Appendix D. Middle Pond Data from WRS

	Date	Time	Depth	Temp	LDO	LDO%	SpCond	Turbidity	pH	Alkalinity	Secchi
	MM.DD.YY	HH:MM:SS	m	°C	mg/l	Sat	µS/cm	NTU	Units	mg/l	m
Middle-1	06.26.11	11:27:25	0.1	22.6	9.3	108	103	1.2	7.4	11.0	4.4
	06.26.11	11:27:54	2.0	22.0	9.8	113	103	1.8	7.4	11.0	
	06.26.11	11:28:18	4.0	21.8	9.8	112	102	1.8	7.4	11.0	
	06.26.11	11:28:54	6.0	19.5	10.0	109	102	1.8	7.3	11.0	
	06.26.11	11:29:55	8.0	15.3	3.0	30	103	1.4	7.0	11.0	
	06.26.11	11:30:34	10.0	14.1	0.6	6	113	2.2	6.5	15.0	
Middle-1	09.22.11	10:29:16	0.1	21.7	8.4	96.6	86	2.5	7.5	12.0	3.3
	09.22.11	10:30:01	2.0	21.2	8.2	92.2	86.0	2.5	7.4		
	09.22.11	10:30:38	4.0	20.9	7.9	89.7	86	2.5	7.4		
	09.22.11	10:31:13	6.0	20.8	7.5	84.6	86	2.6	7.3		
	09.22.11	10:32:02	8.0	20.8	7.0	78.8	86	2.5	7.3		
	09.22.11	10:32:56	10.0	19.8	0.3	2.8	91	7.3	6.8	17.0	

	Date	Depth	Ammonium N	Nitrate N	TKN	Dissolved P	Total P
	MM.DD.YY	m	mg/l	mg/l	mg/l	mg/l	mg/l
Middle-1	06.26.11	0.1	0.280	0.010	0.500	0.015	0.019
	06.26.11	2.0	0.260	0.010	0.450	0.025	0.030
	06.26.11	4.0	0.210	0.010	0.350	0.021	0.026
	06.26.11	6.0	0.190	0.010	0.330	0.028	0.032
	06.26.11	8.0	0.280	0.010	0.480	0.036	0.037
	06.26.11	10.0	0.320	0.010	0.580	0.042	0.045
Middle-1	09.22.11	0.1	0.070	0.010	0.540	0.023	0.030
	09.22.11	2.0	0.020	0.010	0.860	0.030	0.034
	09.22.11	4.0	0.020	0.010	0.560	0.022	0.027
	09.22.11	6.0	0.020	0.010	0.600	0.028	0.032
	09.22.11	8.0	0.020	0.010	0.480	0.024	0.028
	09.22.11	10.0	0.020	0.010	0.680	0.029	0.036

