# Development of an Algae Management Plan for the Congamond Lakes, Southwick, Massachusetts and Suffield, Connecticut

**Prepared by**
**Water Resource Services, Inc.**

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## Introduction and Background

Congamond Lakes include three waterbodies known as North, Middle and South Pond, separated by two large culverts through which boats can pass and over which roads traverse. While the ponds are located in Southwick, Massachusetts, the land abutting the east side of South Pond and most of the east side of Middle Pond is in Suffield, Connecticut (Figure 1). North Pond covers 47 acres (19 hectares) to a maximum depth of 44 feet (12.7 m) with an average depth of 12.2 feet (3.7 m). Middle Pond covers 278 acres (112 hectares) to a maximum depth of 45 feet (13.6 m) with an average depth of 17.8 feet (5.4 m). South Pond covers 147 acres (59.3 hectares) to a maximum depth of 26 feet (7.8 m) with an average depth of 14.1 feet (4.3 m).

Largely natural, these ponds have been slightly altered by outlet manipulations that include construction of a canal heading south from South Pond which provided a commercial waterway over a century ago but is now largely filled in and non-navigable. The natural outlet, Great Brook, flows out of the south end of Middle Pond to the west and the north, but has also filled in to the extent that most water still leaves through the canal. The watershed is relatively small at 3.5 square miles (2222 acres), compared to 472 acres for the three ponds combined, and if water levels rise or fall, they do not equilibrate quickly. The water level is normally between 224 and 225.5 feet above mean sea level.

Human alteration of the shoreline goes back at least 200 years with various construction projects of commercial or recreational nature, including ice houses and an amusement park. Shoreline land is now mostly private residences with some commercial establishments (e.g., dining, marine services) and recreational facilities (e.g., boat launches, beaches). The shoreline slopes steeply in most areas. The primary current use of the lake is recreation, with powerboating most popular in summer. Fishing is also very popular, and the MA DFW stocks the lake and tracks fish populations with periodic surveys. Swimming is popular, but variable and sometimes low summer water clarity impairs this use to some degree.

The watershed of each pond (Figure 2) includes direct drainage with no major permanent tributaries and any upstream pond other than the Congamond Lakes themselves; North Pond flows to Middle Pond which flows to South Pond, although at high water there can be flow from Middle Pond to Great Brook, and North Pond has been known to break out to the north and flow to Great Brook at least once in the past. There are substantial wetlands near South Pond on both sides and some wetland area draining to Middle Pond from the east. There is some agricultural land draining to Middle Pond from the east as well. There is wooded land in the drainage area of each pond, but residential areas and related supporting commercial uses are the dominant land use in each small drainage area.

Congamond Lakes have experienced problems with both algae and rooted plants for several decades. Both invasive (e.g., Eurasian watermilfoil) and native plant nuisances have occurred, and cyanobacteria blooms have been encountered in many summers. Management has included mainly herbicide and algaecide applications (Table 1) with reasonable success but necessary repeated applications. While rooted plant production remains substantial each summer as a function of light penetrating to a hospitable bottom in shallow water, invasive milfoil has been minimized. Algae blooms have been counteracted once they occur, but have not been routinely prevented.

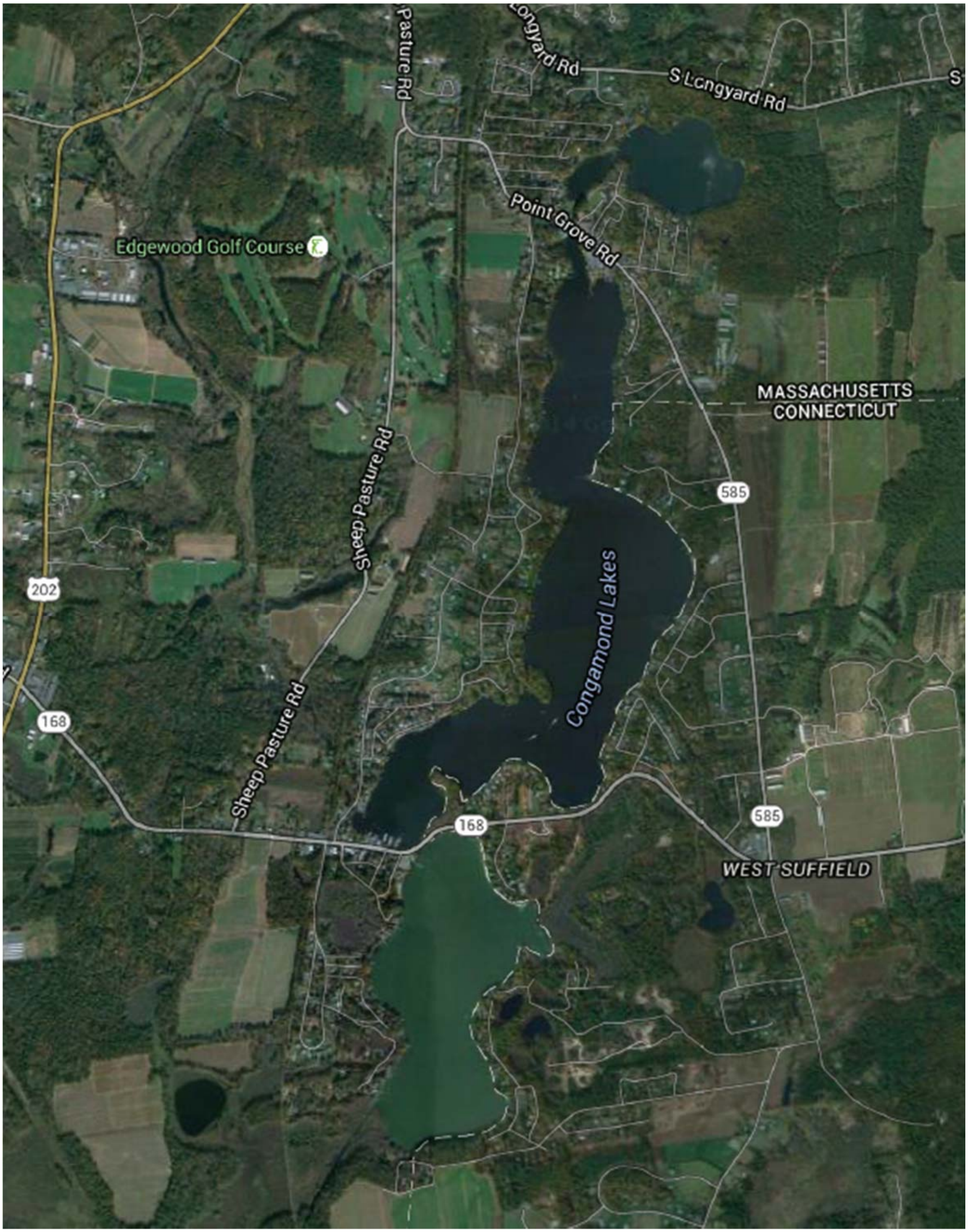



**Figure 1. Congamond Lakes and General Vicinity**

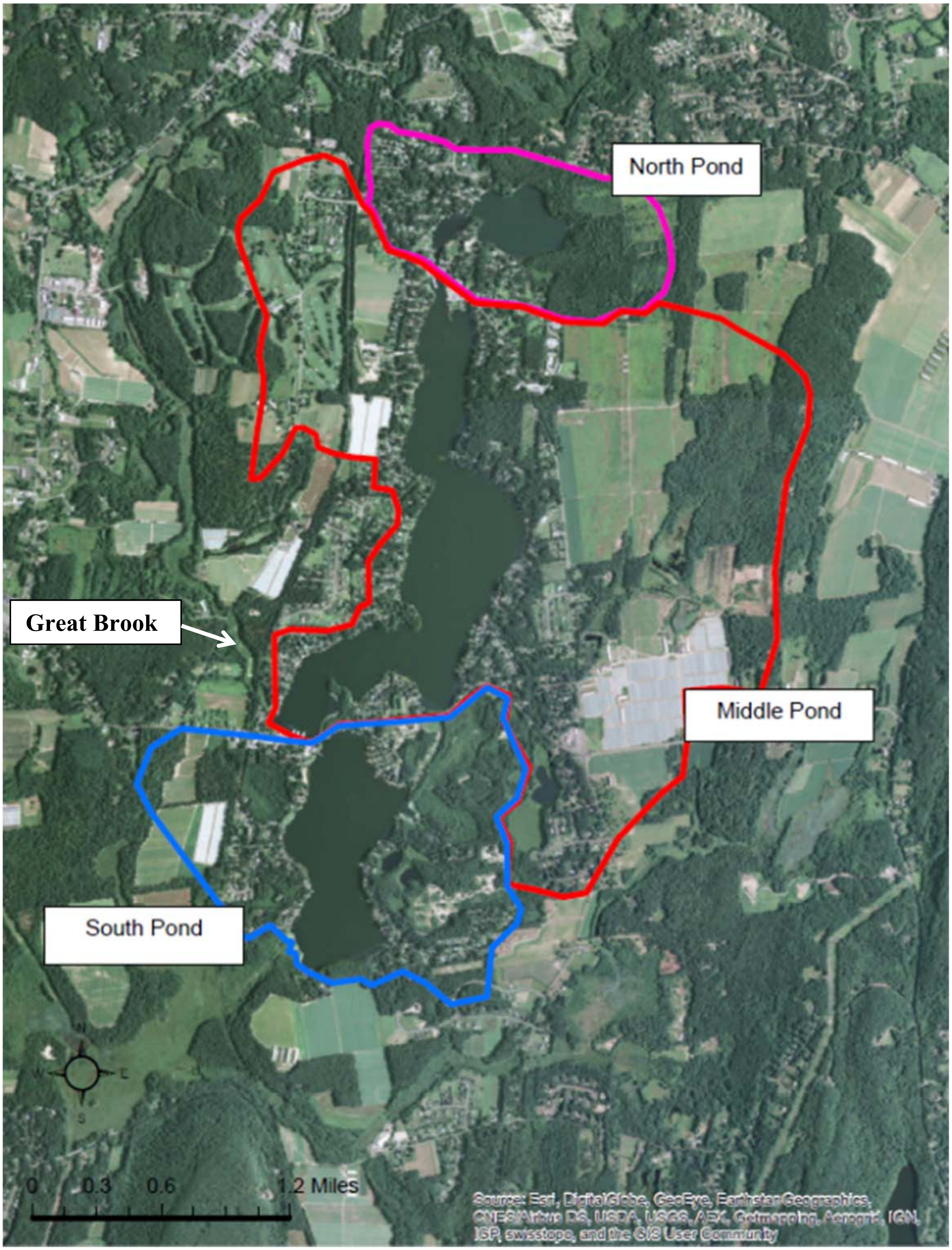



**Figure 2. Congamond Lakes Watershed**

**Table 1. Herbicide/Algaecide Treatment History of Congamond Lakes**

| Date | Acres Treated with Fluridone | Acres Treated with Diquat | Acres Treated with Copper |
|---|---|---|---|
| 5/19/2015 | | 15.5 | |
| 6/15/2015 | | | 210 |
| 5/21/2014 | | 18 | |
| 8/1/2013 | | | 185 |
| 5/14/2013 | | 32 | |
| 6/27/2012 | | | 71 |
| 5/14/2012 | | 29.9 | |
| 5/31/2011 | | 25 | |
| 8/9/2010 | | | 185 |
| 5/24/2010 | | 9.7 | |
| 5/26/2009 | | 25 | |
| 7/24/2008 | | | 71 |
| 5/27/2008 | | 30 | |
| 5/29/2007 | | 23.5 | |
| 5/31/2006 | | 31 | |
| 6/6/2005 | | 34.5 | |
| 6/28/2004 | | 35 | |
| 6/9/2003 | | 33.5 | |
| 6/10/2002 | | 34 | |
| 6/7/2001 | 465 | | |

The Congamond Lakes were treated in 1989 with aluminum, one of the earlier applications of this algae control approach in New England, to bind available phosphorus and reduce algae productivity. The prescribed dose was 30,000 gallons of aluminum sulfate and 17,200 gallons of sodium aluminate, representing a total aluminum input of about 16,500 kg of aluminum. Over the area that appears to need treatment now, that amount of aluminum would have translated into a dose of 26 $g/m^2$, a dose that should have improved conditions with regard to algae blooms for multiple years, but is lower than what appears to be needed now. Yet monitoring was limited and no follow up report has been located. The effect would eventually wear off, and cyanobacteria blooms have occurred in most summers over the last decade.

The Lake Management Committee has performed some monitoring on its own and had Northeast Aquatic Research supply professional services intended to support monitoring and management programs from 2009 through 2013. In 2015 the Lake Management Committee retained Water Resource Services (WRS) to direct, support and supplement a volunteer monitoring program with the intent of developing a management plant focused on reducing algae nuisances, specifically cyanobacteria blooms. This report details the results of this effort.

## Methods and Approach

WRS assisted the Lake Management Committee with development of a rational monitoring program that would supply the water quality and supplemental data necessary to a proper review of algae control options. Key elements included:

1. Temperature and oxygen profiles – Measurement of temperature and dissolved oxygen at 1 m intervals from the surface to the bottom at the deepest point in each pond (Figure 3) roughly every week at least through May and twice monthly thereafter, to support estimation of oxygen demand. Conductivity and pH were also assessed using a multi-probe YSI sonde. Secchi transparency was measured from the water surface.
2. Nutrient concentrations – Monthly sampling and lab assessment of ammonium, nitrate, total Kjeldahl nitrogen (TKN), total phosphorus and dissolved phosphorus near the surface and bottom of each pond from the stations at which temperature and oxygen profiles were assessed. Sampling was conducted from May through October. Testing was provided by Microbac Laboratories. Turbidity was also assessed for these samples with a handheld meter.
3. Algae – Algae samples were collected monthly with nutrient samples, preserved in the field, and delivered to WRS for microscopic analysis, which included identification and quantification of phytoplankton. Additional samples were either provided to WRS or photographed by a Lake Management Committee representative with pictures examined by WRS for identification and response to possible problems. Turbidity and chlorophyll-a measurements were also made in conjunction with algae assessment, including more frequent measurements off a dock in South Pond. Turbidity and chlorophyll-a were measured with handheld meters, the former based on light scattering and the latter as fluorescence.
4. Zooplankton – Zooplankton were sampled twice in each pond, once in May and once in October by WRS using a plankton net with 80 um mesh towed through the water until at least 380 L were concentrated. Samples were examined microscopically by WRS with identification and enumeration of zooplankton.
5. Storm water – First flush samples were collected from ditches and other storm drainage systems (Figure 4) which flow only during wet weather; these are the dominant surface water inputs to the Congamond Lakes. The Lake Management Committee selected sampling locations and performed the sampling after training from WRS, with up to 8 stations sampled per storm. Samples were tested by Microbac Laboratories for the same features as in the nutrient concentration (and turbidity) assessment above. Three storms were assessed.
6. Sediment - Sediment from multiple locations (Figure 5) were obtained with an Ekman dredge, with the upper 10 cm of surficial sediment collected. Testing was conducted by Northeast Laboratories for percent solids, total phosphorus, and iron-bound phosphorus to facilitate assessment of potential internal loading of phosphorus. Additionally, aluminum assays were conducted by Northeast Laboratories on one composite sample from each pond to evaluate the reduction in iron-bound phosphorus with increasing aluminum application. Small amounts of sediment are treated with aluminum compounds and then retested for iron-bound phosphorus. This allows a projection of the level of reduction in internal loading that might be obtained by an inactivation treatment of surficial sediment and facilitates dose planning and cost estimation. Further, core samples were collected from two locations in the outlet canal (Figure 5) and tested for a suite of features used to screen for possible disposal issues relating to dredging. Tests were conducted by Spectrum Analysis for metals (copper, lead, nickel, zinc), extractable petroleum hydrocarbons (3 fractions), polynuclear aromatic hydrocarbons, pesticides, total organic carbon, and percent solids. Additional testing is

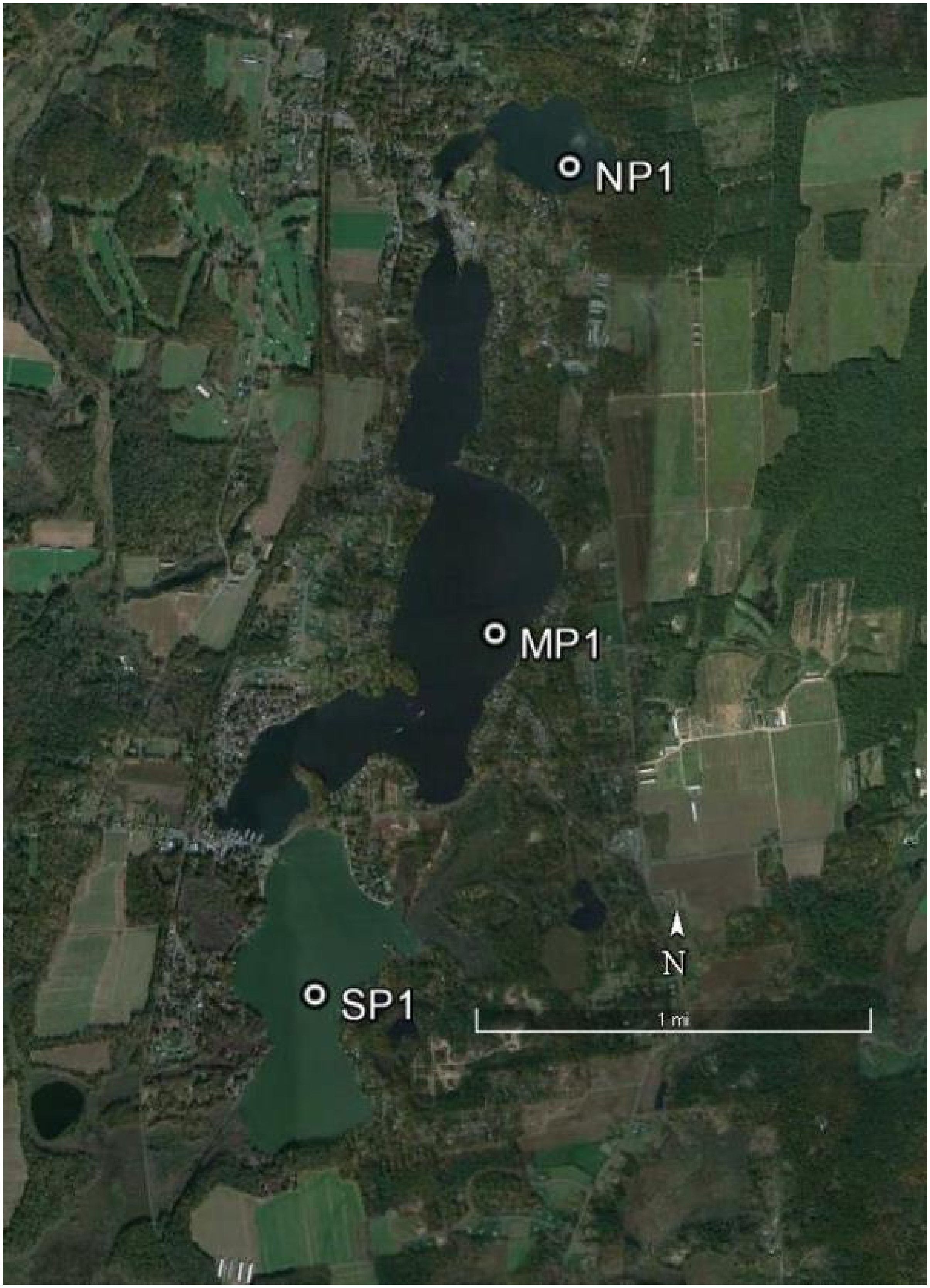
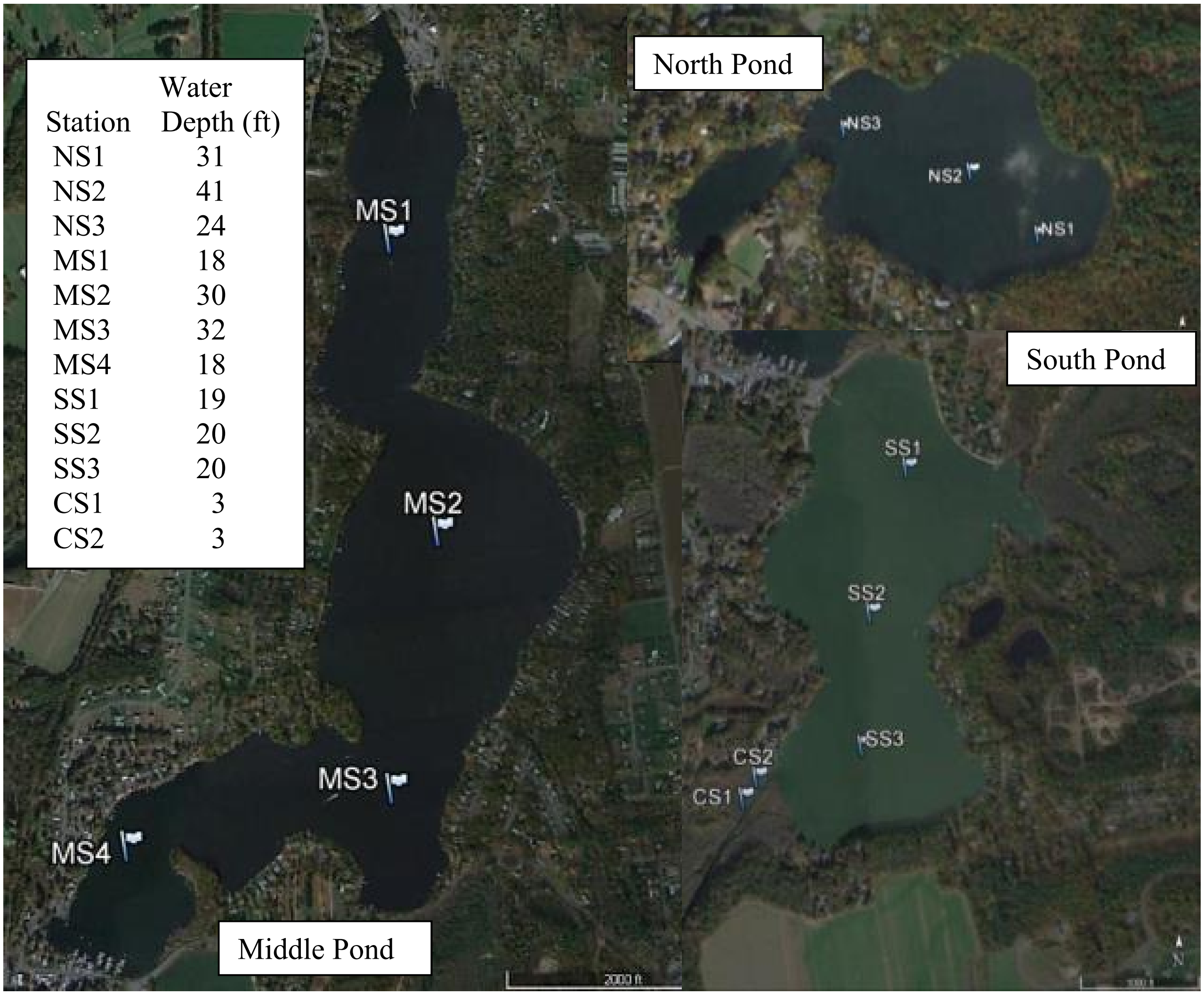



**Figure 3. Congamond Lakes Sampling Locations**


**Figure 4. Congamond Lakes Storm Water Sampling Locations**




| Station | Water Depth (ft) |
|---|---|
| NS1 | 31 |
| NS2 | 41 |
| NS3 | 24 |
| MS1 | 18 |
| MS2 | 30 |
| MS3 | 32 |
| MS4 | 18 |
| SS1 | 19 |
| SS2 | 20 |
| SS3 | 20 |
| CS1 | 3 |
| CS2 | 3 |

**Figure 5. Congamond Lakes Sediment Sampling Locations**

required for a full dredging feasibility analysis, but this screening approach usually identifies the highest probability issues for sediment quality and related disposal restrictions.

7. Lake and watershed model - In order to evaluate the relative magnitude of loading from identified sources and assess the potential benefits from various watershed and in-lake management approaches, the Lake Loading Response Model (LLRM, AECOM 2009) was applied. LLRM is a spreadsheet model that allows actual data to be used to set variable values and provide reality checks on calculated values at key points in the modeling process. The most recently available land use data and the 2015 water quality data were applied, tempered with longer term average values from annual monitoring in recent years. Loading coefficients and attenuation factors were adjusted until the predicted input and in-lake values were close to actual values. Once the model was considered to represent current conditions in the ponds, changes in external and internal loading were evaluated to determine the potential impact from possible management actions.

Resulting data and calculations were then used by WRS to evaluate the condition of Congamond Lakes and causative agents, followed by an evaluation of applicable management methods for addressing algae issues.

## Lake Morphometry

Water depth in the Congamond Lakes does not appear to have been assessed lately. Maps produced over the last decade or two show some possible updates, but appear largely based on older maps. North Pond bathymetry (Figure 6) appears accurate in comparison with observations made in 2015, but there may be slight differences in Middle and South Pond (Figures 6 and 7) with regard to the maximum depth and its location. Depth in Middle Pond to at least 45 feet (13.6 m) has been recorded in temperature-oxygen profiles, with the sampling station (Figure 3) slightly east of the deepest area shown in Figure 7. It could be that the boat was not anchored and the cable was at an angle, but a difference of close to 10 ft (3 m) for maximum depth is substantial. The maximum depth in South Pond appears consistent between maps and field data, but the deepest zone may extend slightly south of where it is shown on Figure 7. Calculation of areas and volumes associated with each pond is not greatly affected by these discrepancies, however, as the affected areas are small relative to the whole lake area.

The area and volume associated with each 3.3 foot (1 m) depth layer of each lake is provided in Table 2. These values are important when considering the area and volume subjected to low oxygen or possibly contributing to internal phosphorus release. They are also useful when evaluating areas of possible treatment for improved oxygen, lowered phosphorus loading, or rooted plant control. We have seen no such set of calculations for these ponds elsewhere, and while an updated bathymetric map may alter these areas and volumes slightly, the values in Table 2 appear to be the best available for management planning at this time.

Middle Pond holds the most water of the three ponds by far, having a substantially greater area and average depth. South Pond holds less than half the volume of Middle Pond, and North Pond holds less than a third of the volume of South Pond. North and South Ponds together hold just slightly more than half the volume of Middle Pond. With small watersheds, these ponds will have long detention times and will have a low rate of flushing. This means that any contaminant inputs are likely to remain in these ponds and may be recycled periodically.

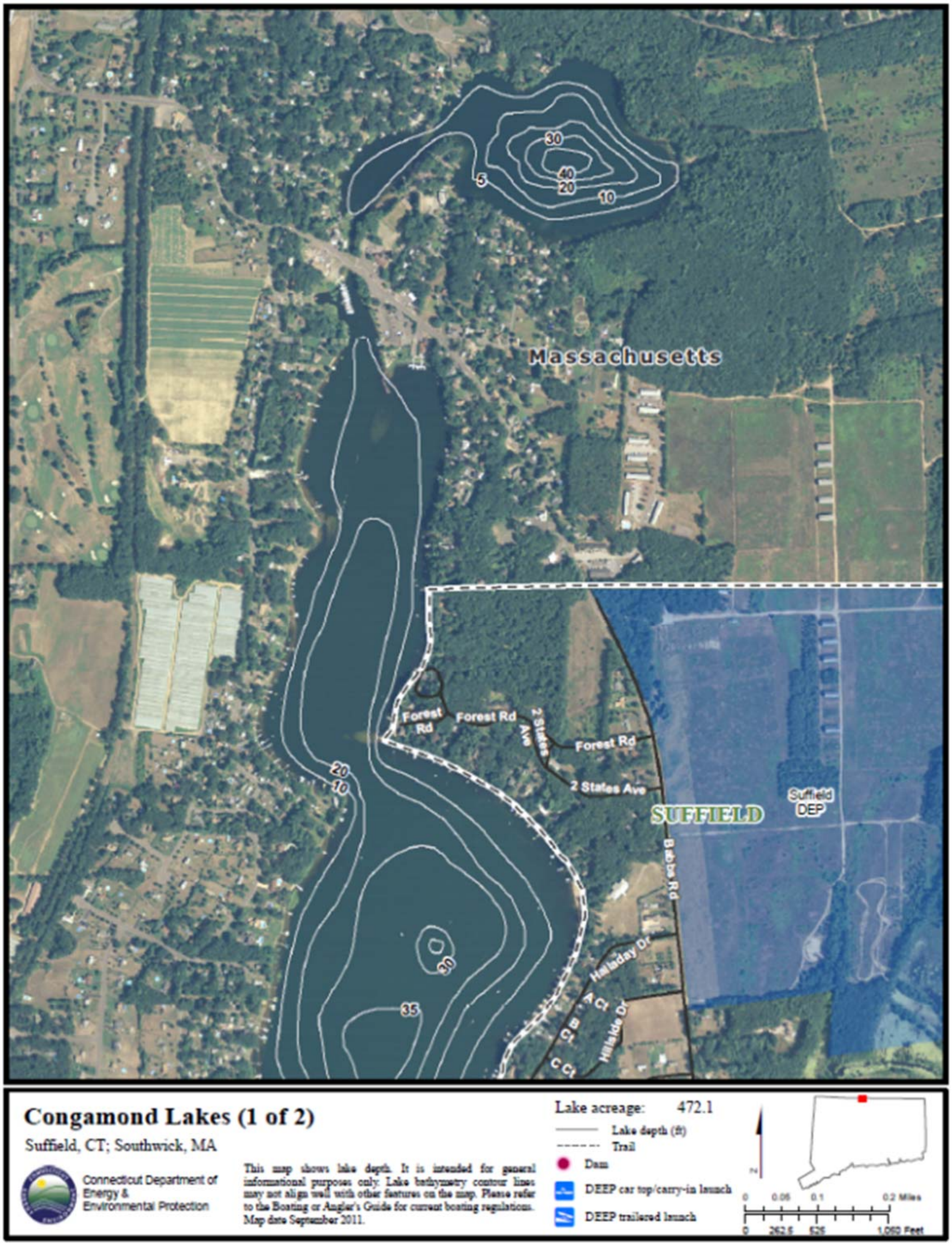



**Figure 6. Congamond Lakes Bathymetry – Northern Half**

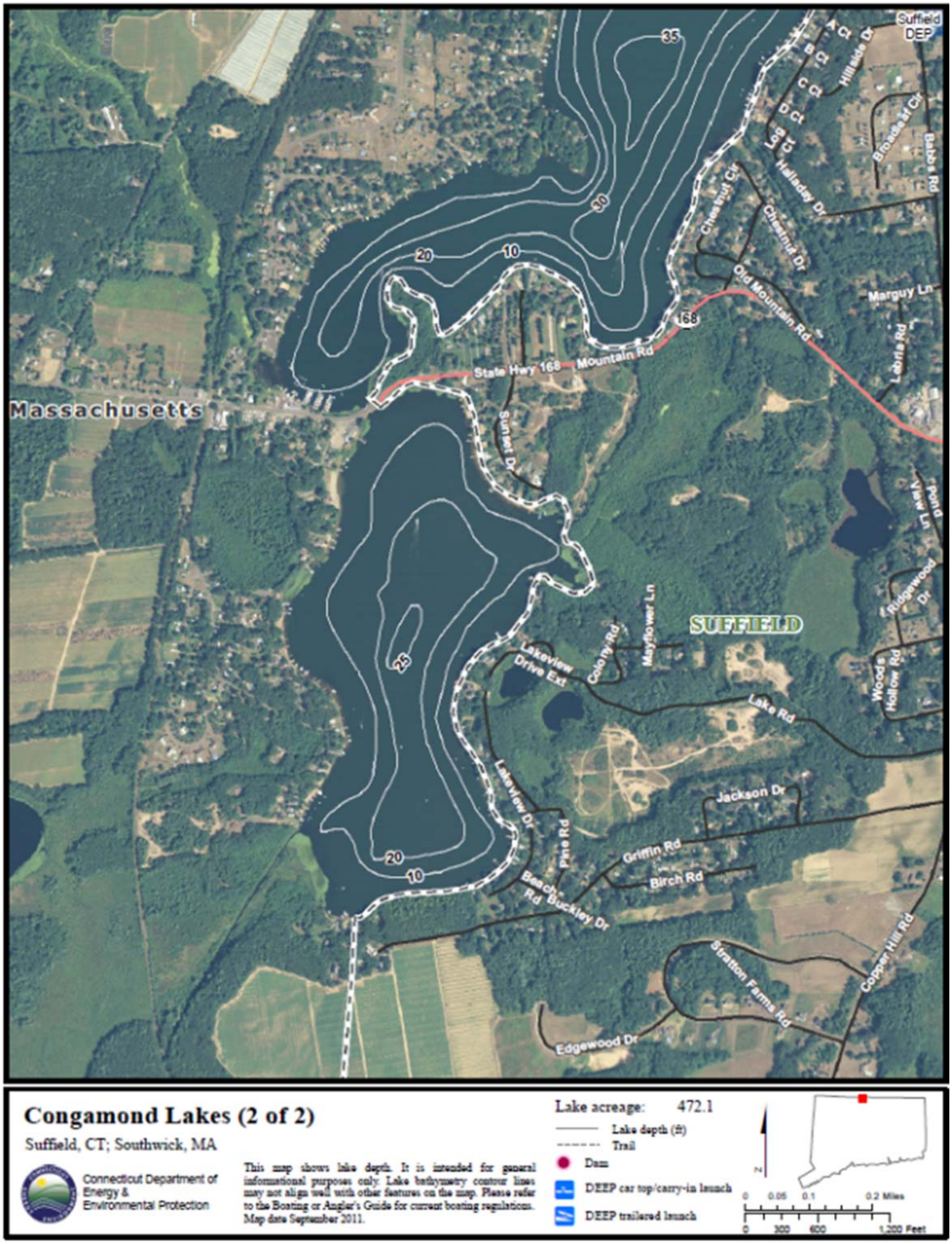



**Figure 7. Congamond Lakes Bathymetry – Southern Half**

## Table 2. Area and Volume Associated with Water Depth in the Congamond Lakes

**North Pond**

| Depth Range (m) | Area (ac) | Area (m2) | Layer Volume (ac-ft) | Layer Volume (m3) | Cumulative Volume (ac-ft) | Cumulative Volume (m3) |
|---|---|---|---|---|---|---|
| 0-1 | 47.0 | 189516 | 154 | 189516 | 572 | 703011 |
| 1-2 | 35.4 | 142742 | 116 | 142742 | 418 | 513495 |
| 2-3 | 24.9 | 100269 | 82 | 100269 | 302 | 370753 |
| 3-4 | 15.4 | 62097 | 51 | 62097 | 220 | 270484 |
| 4-5 | 12.8 | 51613 | 42 | 51613 | 170 | 208387 |
| 5-6 | 10.2 | 41129 | 33 | 41129 | 128 | 156774 |
| 6-7 | 7.6 | 30645 | 25 | 30645 | 94 | 115645 |
| 7-8 | 6.3 | 25269 | 21 | 25269 | 69 | 85000 |
| 8-9 | 4.9 | 19892 | 16 | 19892 | 49 | 59731 |
| 9-10 | 3.6 | 14516 | 12 | 14516 | 32 | 39839 |
| 10-11 | 2.8 | 11344 | 9 | 11344 | 21 | 25323 |
| 11-12 | 2.0 | 8172 | 7 | 8172 | 11 | 13978 |
| 12-13 | 1.2 | 5000 | 4 | 5000 | 5 | 5806 |
| 13+ | 0.4 | 1613 | 1 | 806 | 1 | 806 |