	PHYTOPLANKTON DENSITY (CELLS/ML)							PHYTOPLANKTON BIOMASS (UG/L)					
	Middle	Middle	Middle	Middle	Middle	Middle		Middle	Middle	Middle	Middle	Middle	Middle
	MP-1.0	MP-1.6	MP-1.0	MP-1.7	MP-1.0	MP-1.0		MP-1.0	MP-1.6	MP-1.0	MP-1.7	MP-1.0	MP-1.0
TAXON	06/26/11	06/26/11	08/06/11	08/06/11	08/30/11	09/22/11	**TAXON**	06/26/11	06/26/11	08/06/11	08/06/11	08/30/11	09/22/11
BACILLARIOPHYTA							**BACILLARIOPHYTA**						
Centric Diatoms							**Centric Diatoms**						
Aulacoseira	0	30	0	0	0	45	*Aulacoseira*	0.0	9.0	0.0	0.0	0.0	13.5
Cyclotella	33	0	0	0	60	105	*Cyclotella*	82.5	0.0	0.0	0.0	6.0	10.5
Urosolenia	0	0	0	0	300	45	*Urosolenia*	0.0	0.0	0.0	0.0	360.0	54.0
Araphid Pennate Diatoms							**Araphid Pennate Diatoms**						
Asterionella	0	30	0	0	0	0	*Asterionella*	0.0	6.0	0.0	0.0	0.0	0.0
Fragilaria/related taxa	264	90	0	144	0	0	*Fragilaria/related taxa*	79.2	27.0	0.0	43.2	0.0	0.0
Synedra	99	90	90	96	360	0	*Synedra*	79.2	72.0	72.0	76.8	288.0	0.0
Monoraphid Pennate Diatoms							**Monoraphid Pennate Diatoms**						
Biraphid Pennate Diatoms							**Biraphid Pennate Diatoms**						
Gomphonema/related taxa	0	0	0	0	30	0	*Gomphonema/related taxa*	0.0	0.0	0.0	0.0	30.0	0.0
Nitzschia	0	0	0	0	90	0	*Nitzschia*	0.0	0.0	0.0	0.0	72.0	0.0
CHLOROPHYTA							**CHLOROPHYTA**						
Flagellated Chlorophytes							**Flagellated Chlorophytes**						
Chlamydomonas	0	0	0	48	0	0	*Chlamydomonas*	0.0	0.0	0.0	4.8	0.0	0.0
Pandorina	0	0	0	0	0	240	*Pandorina*	0.0	0.0	0.0	0.0	0.0	24.0
Coccoid/Colonial Chlorophytes							**Coccoid/Colonial Chlorophytes**						
Closteriopsis	0	0	0	0	0	15	*Closteriopsis*	0.0	0.0	0.0	0.0	0.0	7.5
Coelastrum	0	0	0	192	0	0	*Coelastrum*	0.0	0.0	0.0	38.4	0.0	0.0
Elakatothrix	0	0	0	0	60	0	*Elakatothrix*	0.0	0.0	0.0	0.0	6.0	0.0
Golenkinia	33	0	45	96	210	120	*Golenkinia*	6.6	0.0	9.0	19.2	42.0	24.0
Kirchneriella	0	0	0	192	0	0	*Kirchneriella*	0.0	0.0	0.0	19.2	0.0	0.0
Oocystis	0	0	0	0	120	30	*Oocystis*	0.0	0.0	0.0	0.0	48.0	12.0
Pediastrum	132	0	0	0	0	0	*Pediastrum*	26.4	0.0	0.0	0.0	0.0	0.0
Scenedesmus	132	180	0	384	240	120	*Scenedesmus*	13.2	18.0	0.0	38.4	24.0	12.0
Selenastrum	33	0	0	0	0	0	*Selenastrum*	3.3	0.0	0.0	0.0	0.0	0.0
Sphaerocystis	264	0	0	0	0	0	*Sphaerocystis*	52.8	0.0	0.0	0.0	0.0	0.0
Tetraedron	231	240	90	216	120	1455	*Tetraedron*	138.6	144.0	54.0	129.6	72.0	873.0
Tetrastrum	0	0	0	0	0	60	*Tetrastrum*	0.0	0.0	0.0	0.0	0.0	12.0
Filamentous Chlorophytes							**Filamentous Chlorophytes**						
Desmids							**Desmids**						
Cosmarium	0	0	23	0	30	0	*Cosmarium*	0.0	0.0	18.0	0.0	24.0	0.0
Staurastrum	0	0	0	0	60	0	*Staurastrum*	0.0	0.0	0.0	0.0	48.0	0.0
CHRYSOPHYTA							**CHRYSOPHYTA**						
Flagellated Classic Chrysophytes							**Flagellated Classic Chrysophytes**						
Chromulina	99	90	0	0	0	0	*Chromulina*	5.0	4.5	0.0	0.0	0.0	0.0
Dinobryon	0	0	0	72	0	30	*Dinobryon*	0.0	0.0	0.0	216.0	0.0	90.0
Mallomonas	132	60	23	48	0	105	*Mallomonas*	66.0	30.0	11.3	24.0	0.0	52.5
Non-Motile Classic Chrysophytes							**Non-Motile Classic Chrysophytes**						
Haptophytes							**Haptophytes**						
Tribophytes/Eustigmatophytes							**Tribophytes/Eustigmatophytes**						
Raphidophytes							**Raphidophytes**						
CRYPTOPHYTA							**CRYPTOPHYTA**						
Cryptomonas	66	30	45	72	60	90	*Cryptomonas*	13.2	6.0	9.0	48.0	12.0	60.0
CYANOPHYTA							**CYANOPHYTA**						
Unicellular and Colonial Forms							**Unicellular and Colonial Forms**						
Aphanocapsa	5610	1200	0	0	0	0	*Aphanocapsa*	56.1	12.0	0.0	0.0	0.0	0.0
Filamentous Nitrogen Fixers							**Filamentous Nitrogen Fixers**						
Anabaena	0	0	1350	720	0	0	*Anabaena*	0.0	0.0	270.0	144.0	0.0	0.0
Aphanizomenon	0	0	9450	4320	7200	0	*Aphanizomenon*	0.0	0.0	1228.5	561.6	936.0	0.0
Filamentous Non-Nitrogen Fixers							**Filamentous Non-Nitrogen Fixers**						
Planktolyngbya	0	0	108000	26400	96000	0	*Planktolyngbya*	0.0	0.0	1080.0	264.0	960.0	0.0
Pseudanabaena	0	0	1350	20160	15000	900	*Pseudanabaena*	0.0	0.0	13.5	201.6	150.0	9.0
EUGLENOPHYTA							**EUGLENOPHYTA**						
Trachelomonas	0	0	0	24	0	0	*Trachelomonas*	0.0	0.0	0.0	24.0	0.0	0.0
PYRRHOPHYTA							**PYRRHOPHYTA**						
Peridinium	0	0	23	24	60	60	*Peridinium*	0.0	0.0	47.3	50.4	126.0	769.5