**Middle Pond**

| Depth Range (m) | Area (ac) | Area (m2) | Layer Volume (ac-ft) | Layer Volume (m3) | Cumulative Volume (ac-ft) | Cumulative Volume (m3) |
|---|---|---|---|---|---|---|
| 0-1 | 278.0 | 1120968 | 911.8 | 1120968 | 4948.2 | 6082999 |
| 1-2 | 244.9 | 987690 | 803.4 | 987690 | 4036.3 | 4962031 |
| 2-3 | 211.9 | 854412 | 695.0 | 854412 | 3232.9 | 3974341 |
| 3-4 | 178.8 | 721134 | 586.6 | 721134 | 2537.9 | 3119929 |
| 4-5 | 153.0 | 616856 | 501.8 | 616856 | 1951.3 | 2398795 |
| 5-6 | 127.1 | 512577 | 417.0 | 512577 | 1449.5 | 1781939 |
| 6-7 | 101.3 | 408298 | 332.1 | 408298 | 1032.6 | 1269362 |
| 7-8 | 81.4 | 328307 | 267.1 | 328307 | 700.4 | 861064 |
| 8-9 | 61.6 | 248316 | 202.0 | 248316 | 433.4 | 532757 |
| 9-10 | 41.7 | 168325 | 136.9 | 168325 | 231.4 | 284440 |
| 10+ | 28.8 | 116115 | 94.5 | 116115 | 94.5 | 116115 |

**South Pond**

| Depth Range (m) | Area (ac) | Area (m2) | Layer Volume (ac-ft) | Layer Volume (m3) | Cumulative Volume (ac-ft) | Cumulative Volume (m3) |
|---|---|---|---|---|---|---|
| 0-1 | 147.0 | 592742 | 482.2 | 592742 | 2071.3 | 2546395 |
| 1-2 | 125.1 | 504301 | 410.2 | 504301 | 1589.2 | 1953653 |
| 2-3 | 103.1 | 415860 | 338.3 | 415860 | 1179.0 | 1449352 |
| 3-4 | 81.2 | 327419 | 266.3 | 327419 | 840.7 | 1033491 |
| 4-5 | 65.8 | 265457 | 215.9 | 265457 | 574.3 | 706072 |
| 5-6 | 50.5 | 203495 | 165.5 | 203495 | 358.4 | 440615 |
| 6-7 | 35.1 | 141532 | 115.1 | 141532 | 192.9 | 237120 |
| 7-8 | 23.7 | 95588 | 77.8 | 95588 | 77.8 | 95588 |

## Lake Water Quality

Temperature-oxygen profiles (Figures 8-11) reveal much about how lakes change over the course of a year. Data prior to 2015 (Figure 8) demonstrate anoxia in all three ponds, but profiles were not collected at a frequency that allows the extent and duration of anoxia to be assessed in each year. The water tends to be mixed from top to bottom in early spring, with only a slight decrease in temperature or oxygen with increasing depth. As spring progresses, the combination of the heating of the surface by the sun and limitation on the depth of wind mixing cause the ponds to thermally stratify, creating an upper, well-lit, warmer layer (epilimnion) and a lower, dark, colder layer (hypolimnion). The boundary between the two (thermocline) is sometimes sharp and sometimes more of a gradual transition (metalimnion).

The bottom of the pond in contact with the upper layer has access to oxygen from above, may vary substantially in composition from rock to gravel to sand to muck, and is where rooted plants grow. Sometimes decomposition can exceed oxygen resupply, and lower oxygen levels can occur near the bottom, but usually this zone has adequate oxygen for habitat and to minimize undesirable interactions between sediment and the overlying water. Sediment may be resuspended by wind or boats, but chemical release of phosphorus, accumulation of ammonium, generation of hydrogen sulfide, and other reactions associated with anoxia should be very limited in this zone.

The bottom in contact with the lower layer may also vary in composition, but tends to have higher muck content and is functionally isolated from sources of oxygen. The decomposition of organic matter in the muck sediments may exhaust the oxygen supply in the overlying water, leading to lowered oxygen levels to the point of absence (anoxia). The sediment-water interface is often anoxic, and the depth at which anoxia occurs will rise over the period of stratification, possibly including the entire hypolimnion. Under anoxia, reactions in the surficial sediment cause iron and phosphorus to dissociated and become soluble, potentially moving into the overlying water and increasing their concentrations. In that deep, dark zone, algae production will be low, but if the anoxic interface reaches the thermocline, there is enough light to allow blooms to form. In the area where sediment is anoxic but shallow enough to receive some light, algae may grow at the sediment-water interface and then rise in the water column to form blooms, possibly even when the overlying water has low nutrient levels, as the algae can have excess nutrients in their cells from their time near the bottom.

Additional issues with anoxia in the hypolimnion include accumulation of ammonium, which needs oxygen to convert first to nitrite and then to nitrate through biochemical reactions, and generation of hydrogen sulfide, a product of biochemistry in strongly anoxic sediments. None of the interactions of sediment and overlying water during anoxia are beneficial to valued lake uses, ranging from water supply to recreation to fish habitat. A small area and volume of anoxia will not be an overwhelming influence on the whole lake, but as the area and volume rise, the impact increases.

Based on the 2015 profiles for the Congamond Lakes (Figures 9-11), each pond experiences anoxia during the summer. The bottom of North Pond takes longer to go anoxic and the entire hypolimnion is not affected; the minimum depth of anoxia is about 8 m (Figure 9) , which translates into an area of about 5 acres (10.5% of the total pond area) and a volume of around 50