2011 Middle Pond Phytoplankton
Algal Biomass (ug/L)
3500
3000
2500
2000
1500
1000
500
0
6/26/11
6/26/11
8/6/11
8/6/11
8/30/11
9/22/11
MP-1.0
MP-1.6
MP-1.0
MP-1.7
MP-1.0
MP-1.0
Date and Stations
PYRRHOPHYTA
EUGLENOPHYTA
CYANOPHYTA
CRYPTOPHYTA
CHRYSOPHYTA
CHLOROPHYTA
BACILLARIOPHYTA

	ZOOPLANKTON DENSITY (#/L)			ZOOPLANKTON BIOMASS (UG/L)	
	Middle	Middle		Middle	Middle
	MP-1	MP-1		MP-1	MP-1
TAXON	6/26/11	9/22/11	TAXON	6/26/11	9/22/11
PROTOZOA			**PROTOZOA**		
Ciliophora	0.0	0.0	**Ciliophora**	0.0	0.0
Mastigophora	0.0	0.0	**Mastigophora**	0.0	0.0
Sarcodina	0.0	0.0	**Sarcodina**	0.0	0.0
ROTIFERA			**ROTIFERA**		
Asplanchna	0.6	0.0	*Asplanchna*	0.9	0.0
Conochilus	1.7	5.2	*Conochilus*	0.1	0.2
Keratella	0.1	0.0	*Keratella*	0.0	0.0
Polyarthra	0.0	0.1	*Polyarthra*	0.0	0.0
Trichocerca	0.1	0.1	*Trichocerca*	0.0	0.0
COPEPODA			**COPEPODA**		
Copepoda-Cyclopoida			**Copepoda-Cyclopoida**		
Mesocyclops	0.2	0.1	*Mesocyclops*	0.6	0.1
Copepoda-Calanoida			**Copepoda-Calanoida**		
Diaptomus	1.6	0.0	*Diaptomus*	0.8	0.0
Other Copepoda-Nauplii	0.5	0.2	**Other Copepoda-Nauplii**	1.3	0.6
CLADOCERA			**CLADOCERA**		
Bosmina	0.0	13.1	*Bosmina*	0.0	12.8
Ceriodaphnia	0.0	0.4	*Ceriodaphnia*	0.0	1.1
Diaphanosoma	0.0	0.0	*Diaphanosoma*	0.0	0.0
Holopedium	0.1	0.0	*Holopedium*	0.8	0.0
OTHER ZOOPLANKTON			**OTHER ZOOPLANKTON**		
SUMMARY STATISTICS			**SUMMARY STATISTICS**		
DENSITY			**BIOMASS**		
PROTOZOA	0.0	0.0	PROTOZOA	0.0	0.0
ROTIFERA	2.5	5.4	ROTIFERA	1.0	0.2
COPEPODA	2.3	0.3	COPEPODA	2.7	0.7
CLADOCERA	0.1	13.5	CLADOCERA	0.8	14.0
OTHER ZOOPLANKTON	0.1	0.0	OTHER ZOOPLANKTON	1.0	0.0
TOTAL ZOOPLANKTON	5.0	19.3	TOTAL ZOOPLANKTON	5.6	14.9
TAXONOMIC RICHNESS					
PROTOZOA	0	0			
ROTIFERA	4	3			
COPEPODA	3	2			
CLADOCERA	1	2			
OTHER ZOOPLANKTON	1	0			
TOTAL ZOOPLANKTON	9	7			
S-W DIVERSITY INDEX	0.72	0.37			
EVENNESS INDEX	0.75	0.43			
MEAN LENGTH (mm): ALL FORMS	0.40	0.25			
MEAN LENGTH: CRUSTACEANS	0.63	0.31			