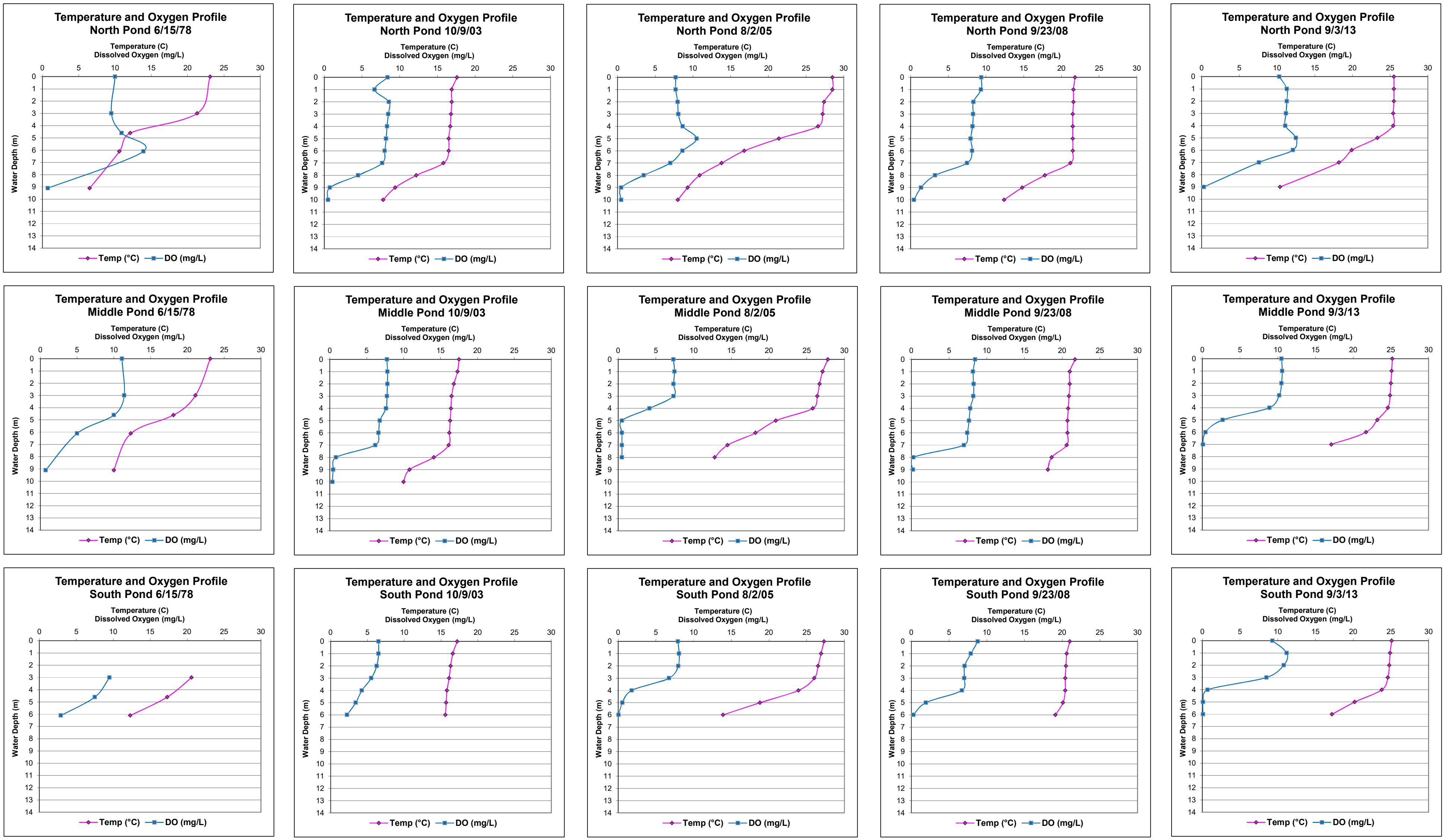



**Figure 8. Selected Temperature-Oxygen Profiles from Congamond Lakes Prior to 2015**

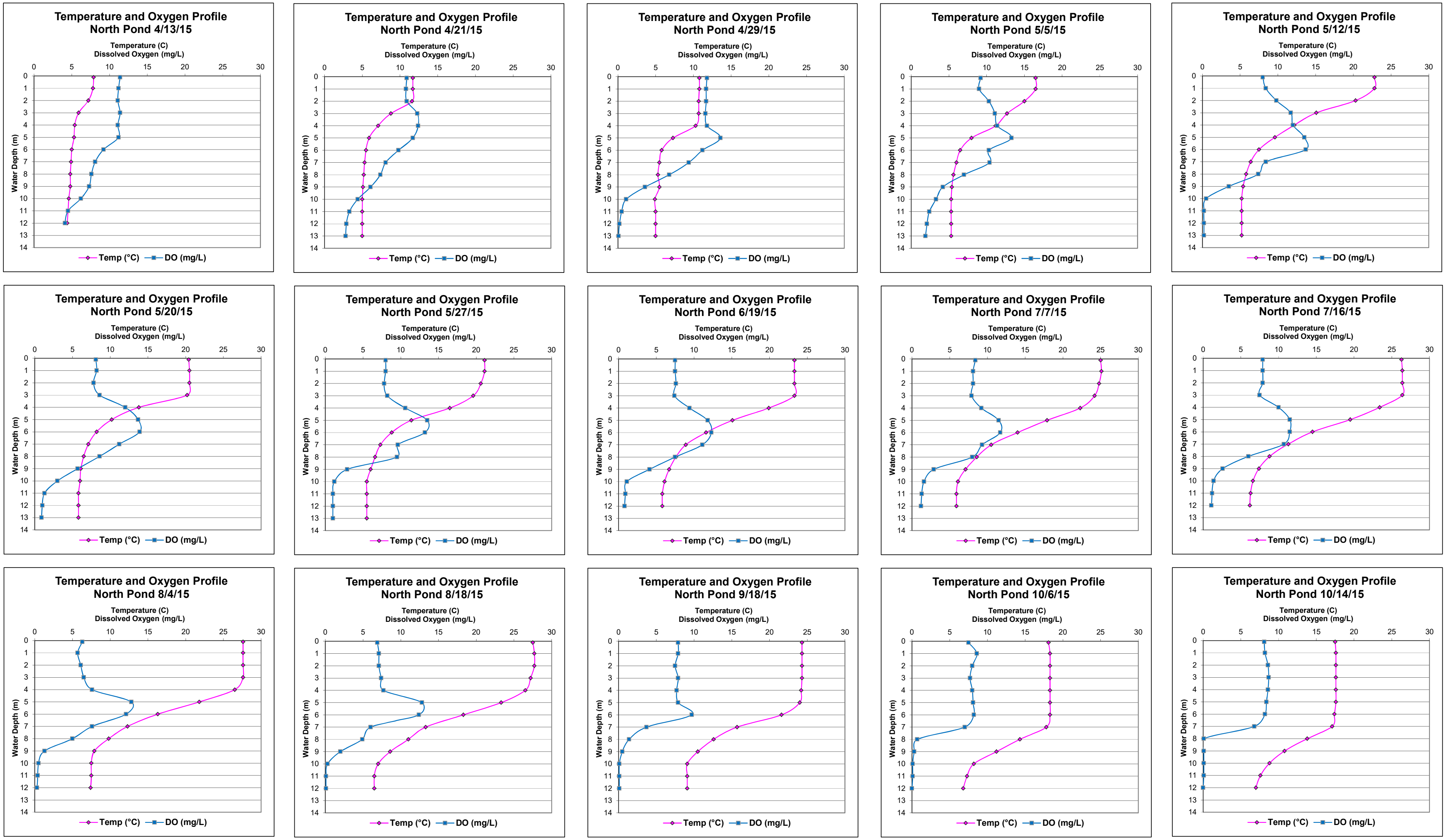



**Figure 9. North Pond Temperature-Oxygen Profiles for 2015**

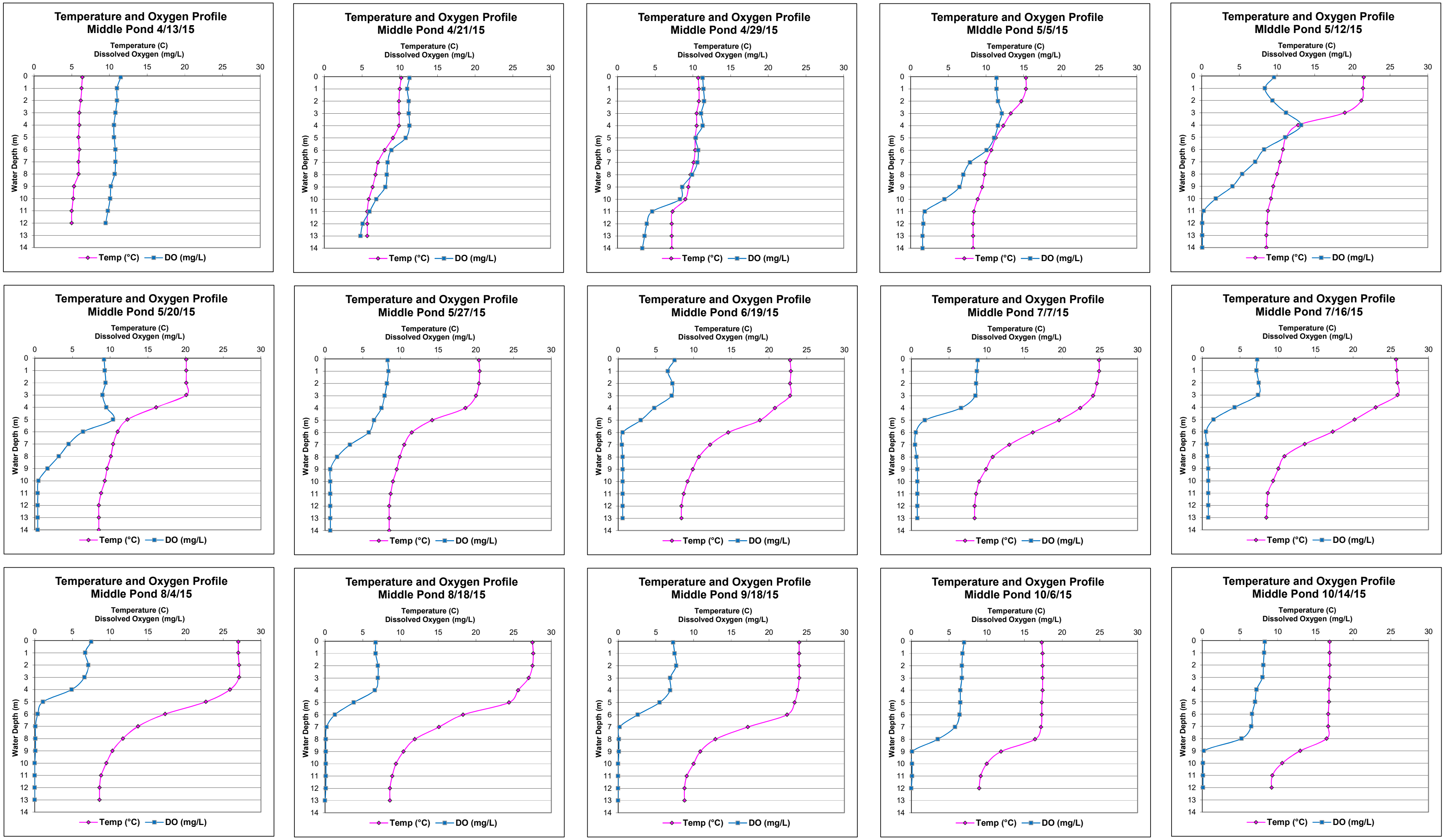



**Figure 10. Middle Pond Temperature-Oxygen Profiles for 2015**

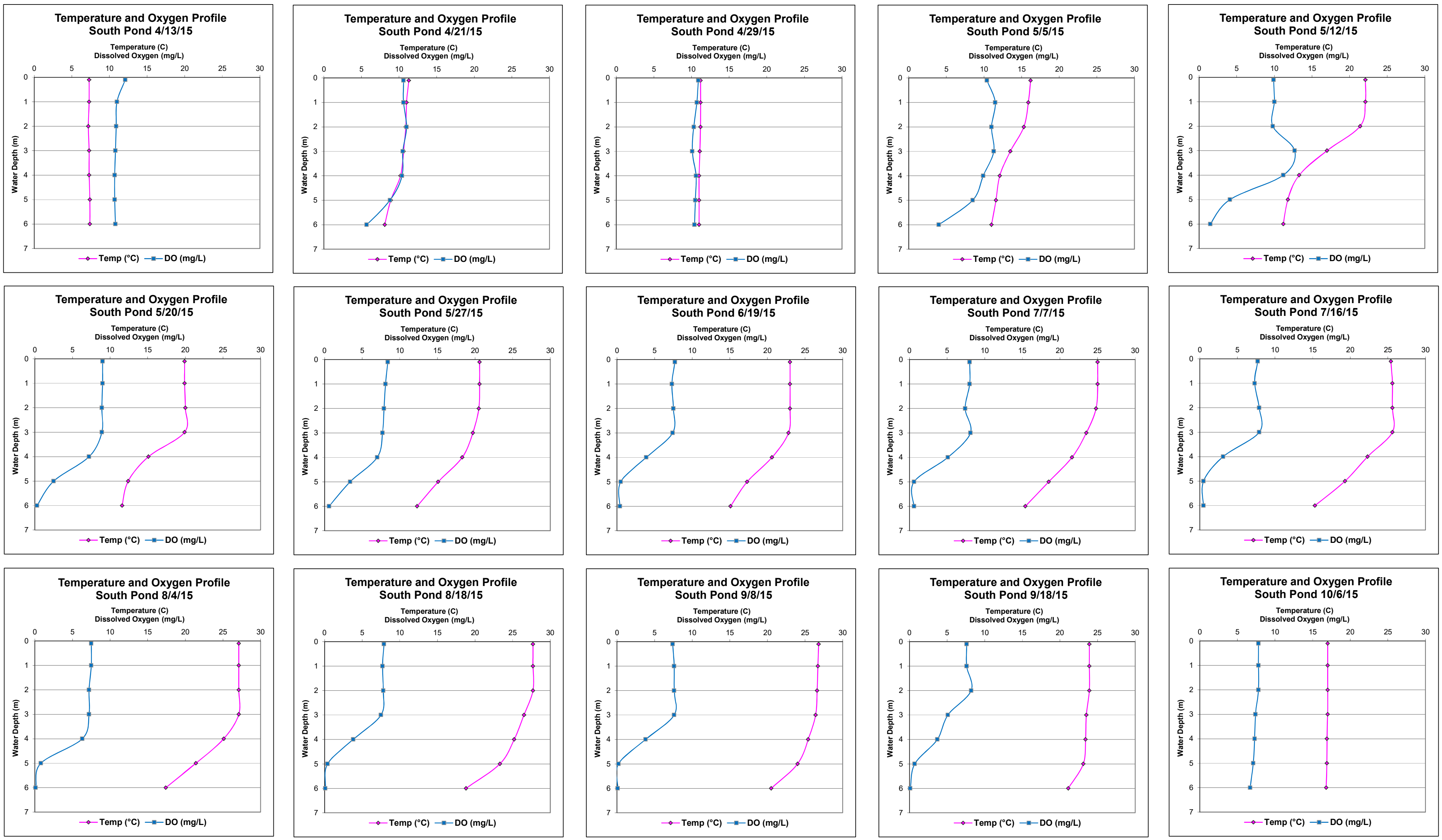



**Figure 11. South Pond Temperature-Oxygen Profiles for 2015**

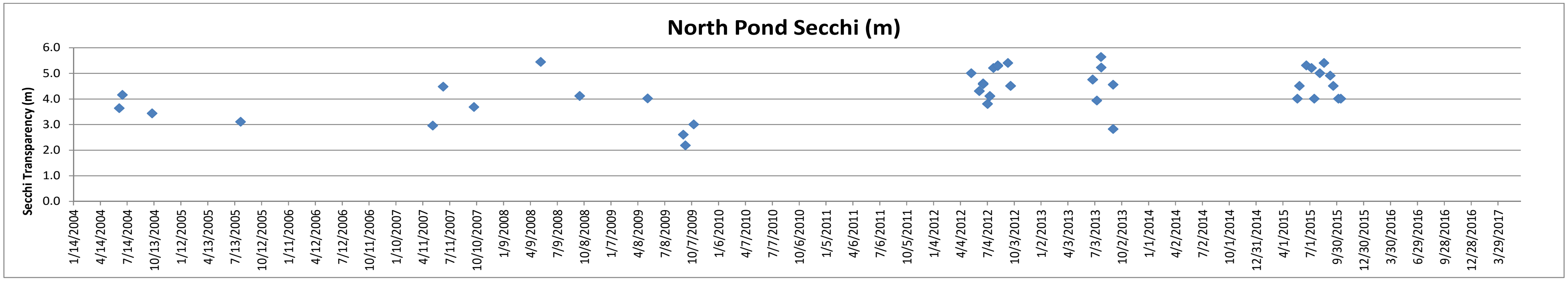
acre-feet (8.6% of total). Greater extent of anoxia may have occurred in other years, but the areal or volumetric percent of North Pond experiencing anoxia does not appear to be greater than 16-17% of the corresponding total. North Pond did exhibit an oxygen increase between 16 and 20 feet (5 and 6 m) in 2015, suggesting algae accumulation related to adequate light and maximum nutrient availability, as explained above. This will tend to lead to late summer or autumn algae blooms, when the thermal stratification breaks down and those algae mix into the upper waters; this is what was observed in North Pond in 2015.

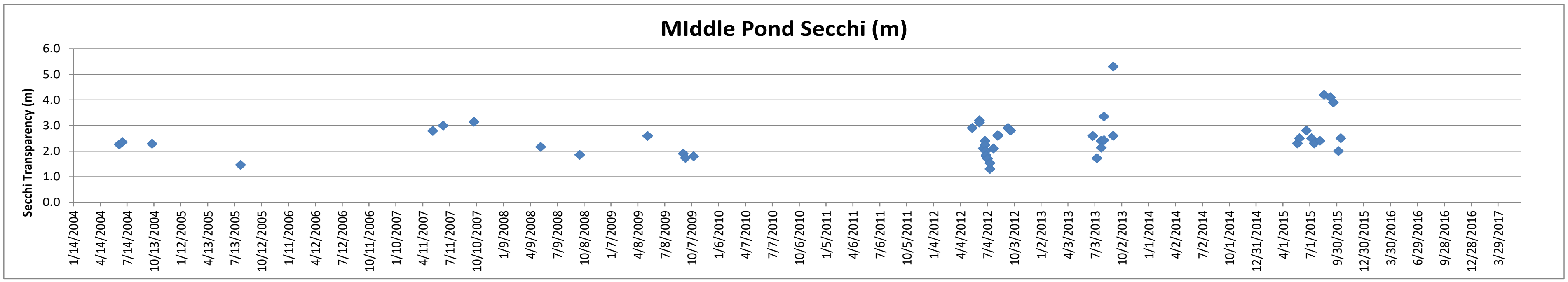
The profiles for Middle Pond (Figure 10) indicate that stratification sets up in May, with oxygen lost at the bottom in the deepest water in early May and anoxia extending to all areas deeper than 20 feet( 6 m) by the middle of June and persisting until early October. The portion of the pond area exposed to anoxia over summer is at least 101 acres or 36% of the total. For pond volume, 1033 acre-feet or almost 21% of the total area experiences anoxia. Anoxia may creep upward to 16.5 feet (5 m) in some years, representing almost 46% of the area and over 29% of the volume of Middle Pond.

South Pond has the lowest maximum depth and much of the pond is barely deep enough to have a hypolimnion form over summer. Strong winds could overcome the relatively weak stratification that develops, but the oxygen demand is high enough to cause low oxygen in the bottom 3-7 feet (1-2 m) of the pond anywhere deeper than about 16.5 feet (5 m) (Figure 11). Anoxia at the bottom occurs in the second half of May and dissipates quickly in late September. Anoxia did persist throughout summer of 2015, but could be disrupted in some years. At least 50 acres of area (34% of total) and 358 acre-feet of volume (17% of total) experience anoxia, and these could increase to 45% and 28%, respectively, in a "bad" year.

For both Middle and South Pond, a substantial amount of pond bottom area is conducive to algal growth with mass rises that cause blooms to appear at the surface fairly quickly. This is a common mode of early to mid-summer cyanobacteria blooms which have been observed in these ponds. Dying algae eventually settle, adding organic matter and nutrients to all areas, even shallow zones. While gradual movement into deeper water is expected, some cycling in shallow water will occur and my foster additional algae growth. In shallower water, even without anoxia, green algae mats may grow and create algae "clouds" in the water column or surface mats in some areas. The role of accumulated nutrients in the bottom sediments of the Congamond Lakes may be extremely important in the observed blooms. While this accumulation is largely a product of watershed inputs over many years, it is no longer dependent upon ongoing seasonal inputs to be sustained. Internal cycling of nutrients under the observed thermal and oxygen regimes of at least Middle and South Ponds is likely to be the dominant cause of algae blooms, and such cycling is likely important in North Pond as well.

Water clarity is largely dependent on algae levels in ponds such as these, and is often assessed as the depth of visibility of a Secchi disk. Secchi transparency is a simple but useful measure that relates to many important pond characteristics. Values <4 feet (1.22 m) are indicative of unsafe swimming conditions, while values >5 m are rare in summer in southern New England. For the Congamond Lakes since 2004, Secchi transparency values have been moderately stable and certainly not in decline, but are sometimes low in all three ponds and are often low in Middle and South Ponds (Figure 12). Data from prior to 2004 are too limited to draw conclusions, but values



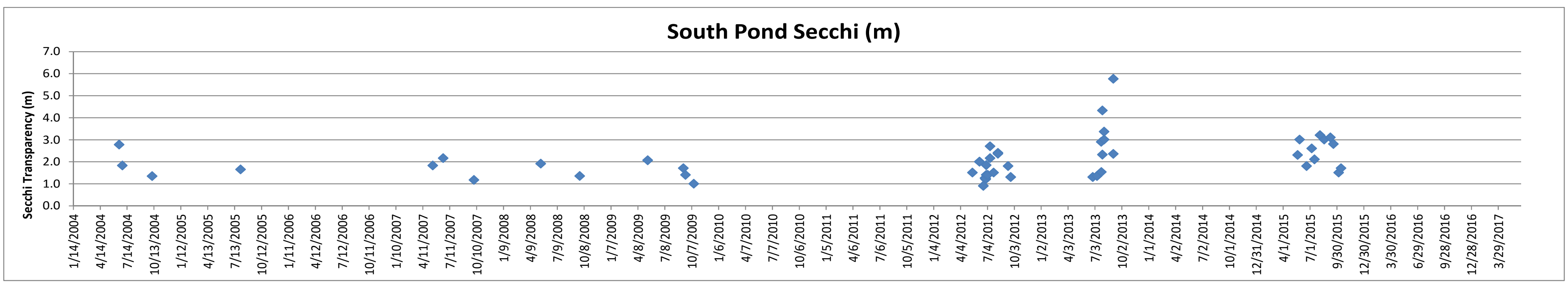




**Figure 12. Secchi Transparency in the Congamond Lakes from 2012-2015**

from the early 1980s were low and some improvement after the 1989 aluminum treatment would have been expected.

The average Secchi transparency values for the three ponds between May and October since 2012 are 15.2 feet (4.6 m) for North Pond, 8.3 feet (2.5 m) for Middle Pond, and 6.9 feet (2.1 m) for South Pond. These averages are acceptable for current pond uses, but it is not the average that creates issues for most lake users. Rather, it is the minimum value and the frequency of low values that cause concerns. All three ponds have experienced low values, Middle and South Ponds more often than North Pond, and cyanobacteria have been the cause of nearly all of those low values. While only a few Secchi transparency measures have been <7 feet (2.1 m) in North Pond in recent years, over 36% of Middle Pond clarity values and 66% of South Pond values have been below that threshold since 2012.

Conductivity and pH measurements are obtained with temperature and oxygen profiles. Conductivity is a measure of electrical potential and represents dissolved solids in the water. While it does not qualify the types of solids present, values <100 µmhos/cm are considered low and indicative of limited fertility, while values >300 µmhos/cm suggest higher solids content and values >500 µmhos/cm are uncommon in this geographic area without strong salt influence. Values for the Congamond Lakes (Figure 13) suggest similar levels and patterns in all three ponds. Values were between 100 and 125 µmhos/cm in early spring before any thermal stratification and exhibited a slight increase with increasing depth. In summer, with stratification, surface values were near 150 µmhos/cm and increased to >200 µmhos/cm in North and Middle Ponds below the thermocline. The increase was not so pronounced in South Pond, which has a thinner hypolimnion and may experience more vertical mixing with weaker stratification. The influence of release of substances from anoxic sediment is apparent from these data.

The pH is a measure of acid content, with lower pH value indicating higher hydrogen ion concentration and more acidity. Values for the Congamond Lakes (Figure 13) tend to track the temperature profiles over time. In the early spring the pH of the upper waters is between 7.6 and 8, a bit higher than they would be based on area geology if not for algal photosynthesis, which removed carbon dioxide and raises the pH. The pH declines with depth, as does algal photosynthesis as light becomes limiting, and values decline slightly but are still above neutral (pH=7.0, neither basic or acidic). As the water warms and stratifies in late spring and summer, and algae become more abundant in surface water, the pH rises near the top of each pond, exceeding 8 standard units by the end of summer. The pH of deeper water, without much algal photosynthesis but having considerable decomposition and acid production, declines to between 6.5 and 7 standard units (slightly acidic). The range of pH is not a major threat to fish or people, but does indicate a level of algae abundance that is undesirable, and the level of variability in pH over space and time is not ideal for lake ecology.

Phosphorus and nitrogen concentrations in the Congamond Lakes (Figures 14 and 15) are very important to lake ecology and human uses. These are the most important nutrients for algae and rooted plant growth and their concentrations determine productivity in each pond. Most rooted plants get most nutrients from the sediment, but water column nutrients affect some rooted plants, but are more important to algae. As noted previously, some algae can grow at the sediment-water interface and rely less on nutrients in the upper water column, but water column

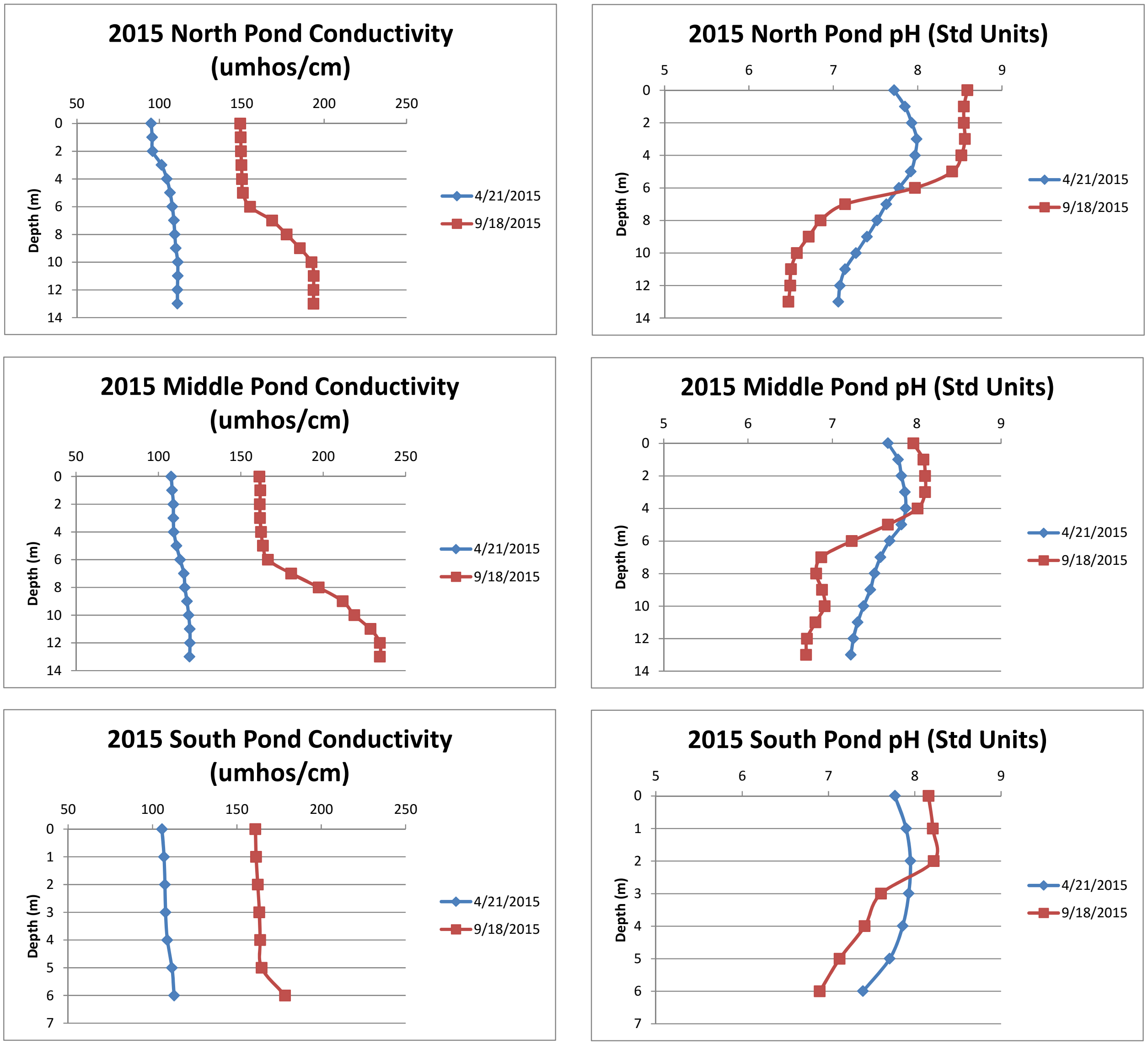



**Figure 13. Selected Conductivity and pH Profiles for the Congamond Lakes**

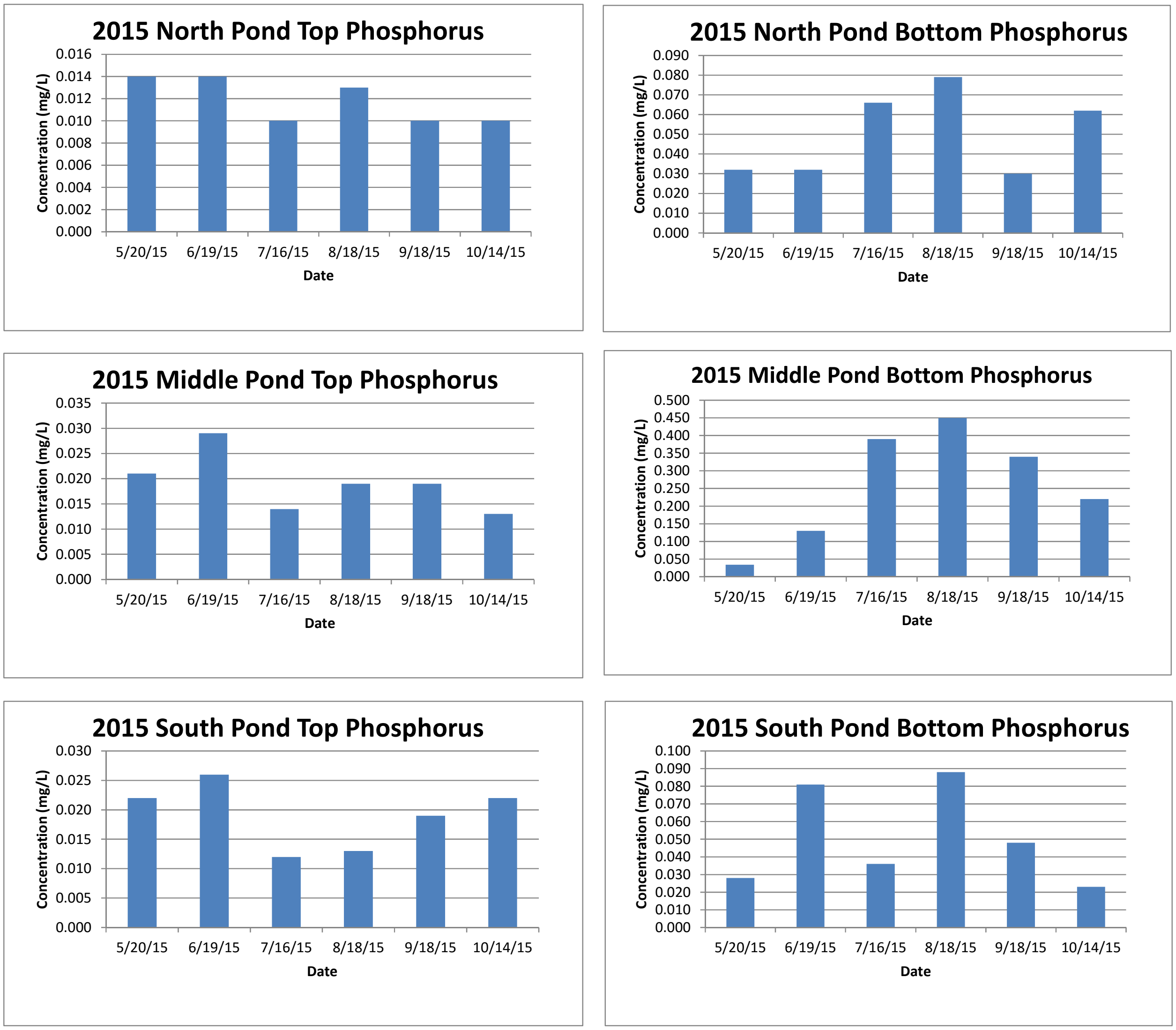



**Figure 14. Phosphorus in the Congamond Lakes in 2015**

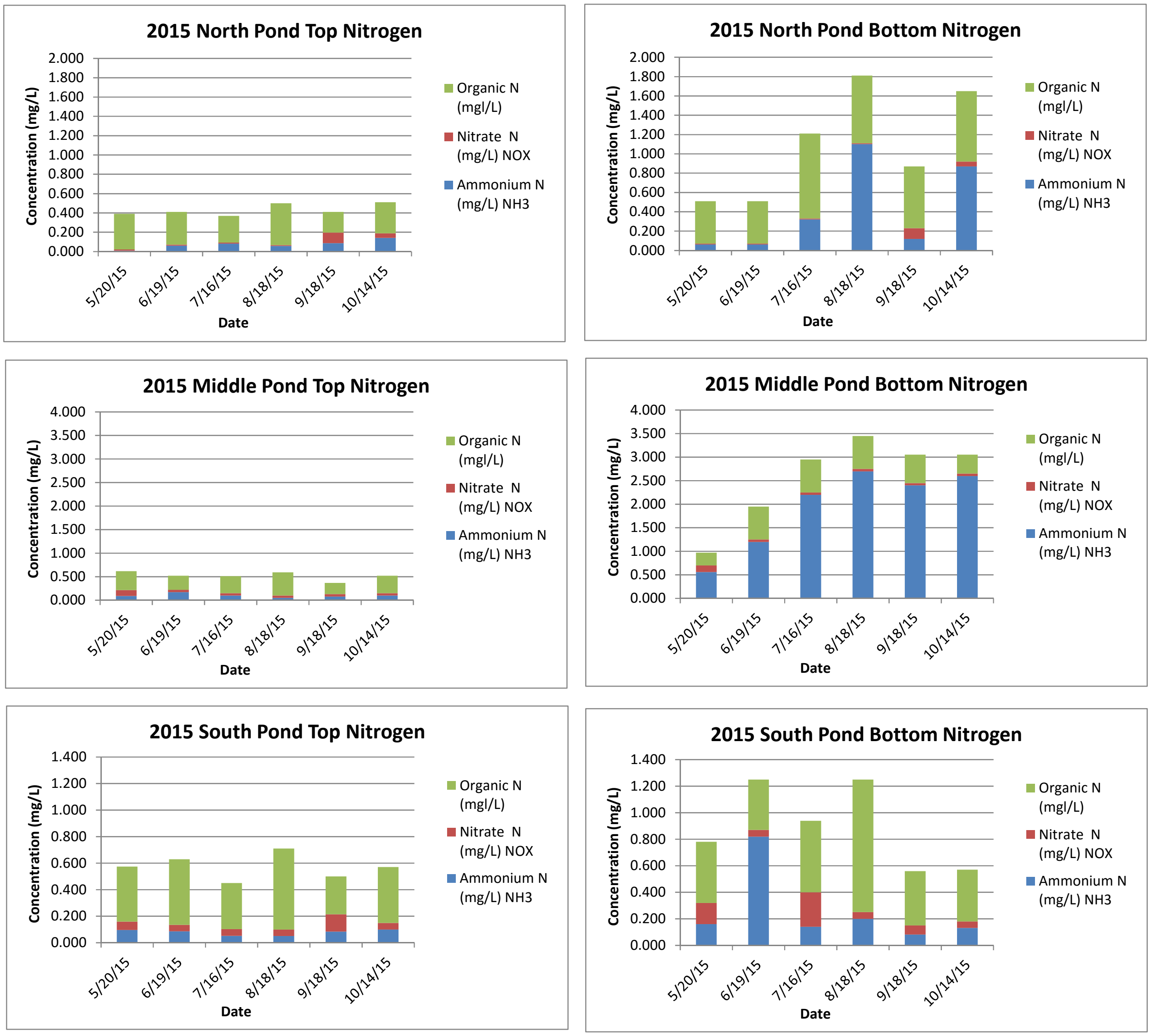



**Figure 15. Nitrogen in the Congamond Lakes in 2015**

concentrations remain important to the growth of many species and to the duration of algae blooms once formed. Phosphorus levels in excess of 10 µg/L can support blooms, but in most cases severe blooms are not observed at levels <20 µg/L. Nitrogen is less influential that phosphorus to the overall magnitude of algae in the water, but is critical to the types of algae that are found. Low N:P ratios tend to favor cyanobacteria, while elevated N:P ratios tend to favor green algae, but the form of nitrogen matters greatly and nutrient dynamics can be quite complicated.

Total and dissolved phosphorus were assessed, but dissolved phosphorus was below the detection limit of 10 µg/L in the vast majority of samples. This suggests that phosphorus is the limiting nutrient and that most phosphorus is found in particulate form, having been taken up by algae or otherwise incorporated into organic matter. Total phosphorus values in North Pond were between 10 and 14 µg/L in the surface water of North Pond and increased from about 30 to 80 µg/L in the bottom waters over the course of the summer. This suggests accumulation of phosphorus in the deep water, consistent with release from anoxic sediment, but it is somewhat unusual that more of the deep water phosphorus is not in dissolved form. Some of the deep water phosphorus undoubtedly mixed into surface waters in the fall, but with the hypolimnion being so small relative to total pond volume, no increase in surface phosphorus is observed. The more important implication is that algae growth is best supported near the thermocline during stratification.

Phosphorus levels in Middle Pond and South Pond averaged 19 µg/L in the surface water in each case, well above the desired 10 µg/L but not quite to the 20 µg/L threshold. However, the copper treatment in June may have sent considerable phosphorus to the sediment and slowed the summer increase. Deeper water phosphorus was much higher in Middle Pond, increasing from 34 to 450 µg/L from May into August and averaging 261 µg/L for the sampling period. Deeper water levels in South Pond were not as high, averaging 51 µg/L but exhibiting an oscillating pattern over time that suggests that phosphorus was either moving into upper waters or back into the sediment (likely both) with mixing events over the summer.

Total nitrogen concentrations were low to moderate in surface waters of all three ponds, but most nitrogen from the upper waters was in particulate organic form. Levels of nitrate and ammonium, the desired forms for uptake, were very low. This suggests that nitrogen limitation is possible for algae unable to use dissolved nitrogen gas, which is what gives certain types of cyanobacteria a competitive edge. TN:TP ratios were almost all >20:1 for samples from the epilimnion of the ponds, suggesting phosphorus limitation overall, but the low availability of nitrates and ammonium will affect which algae dominate.

Total nitrogen in the hypolimnion of each pond was much higher, with substantial accumulation of ammonium in North Pond and especially Middle Pond. Ammonium was less abundant in the deeper water of South Pond, but nitrates were more abundant than in the other two ponds, suggesting that oxygen input to deeper water in South Pond allowed more conversion of ammonium to nitrate. Deeper water ratios were lower in all three ponds than in the surface water, but were <10:1 (strongly favoring cyanobacteria) only in Middle Pond in July and August. Both phosphorus and nitrogen levels were higher in the deeper water, but the increase in phosphorus was proportionately greater.

Consideration of some of the water quality relationships (Figures 16-19) can be helpful in visualizing linkages and sometimes with streamlining monitoring. Turbidity is easy to measure with a one-time instrument purchase, and can sometimes act as an inexpensive surrogate parameter. However, neither phosphorus nor nitrogen exhibited a strong relationship with turbidity in the Congamond Lakes in 2015. Turbidity and Secchi transparency were strongly linked, and turbidity also correlated with chlorophyll-a as assessed by fluorescence. The actual chlorophyll-a values are suspect, as there has been no calibration of the fluorometric sensor, but the relative magnitude of values is reliable and helpful. The linkage between turbidity, Secchi and chlorophyll-a is as expected and suggests that any one of these measures can provide an indication of the level of the other two.

Actual algae measurement was restricted largely to monthly samples, which exhibited a transition from spring through summer into fall within the three ponds (Figure 20). It is unfortunate that the June samples were misplaced, as this was a pivotal month for at least Middle and South Ponds, but qualitative data were collected. A bloom of the cyanobacterium Dolichospermum (aka Anabaena) was detected early as a consequence of volunteer monitoring and electronic photograph conveyance to WRS. Conditions were deteriorating quickly, but rapid response by an on-call treatment contractor, ACT (now part of Solitude Lake Management), averted serious problems. Copper application to just under half of the combined area of Middle and South Ponds reversed the trend and maintained acceptable water clarity through the summer.

The timing and nature of the bloom was indicative of growth at the sediment-water interface with mass rise to the surface in response to changing temperature and light regime. The treatment was not in the deepest water, but more peripheral, where the combination of light and nutrients at the sediment-water interface would be most advantageous for algae growth. The treatment killed the cyanobacteria that were already in the water column, but likely also impacted remaining growths still at the bottom, averting further blooms. This is a strategy applied by many water suppliers for reservoirs prone to such blooms; treating slightly in advance of a major bloom not only minimizes poor water quality associated with the bloom, but also can provide benefits of longer duration than waiting for the bloom to reach its peak.

Phytoplankton levels were below any thresholds of concern for the remainder of the summer. Blooms were observed in fall, including pulses of the golden algae Dinobryon in Middle and South Ponds in October, the cyanobacteria Dolichospermum in North Pond in November, and the cyanobacterium Aphanizomenon in Middle and South Ponds in November. These later blooms are undoubtedly linked to the breakdown of thermal stratification and greater availability of the phosphorus that was present and largely sequestered in deeper water before then.

Zooplankton were assessed early and late in the sampling season, just to check on the nature of the community at times of expected peak abundance (spring) and minimum biomass (early fall). Zooplankton biomass (Figure 21) and mean body length (Figure 22) are useful measures that indicate the level of grazing on algae by zooplankton that can be expected. Biomass was low in North Pond in May 2015, but was substantial in Middle and South Ponds at that time. Biomass in all three ponds was low by fall, an indication of strong predation by small fish. A pattern of high spring abundance and low fall abundance is not unusual and suggests successful fish reproduction, but it does limit summer grazing on phytoplankton and that avenue of control.

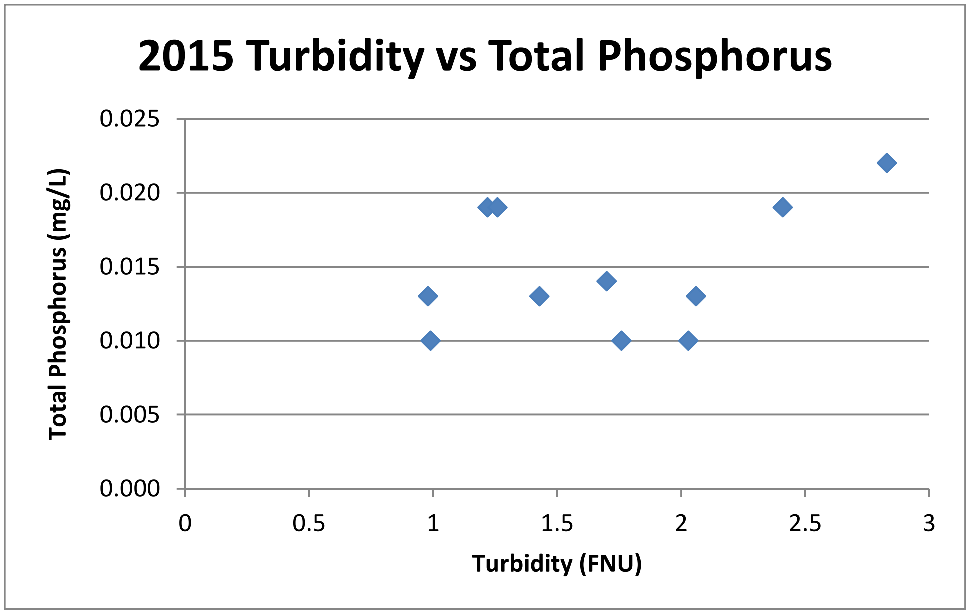



**Figure 16. Turbidity vs. Phosphorus in the Congamond Lakes**

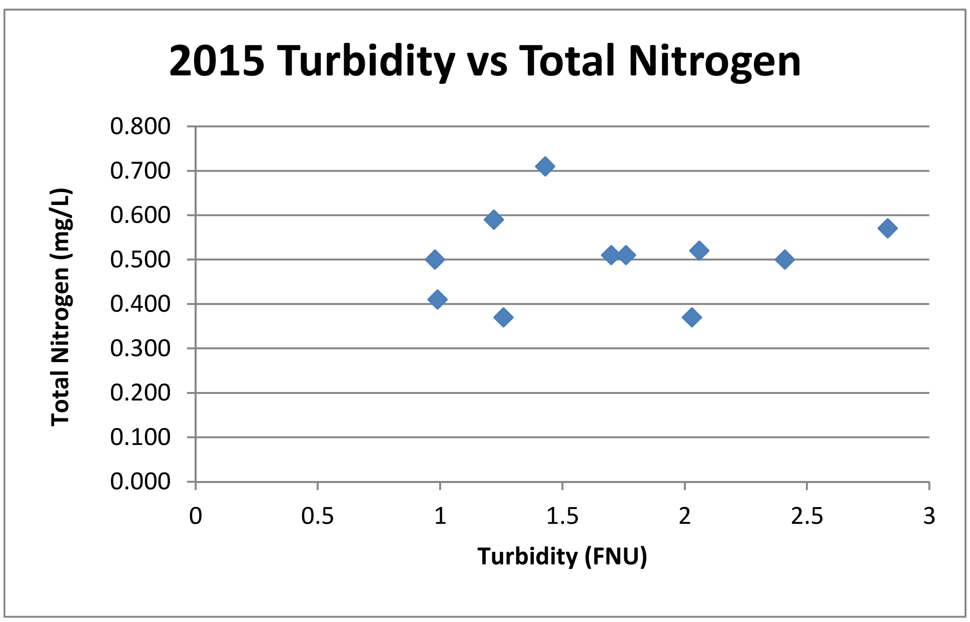



**Figure 17. Turbidity vs. Nitrogen in the Congamond Lakes**

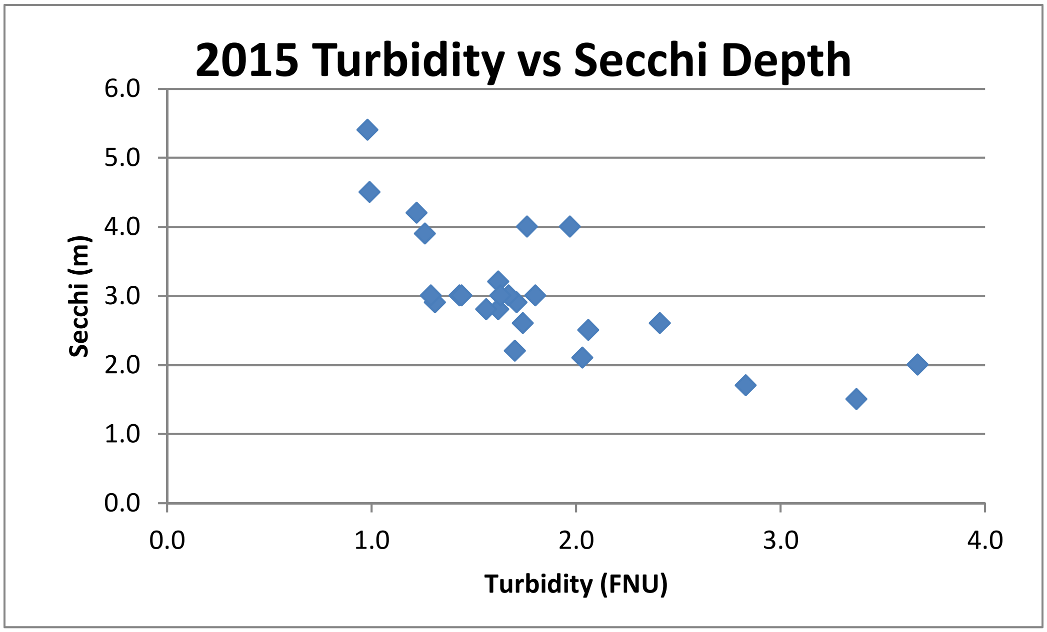



**Figure 18. Turbidity vs. Secchi Transparency in the Congamond Lakes**

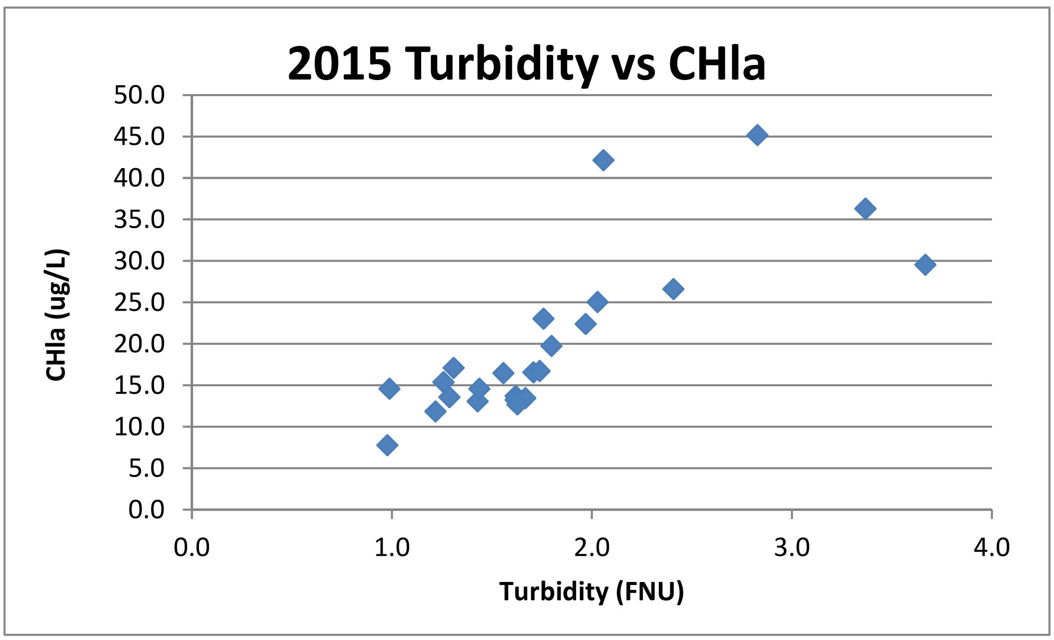



**Figure 19. Turbidity vs. Chlorophyll-a in the Congamond Lakes**

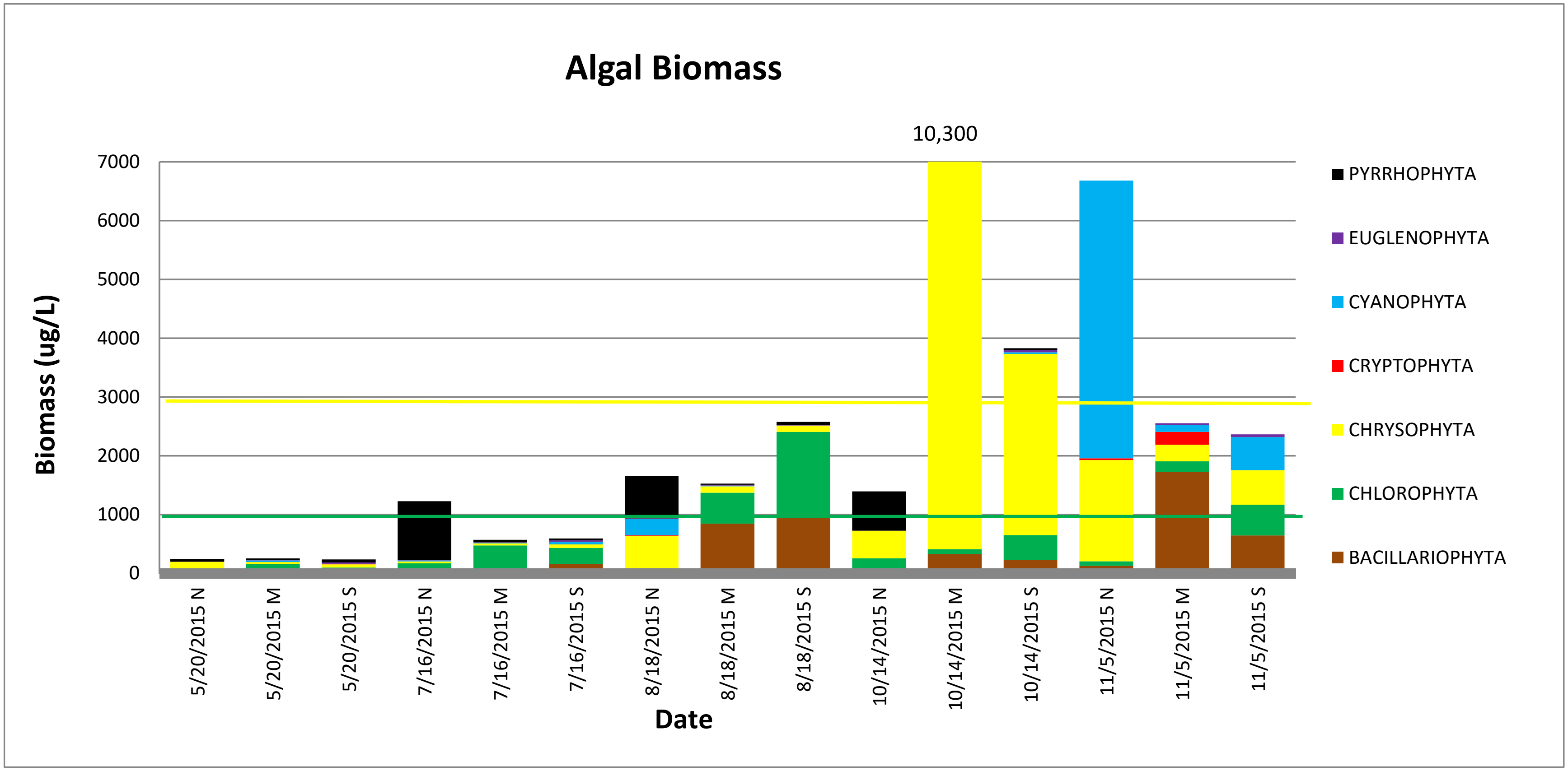



**Figure 20. Algae Biomass in the Congamond Lakes**

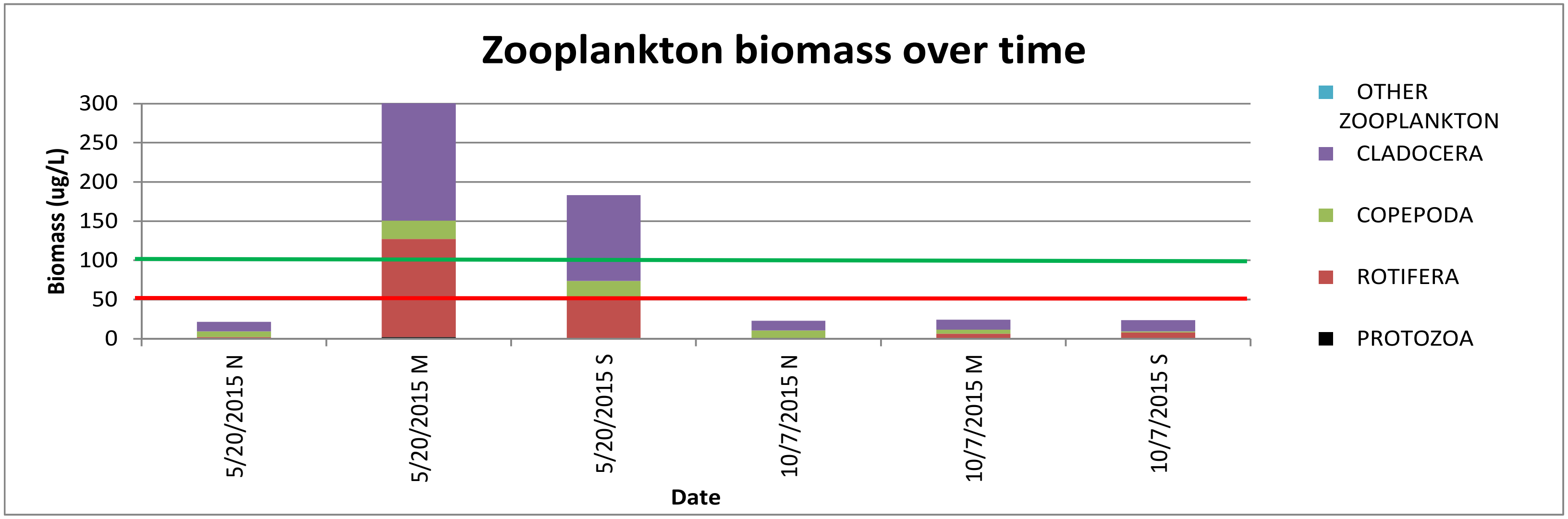



**Figure 21. Zooplankton Biomass in the Congamond Lakes**

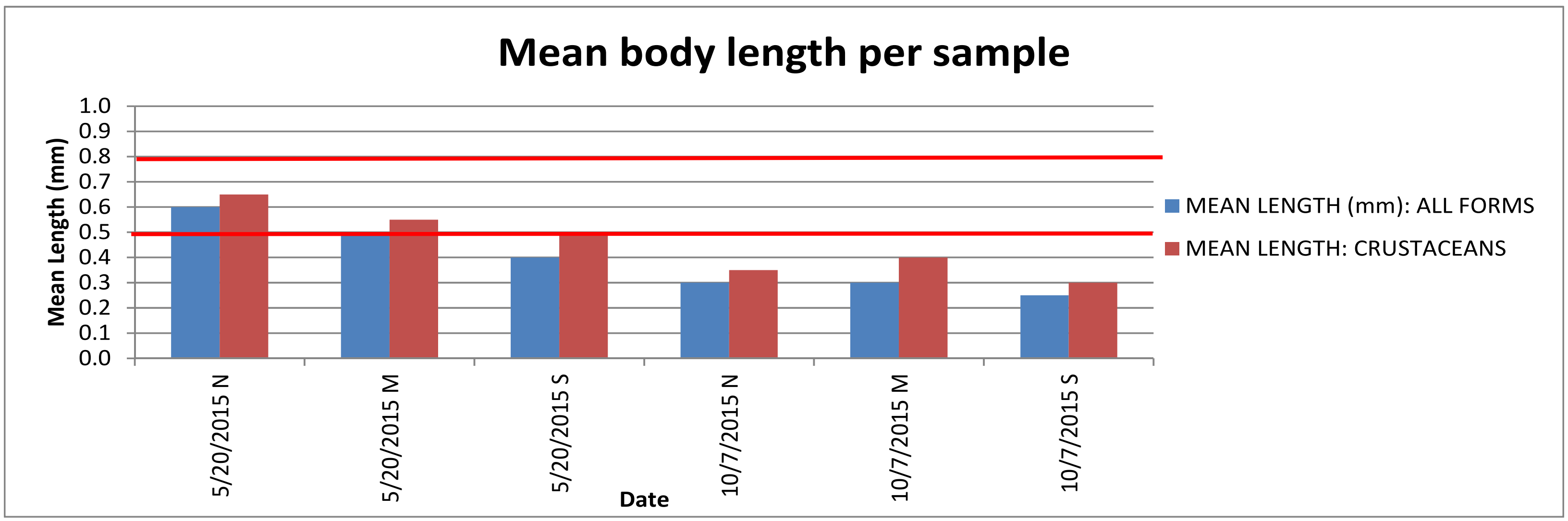



**Figure 22. Zooplankton Mean Length in the Congamond Lakes**

Mean zooplankter length was in a desirable zone for all three ponds in May, but was much reduced in fall. This means that plenty of larger bodied forms were present in spring, but that selective predation reduced mean size over the summer. The actual composition suggested only a limited population of Daphnia, a large bodied cladoceran important as both an algae grazer and food source for small fish; Daphnia are not abundant at any time and are eliminated by predation over summer. Most forms present were very small and do not indicate either much grazing pressure potential or a high energy food source for fish. Zooplankton are an important link between algae and fish, and a poorly structured zooplankton community has negative implications for both water clarity and fishing.

## Storm Water Quality

Storm water samples were collected with passive samplers capturing the first flush of storm water from up to 8 small drainage areas during 3 storms. Such samples typically represent the worst water quality during a storm event, but where there is no flow before the storm and little flow afterward, this is the only available estimate of inputs derived from storm water. The results (Figures 23 and 24, Table 3) suggest fairly typical storm water runoff from residential areas. Phosphorus and nitrogen are much elevated over what would be expected in background dry weather flows from streams or from forested drainage areas, but most of the nutrient load is in particulate form and will settle rapidly. Most of these nutrients will not be readily available to fuel algae growth, but become part of the longer term internal load. In just a few samples the dissolved phosphorus component was dominant, but the relatively small flows with which these concentrations are associated will limit the overall load to the ponds. The ammonium and nitrate fractions of total nitrogen were almost always low and the associated dissolved nitrogen load to the ponds appears relatively minor.

The poor water quality of storm water runoff is a concern, but not for what any individual storm does to short term pond water quality. The volumes of runoff are too small relative to the volumes of the ponds to greatly alter water quality during any one storm event, but over the long run these inputs have led to the potentially high internal load or phosphorus and recycling of nitrogen within the ponds.

Storm water is generally known for high variability in content, so sampling of a limited number of small drainage basins during 3 storm events in just part of one year is not enough to thoroughly characterize inputs, but the consistency of average values among stations and ponds is fairly high. Certainly there is variability, but over the potential range of storm water values, the data for storm inputs to Congamond Lakes is very close to the average from nationwide studies of such runoff. Grand mean total phosphorus was 347 µg/L, while the national average for residential lands is between 300 and 400 µg/L. The grand mean for total nitrogen of nearly 2 mg/L is also close to the expected average. The relationship between phosphorus and nitrogen is fairly strong, with these two nutrients co-varying over the range encountered (Figure 25). As with the in-lake data, the relationship between storm water turbidity and either phosphorus or nitrogen was not strong (Figures 26 and 27); sometimes the storm water linkage is quite strong, allowing assessment of likely nutrient levels from more economically obtained turbidity data, but not in this case.

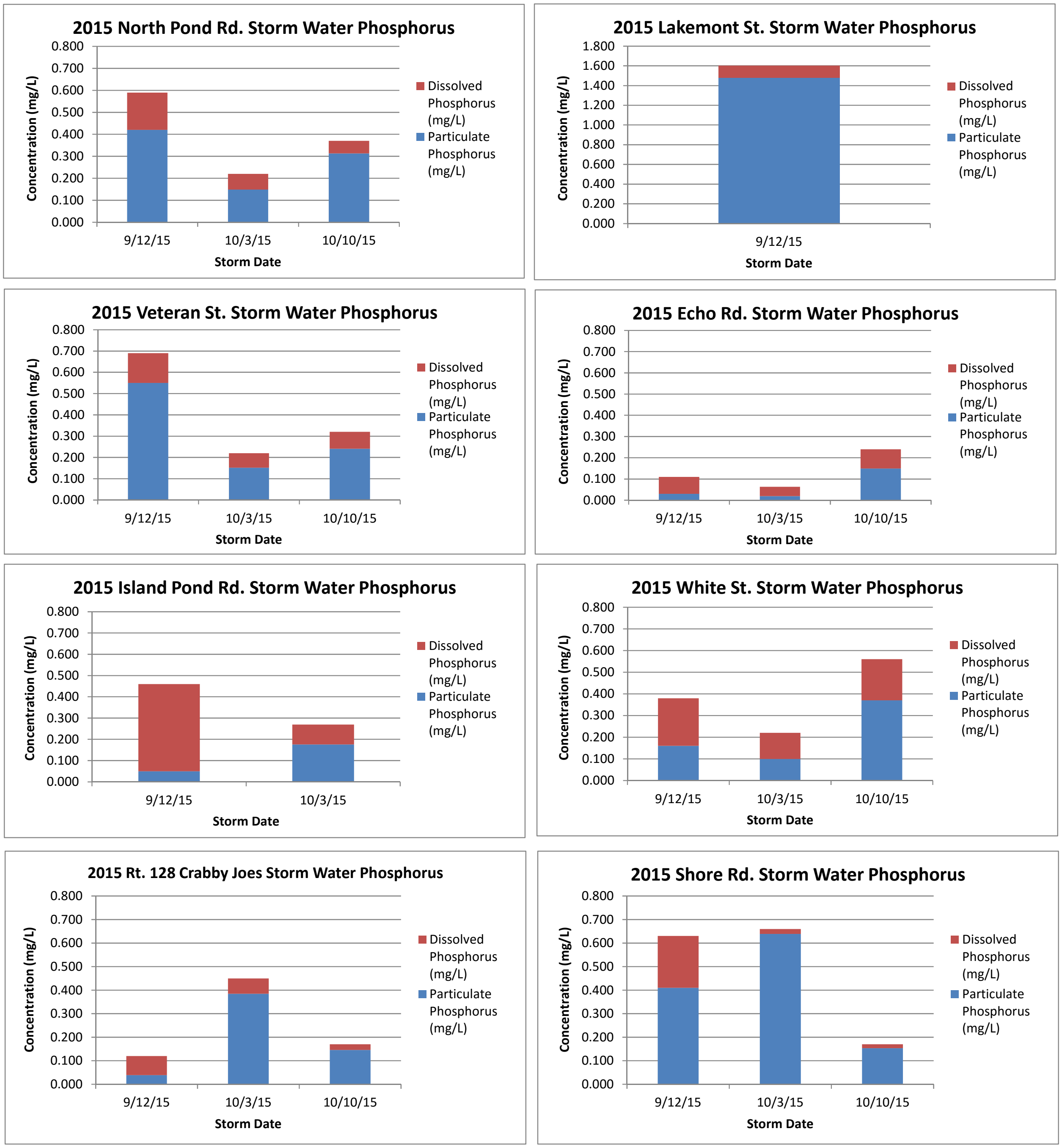



**Figure 23. Storm Water Phosphorus in the Congamond Lakes Watershed in 2015**


**Figure 24. Storm Water Nitrogen in the Congamond Lakes Watershed in 2015**

## Table 3. Summary of Storm Water Data for the Congamond Lakes in 2015

| Station ID | Goes To | Station Location | Ammonium Nitrogen | Nitrate Nitrogen | TKN | Total Nitrogen | Total Phosphorus | Dissolved Phosphorus | Turbidity |
|---|---|---|---|---|---|---|---|---|---|
| | | | mg/L | mg/L | mg/L | mg/L | mg/L | mg/L | FNU |
| CLSW1 | North | North Pond Rd | 0.27 | 0.16 | 1.77 | 1.92 | 0.393 | 0.099 | 40.1 |
| CLSW2 | North | Lakemont Street | 0.70 | 0.29 | 6.90 | 7.19 | 1.600 | 0.120 | 11.4 |
| CLSW3 | North | Veteran Street | 0.30 | 0.06 | 2.03 | 2.09 | 0.410 | 0.096 | 20.2 |
| CLSW4 | Middle | Echo Road | 0.12 | 0.06 | 0.60 | 0.66 | 0.138 | 0.072 | 10.3 |
| CLSW5 | Middle | Island Pond Raod | 0.25 | 0.19 | 1.20 | 1.39 | 0.365 | 0.252 | 13.0 |
| CLSW6 | Middle | White Street | 0.20 | 0.12 | 1.47 | 1.58 | 0.387 | 0.177 | 52.0 |
| CLSW7 | Middle | Rt 168 Crabby Joes | 0.43 | 0.06 | 2.16 | 2.22 | 0.247 | 0.057 | 57.1 |
| CLSW8 | South | Shore Road | 0.52 | 0.06 | 3.87 | 3.93 | 0.487 | 0.086 | 47.3 |
| | | North Pond Mean | 0.29 | 0.11 | 1.90 | 2.01 | 0.402 | 0.098 | 30.1 |
| | | Middle Pond Mean | 0.25 | 0.11 | 1.36 | 1.46 | 0.284 | 0.139 | 33.1 |
| | | South Pond Mean | 0.52 | 0.06 | 3.87 | 3.93 | 0.487 | 0.086 | 47.3 |
| | | Grand Mean | 0.30 | 0.10 | 1.87 | 1.97 | 0.347 | 0.120 | 34.3 |
| | | | | | | | | | |
| Note: Lakemont St. off North Pond sampled only once while construction was ongoing; data not included in mean values. | | | | | | | | | |

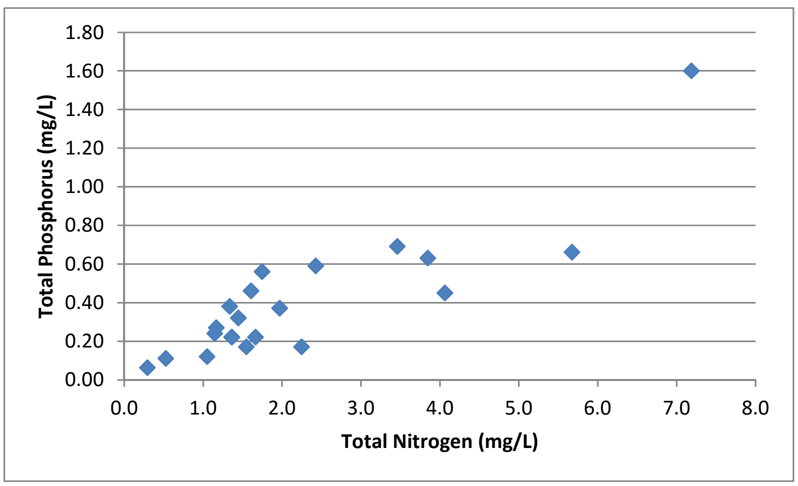



Figure 25. Phosphorus vs Nitrogen in Congamond Lakes Storm Water

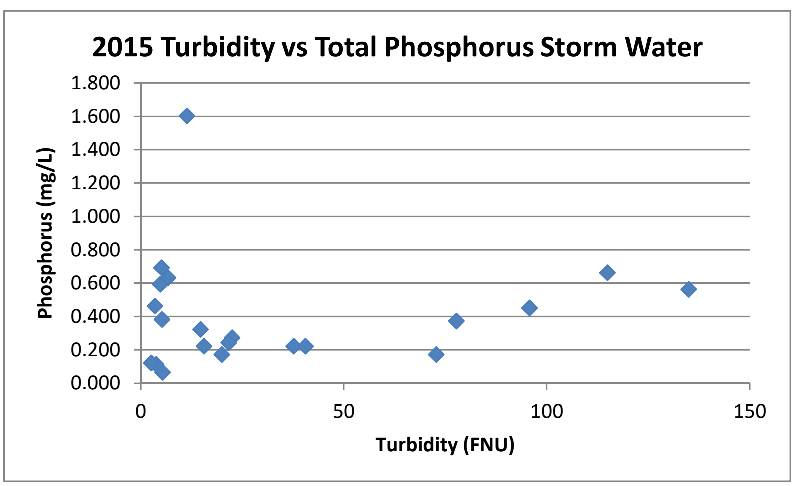



Figure 26. Turbidity vs. Phosphorus in Congamond Lakes Storm Water

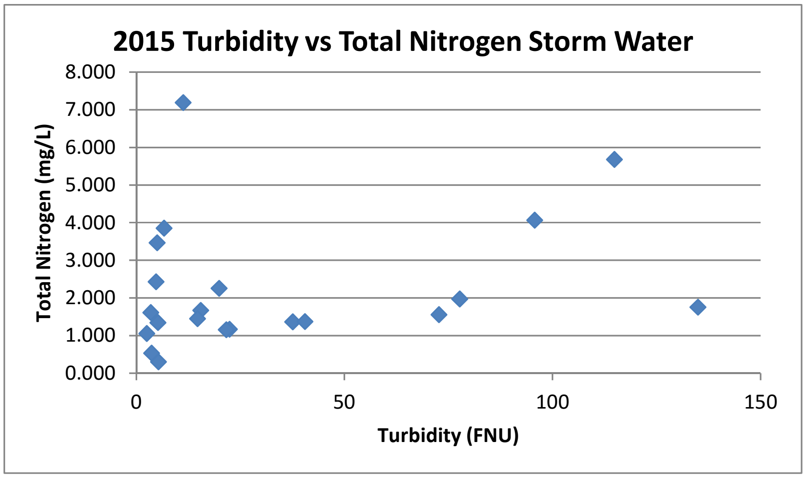



**Figure 27. Turbidity vs. Nitrogen in Congamond Lakes Storm Water**

## Sediment Quality

Given the potential for release of iron-bound phosphorus to be a substantial source in the Congamond Lakes, surficial sediment samples were collected from each pond and tested (Table 4). Solids content was low but typical for highly hydrated pond muck except at North Pond station NS3, which was in a sandier substrate. Total phosphorus in the sediment was elevated at all but NS3, but much of this phosphorus is not available for uptake and is not released under anoxic conditions. The iron-bound phosphorus fraction is the most readily available, although even all of this portion is not quickly released when oxygen is depleted. There is a general but not especially strong correlation between total and iron-bound phosphorus, and specific testing in each waterbody reveals variability.

In general, iron-bound phosphorus <50 mg/kg does not represent a threat of substantial internal loading. Values in excess of 100 mg/kg become a possible concern, and values in excess of 200 mg/kg can generate a substantial load. Values >500 mg/kg are a major threat if anoxia occurs, and values >1000 mg/kg are not especially common in New England but can have great impact on water quality. Values in the Congamond Lakes are of the same order of magnitude, distinctly high but not extreme, except again for North Pond station NS3, which is of a different substrate composition. Excluding data from NS3, the mean values for iron-bound phosphorus in the three ponds range from 372 mg/kg to 472 mg/kg. For waterbodies of the area of Middle and South Ponds, more samples might be appropriate to characterize variation, but the obtained values do allow a rational analysis of potential internal loading and approximation of benefits to be gained by curtailing that loading.

Two additional core samples were obtained from the canal leaving South Pond and subjected to assessment of a reduced suite of parameters applicable to dredging feasibility evaluation. The chosen tests represent the sediment features most likely to cause problems with disposal due to quality in relation to environmental regulations. The core samples were homogenized to represent the likely depth of dredging before testing.

In this case, the sediment was remarkably clean (Table 5), not exceeding any standard or regulatory threshold for Massachusetts. The sediment has high organic content based on organic carbon, but the solids content was also high, suggesting easier handling. While additional testing would be needed to support a full dredging feasibility analysis, the sediment appears clean enough for disposal in any upland setting. Use on an area farm field would be most advantageous if follow up testing indicated the high quality observed in this analysis.

**Table 4. Summary of Sediment Phosphorus Data for the Congamond Lakes in 2015**

| Sample ID # | % Total Solids | Total Phosphorus (mg/kg) | Iron-Bound Phosphorus (mg/kg) |
|---|---|---|---|
| NS1 | 10.6 | 1530 | 307 |
| NS2 | 9.2 | 1930 | 436 |
| NS3 | 31.6 | 573 | 81 |
| MS1 | 8.5 | 935 | 303 |
| MS2 | 5.3 | 1340 | 530 |
| MS3 | 6.0 | 1190 | 619 |
| MS4 | 7.7 | 1090 | 366 |
| SS1 | 7.8 | 1100 | 378 |
| SS2 | 4.8 | 1390 | 581 |
| SS3 | 6.4 | 910 | 458 |

# Table 5. Canal Sediment Screening Data from 2015

| Parameter | Method | Units | Background Soil Data Set 90th Percentile | MCP RCS1 | BUD S-1, GW-1 | Freshwater Sediment Screening Value | BUD S-2, GW-1 | Canal Sed 1 | Canal Sed 2 |
|---|---|---|---|---|---|---|---|---|---|
| **Total Metals** | | | | | | | | | |
| Copper | 6010B, SW-846 | mg/kg | 37.7-47.5 | 1000 | | 150 | | 4.81 | 3.73 |
| Lead | 6010B, SW-846 | mg/kg | 78.9-640 | 300 | 19 | 130 | 110 | 6.09 | 5.88 |
| Nickel | 6010B, SW-846 | mg/kg | 16.6-67.5 | 300 | 7.2 | 49 | 350 | 3.87 | 4.22 |
| Zinc | 6010, EPA 1987 | mg/kg | 103-340 | 2500 | 20 | 420 | 3000 | 18 | 15 |
| **Extractable Petroleum Hydrocarbons** | | | | | | | | | |
| C9-C18 Aliphatics | EPH | mg/kg | | 1000 | 780 | | | <26.2 | <19.9 |
| C19-C36 Aliphatics | EPH | mg/kg | | 2500 | 3000 | | | <26.2 | <19.9 |
| C11-C22 Aromatics | EPH | mg/kg | | 200 | 48 | | | <26.2 | <19.9 |
| **Polynuclear Aromatic Hyrdocarbons** | | | | | | | | | |
| Acenaphthene | EPA 8270 | mg/kg | 1.9 | 20 | 3.9 | | | <0.871 | <0.661 |
| Acenaphthylene | EPA 8270 | mg/kg | 1 | 100 | 1.1 | | | <0.871 | <0.661 |
| Anthracene | EPA 8270 | mg/kg | 3.8 | 1000 | 1000 | | | <0.871 | <0.661 |
| Benzo(a)anthracene | EPA 8270 | mg/kg | 2.39-17.6 | 0.7 | 3.7 | 110 | | <0.871 | <0.661 |
| Benzo(a)pyrene | EPA 8270 | mg/kg | 2.02-15.3 | 0.7 | | | | <0.871 | <0.661 |
| Benzo(b)fluoranthene | EPA 8270 | mg/kg | 6.78-11.0 | 0.7 | 3.7 | | | <0.871 | <0.661 |
| Benzo(k)fluoranthene | EPA 8270 | mg/kg | 3.35-11.4 | 7 | 37 | | | <0.871 | <0.661 |
| Benzo(g,h,i)perylene | EPA 8270 | mg/kg | 1.2-3.1 | 1000 | | | | <0.871 | <0.661 |
| Chrysene | EPA 8270 | mg/kg | 2.1-20.3 | 7 | 370 | 170 | | <0.871 | <0.661 |
| Dibenzo(a,h)anthracene | EPA 8270 | mg/kg | 0.49-1.1 | 0.7 | 0.66 | 33 | | <0.871 | <0.661 |
| Fluoranthene | EPA 8270 | mg/kg | 4.2-14.0 | 1000 | 1000 | 420 | | <0.871 | <0.661 |
| Fluorene | EPA 8270 | mg/kg | 2.3 | 400 | 1000 | 77 | | <0.871 | <0.661 |
| Indeno(1,2,3-cd)pyrene | EPA 8270 | mg/kg | 1.5-6.3 | 0.7 | 3.7 | | | <0.871 | <0.661 |
| 1-Methylnaphthalene | EPA 8270 | mg/kg | 0.96 | | | | | <0.871 | <0.661 |
| 2-Methylnaphthalene | EPA 8270 | mg/kg | 0.96 | | | | | <0.871 | <0.661 |
| Naphthalene | EPA 8270 | mg/kg | 1.4 | 4 | 0.66 | 180 | | <0.871 | <0.661 |
| Phenanthrene | EPA 8270 | mg/kg | 2.7-15.0 | 100 | 10 | 200 | | <0.871 | <0.661 |
| Pyrene | EPA 8270 | mg/kg | 4.29-16.0 | 700 | 1000 | 200 | | <0.871 | <0.661 |
| **Pesticides** | | | | | | | | | |
| alachlor | EPA 8081 | ug/kg | | | | | | <13.3 | <10.2 |
| aldrin | EPA 8081 | ug/kg | | 30 | 22 | | | <13.3 | <10.2 |
| alpha-BHC | EPA 8081 | ug/kg | | 50,000 | | | | <13.3 | <10.2 |
| beta-BHC | EPA 8081 | ug/kg | | 10,000 | | | | <13.3 | <10.2 |
| delta-BHC | EPA 8081 | ug/kg | | 10,000 | | | | <13.3 | <10.2 |
| gamma-BHC (Lindane) | EPA 8081 | ug/kg | | 100 | | 2400 | | <7.96 | <6.14 |
| alpha-chlordane | EPA 8081 | ug/kg | | 1,000 | 70 | 3,200 | | <13.3 | <10.2 |
| gamma-chlordane | EPA 8081 | ug/kg | | 1,000 | 70 | 3,200 | | <13.3 | <10.2 |
| chlordane | EPA 8081 | ug/kg | | 1,000 | 70 | 3,200 | | <53.1 | <40.9 |
| 4,4'-DDD | EPA 8081 | ug/kg | | 2,000 | 1,800 | 4,900 | | <21.2 | <16.4 |
| 4,4'-DDE | EPA 8081 | ug/kg | | 2,000 | 1,300 | 3,200 | | <13.3 | <10.2 |
| 4,4'-DDT | EPA 8081 | ug/kg | | 2,000 | 1,300 | 4,200 | | <21.2 | <16.4 |
| dieldrin | EPA 8081 | ug/kg | | 30 | 23 | 1900 | | <13.3 | <10.2 |
| endosulfan I | EPA 8081 | ug/kg | | 50 | 36 | | | <13.3 | <10.2 |
| endosulfan II | EPA 8081 | ug/kg | | 50 | | | | <21.2 | <16.4 |
| endosulfan sulfate | EPA 8081 | ug/kg | | 50 | | | | <21.2 | <16.4 |
| endrin | EPA 8081 | ug/kg | | 600 | 3900 | 2200 | | <21.2 | <16.4 |
| endrin ketone | EPA 8081 | ug/kg | | 600 | | | | <21.2 | <16.4 |
| endrin aldehyde | EPA 8081 | ug/kg | | 600 | | | | <21.2 | <16.4 |
| heptachlor | EPA 8081 | ug/kg | | 100 | 96 | | | <13.3 | <10.2 |
| heptachlor epoxide | EPA 8081 | ug/kg | | 60 | 56 | 2500 | | <13.3 | <10.2 |
| methoxychlor | EPA 8081 | ug/kg | | 30,000 | 76 | | | <21.2 | <16.4 |
| toxaphene | EPA 8081 | ug/kg | | 10,000 | | | | <265 | <205 |
| **Total Organic Carbon** | EPA 415-1 | | | | | | | 49000 | 34000 |
| **Total Solids** | 2540B SM | % | | | | | | 37.5 | 48.4 |

## Oxygen Demand Assessment

Loss of oxygen in deep water has major impacts on lake ecology and uses. Low oxygen restricts habitat for fish and invertebrates, and complete loss of oxygen (anoxia) fosters chemical and biochemical reactions that either release certain contaminants from the surficial sediment or prevent processing of contaminants that enter the deep water from upper waters. Decomposition slows, ammonium cannot be converted to nitrate, and phosphorus can be released from sediment as it dissociates from iron compounds which often represent a substantial fraction of the total phosphorus in those sediments. Anoxia is not uncommon in lakes that stratify during summer, as the movement of oxygen downward across the boundary between upper and lower water layers (the thermocline) is very slow and decomposition and related bacterial metabolism consumes the available oxygen faster than it can be replaced. However, elevated loading of organic matter and nutrients from the watershed accelerates this process, causing faster and more severe anoxia over a greater area than might otherwise occur.

The data from spring of 2015 were collected at intervals intended to support calculation of oxygen demand. Spring data are best for this purpose, as oxygen uptake is hindered at low oxygen levels; oxygen demand can be most easily expressed (and therefore more accurately measured) when all values are higher than about 2 mg/L. Below that level, the uptake kinetics are altered and underestimation of actual demand is likely. Data from 8 profiles were used to calculate oxygen demand in deeper water where replenishment from above is limited. Some vertical oxygen flux may introduce error to these calculations, but using data from the water column below the point of eventual stratification usually provides valid results. Consequently, the difference in oxygen level between corresponding values at each 1 m increment from 8 m of depth in North Pond, 6 m of depth in Middle Pond, and 5 m of depth in South Pond was summed to represent the loss of oxygen over the period of time between each pair of measurements. These differences are adjusted to account for loss of oxygen due to temperature increase. Rising temperature occurs over the targeted period and higher temperatures create lower oxygen saturation levels; not correcting for that factor will cause an overestimate of oxygen demand.

The results are expressed as oxygen loss in g/m$^2$/day, with values over 0.55 g/m$^{2/}$day usually causing eventual anoxia over the period of stratification. Higher oxygen demand causes more rapid loss of oxygen. The values for North Pond based on the spring 2015 data (Figure 28) vary considerably and suggest mixing that will lead to underestimation of actual demand, but 4 of the 7 calculated demand values were >0.55 g/m$^2$/day and averaged close to 1 g/m$^2$/day, a substantial but not overwhelming oxygen demand. This seems consistent with observed oxygen profiles; anoxia does not extend throughout the hypolimnion during the period of stratification.

Oxygen demand for Middle Pond was also variable but generally higher than in North Pond, averaging 1.6 g/m$^2$/day and hitting a peak of almost 3.5 g/m$^2$/day in early May. Oxygen loss was rapid in the spring and extended to the thermocline through summer. South Pond oxygen demand was harder to measure, given a very thin hypolimnion and higher potential for oxygen to diffuse or be mixed downward. The average demand was about 0.7 g/m$^2$/day, but is probably at least twice that, given the likely replacement of consumed oxygen from nearby oxygenated strata. Anoxia was found at the bottom by mid-May and persisted through summer despite shallow depth.

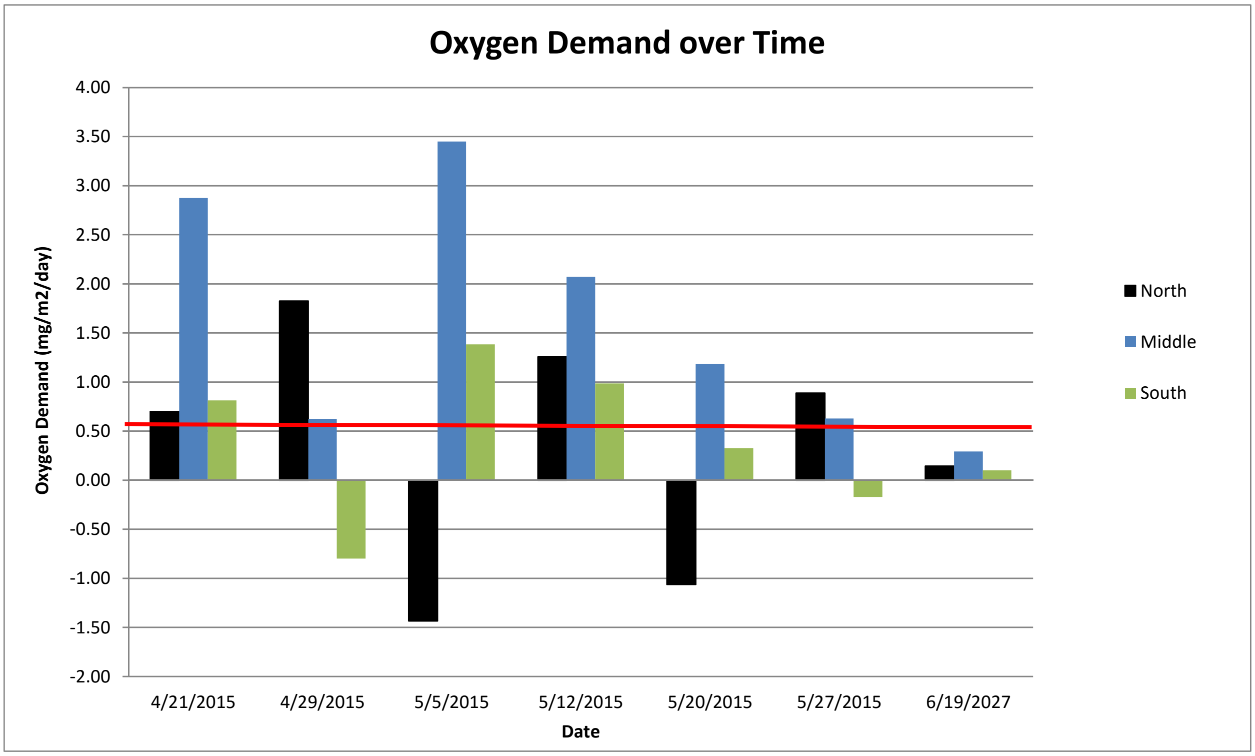



**Figure 28. Oxygen Demand in the Congamond Lakes in 2015**

## Internal Load Assessment

The 2015 data for phosphorus concentration in the two layers of each Congamond pond during summer stratification (Figure 14) are useful for estimating potential internal loading. The data for deep water phosphorus can be used to evaluate release rates, much as the oxygen demand was estimated, using periods of time within the period of stratification during which anoxia is present, mixing appears minimal and changes in concentration are most likely to represent actual releases from sediment. Based on the concentration change pattern in Figure 14, there are several periods where phosphorus accumulation appears suitable for estimating release from each pond. Multiplying the increase in hypolimnetic concentration during each time period by the corresponding volume of the hypolimnion and dividing by area, estimated release rates can be derived (Table 6, first row).

There was considerable variability among dates and ponds, but the best available estimates are shown, averages of values that appeared most representative. The normal range for release rates of iron-bound phosphorus under anoxia is about 2 to 20 mg/m$^2$/day, but most values tend to fall between 3 and 12 mg/m$^2$/day. The value of 3.0 mg/m$^2$/day for North Pond is consistent with oxygen profiles and related expectations for this pond. The value of 18.7 mg/m$^2$/day for Middle Pond seems high, and may be related to the copper treatment in June, which would have caused a lot of algae and associated total phosphorus to enter the deeper water and possibly be recorded as a phosphorus increase that was not related to actual release from sediment. The value of 9.0

**Table 6. Calculation of Phosphorus Release from Sediment in the Congamond Lakes**

| Method or Feature | Units | North | Middle | South |
|---|---|---|---|---|
| Hypolimnetic Mass Change over Area and Time | mg/m2/d | 3.0 | 18.7 | 9.0 |
| Sediment Fe-P % Release over Area and Time | mg/m2/d | 3.4 | 5.8 | 8.2 |
| Minimum Depth of Anoxia | m | 8 | 6 | 5 |
| Maximum Area of Anoxia | m2 | 18300 | 408300 | 203500 |
| Days of Anoxia | Days | 180 | 180 | 120 |
| Anoxic Factor | Days at Lake Area | 11.4 | 49.4 | 54.3 |
| Internal Phosphorus Load | kg/yr | 6.9 | 321.2 | 276.8 |
| Possible P Increase from Internal Load | ug/L | 9.8 | 52.8 | 108.7 |

mg/m$^2$/day for South Pond is actually a doubling of the calculated value, adjusted because we know that mixing is occurring and will move much of the released phosphorus out of the measurement zone over the period of measurement.

The alternative calculation (Table 6, second row) assumes that some percentage of the mass of iron-bound phosphorus can be released in a season of stratification and mixed into the overly water. Dividing this mass by the contributing area and the number of days that contribution is made estimates the average release rate. For North Pond, where anoxia does not reach the thermocline, it is estimated that about 10% of the iron-bound phosphorus can be released. For Middle and South Ponds, with anoxia to the thermocline and both diffusion and mixing events aiding phosphorus movement and limiting the gradient, about 20% of the iron-bound phosphorus may be released. This process results in release estimates of 3.4 mg/m$^2$/day for North Pond, 5.8 mg/m$^2$/day for Middle Pond, and 8.2 mg/m$^2$/day for South Pond. The North and South Ponds estimates are consistent with the values obtained by the other analysis of phosphorus release. The difference between the Middle Pond estimates is taken as further evidence that the first approach may have overestimated release.

The depth at which anoxia occurs influences the size of the area that can contribute phosphorus from the sediment, and the duration of anoxia controls how long that release goes on. Values for those important variables are provided in Table 6 as they are important to management planning, but the anoxic area changes over time, complicating calculation of how much area contributes for how long. A useful construct is the Anoxic Factor, which expresses the anoxic area on any given day as a function of overall lake area, then sums up all areas to get the number of days that an area equivalent to the lake area would be exposed to anoxia (Nurnberg 1995). For North Pond, the Anoxic Factor is 11.4 days (of anoxia at full lake area), while for Middle Pond the Anoxic Factor is 49.4 days, and for South Pond the Anoxic Factor is 54.3 days.

Applying the average release rate for North and South Ponds from Table 6 and the 5.8 mg/m$^2$/day rate obtained for Middle Pond from just the second estimation approach, and multiplying by the Anoxic Factor and the area of each pond, a total possible internal load can be approximated. As a substantial portion of the internal load of phosphorus may recombine with iron and precipitate back out of solution when exposed to oxygen, these estimates are likely maximum values. Yet they provide some feel for the magnitude of possible internal phosphorus loading.

The estimation method above suggests a load of just under 7 kg/yr for North Pond, and given the apparent accumulation of algae near the thermocline using this phosphorus source, much of that probably is manifested. If that 7 kg of phosphorus was mixed into the entire water volume of North Pond, it would raise the concentration by almost 10 µg/L. As the average concentration was only 12 µg/L in 2015 and averages 19 µg/L over the last decade, this is a large fraction of the expected total load. If even half of the internal load becomes available, it is still an important portion of the observed phosphorus level, but that level is not normally excessive.

For Middle Pond, even ignoring the large release estimate by one method, the internal load by this calculation is about 321 kg/yr and could raise the phosphorus concentration of the entire pond by almost 52 µg/L. In-lake values are not that high, so much of this load probably precipitates out and is not active, but the potential for internal loading to dominate the phosphorus supply is apparent. Based on the observed pattern of phosphorus build-up in deep water, it would be reasonable to assume that about half the generated internal load is assimilated into the upper water layer, more than enough to support algae blooms.

The South Pond internal load calculates to almost 277 kg/yr, enough to raise the phosphorus concentration in the entire pond by over 100 µg/L. As with Middle Pond, much of the potential internal load probably precipitates out upon oxygenation, but again the potential for internal loading to dominate phosphorus supply is evident. Given the thin bottom layer and observed pattern of build-up, it is reasonable to assume that about two thirds of the internal load is assimilated into the upper water layer, more than enough to support algae blooms.

## Modeling of Watershed and Lake

The LLRM model was applied, using land use data from 2010 and export coefficients for water, phosphorus and nitrogen representative of similar land uses in the region for which actual export values could be calculated from downstream data. Land use (Table 7) does not include the ponds themselves, although North Pond must be considered in the Middle Pond model and North and Middle Ponds must be included in the South Pond model. Land use is variable among the direct drainage basins of the three ponds, with North Pond having mostly forest and medium density residential area in its watershed, Middle Pond having a mix of forest, residential, and crop land, and South Pond having a broad array of uses with a more even distribution of land among them.

The North Pond drainage area covers 105 hectares (260 acres), an area 5.5 times the area of the pond. The Middle Pond direct drainage area covers 603 hectares (1495 acres), but North Pond and its watershed also drain to Middle Pond, resulting in a contributing area 6.5 times the area of the pond. The South Pond drainage area covers 188 hectares (466 acres), but with the North and Middle Pond drainage also contributing to South Pond, the total drainage area to South Pond is 17.3 times the area of the pond. When the watershed to pond area ratio is <10:1, conditions are easier to manage and watershed features tend to have less immediate influence on pond water quality. As the watershed:lake area ratio increases, the probability of water quality problems increases, even with a largely forested watershed.

**Table 7. Land Use Data for the Direct Drainage Areas of Congamond Lakes**

| | North | Middle | South |
|---|---|---|---|
| **Land_Use_Description** | **Area (Ha)** | **Area (Ha)** | **Area (Ha)** |
| Urban 1 (LDR) | 5.39 | 25.63 | 2.09 |
| Urban 2 (MDR/Hwy) | 34.92 | 33.70 | 17.59 |
| Urban 3 (HDR/Com) | 2.16 | 76.19 | 28.45 |
| Urban 4 (Ind) | 0.00 | 0.00 | 0.00 |
| Urban 5 (P/I/R/C) | 0.02 | 31.60 | 0.01 |
| Agric 1 (Cvr Crop) | 0.00 | 12.30 | 2.63 |
| Agric 2 (Row Crop) | 0.00 | 193.46 | 17.61 |
| Agric 3 (Grazing) | 0.00 | 0.00 | 0.00 |
| Agric 4 (Feedlot) | 0.00 | 0.00 | 0.00 |
| Forest 1 (Upland) | 47.79 | 167.69 | 55.24 |
| Forest 2 (Wetland) | 0.00 | 15.03 | 25.64 |
| Open 1 (Wetland/Lake) | 0.87 | 15.58 | 12.66 |
| Open 2 (Meadow) | 0.21 | 12.50 | 7.94 |
| Open 3 (Excavation) | 13.87 | 19.30 | 17.74 |
| | | | |
| Total | 105.23 | 602.98 | 187.61 |

Land use areas are multiplied by export coefficients representing corresponding water, phosphorus and nitrogen outputs, and the resulting loads are multiplied by an attenuation factor representing loss of load on the way to the pond, based on the routing path and any management practices in place. Export coefficients and attenuation factors can be adjusted within the range of known values to get the predicted in-lake values for nutrients, clarity and chlorophyll to match the actual data. Model calibration did require export coefficients and attenuation factors near the low end of the normal range to get reasonable agreement between model predictions and known values, but in the absence of major tributaries this is not surprising. All predicted water load and phosphorus and nitrogen concentrations were within 7% of "reality check" values based on data.

Atmospheric loading was estimated as the volume of direct precipitation times the average phosphorus and nitrogen concentrations in rainfall in this region from other studies. Internal loading was based on the release rates (in mg/m2/day) derived from 2015 data, with extent and duration of contribution considered and downward adjustment made to account for much of each internal load never getting assimilated into the upper water layer. Waterfowl loading was estimated as the equivalent of 50 birds being present all year with average inputs of 0.2 mg P/bird/yer and 0.95 mg N/bird/yr, typical literature rates; there are few data on waterfowl inputs, but birds are unlikely to be a major factor in this system. Direct septic system inputs were assumed to be negligible from the Massachusetts drainage area to the ponds in light of sewering. On-site wastewater inputs from the Connecticut drainage area are estimated based on number of dwellings, distance from the lake, and an assumed occupancy level of 3 people nearly full time.

The resulting loading summaries for current conditions (Tables 8-10) indicate that internal loading is the dominant component of phosphorus load in Middle Pond, is comparable to watershed loading in South Pond, but is not the major component in North Pond. This situation is more complicated than that simple statement would indicate, however. For North Pond, the internal load occurs all in summer into fall, the prime period for algae blooms, while other sources are distributed more evenly around the year. Consequently, the internal load is as large as from any other source during the time it is added. Additionally, algae may be growing on bottom sediment then floating upward, making the sediment a source of nutrients beyond that which is measured as internal load. Also, about half of the internal load is assumed to return to the sediment before it can be used by algae, potentially underestimating the real contribution. Finally, the watershed inputs as measured in storm water sampling are largely particulate forms that are not readily available for algae uptake. This particulate phosphorus settles to the bottom, is processed, and may be later manifest as internal load. Consequently, internal loading in North Pond is a highly significant source despite only representing 17% of the calculated total phosphorus load.

For Middle Pond internal loading is already the dominant component of the phosphorus load, and the logic above suggests it is even more dominant than the 55% of total phosphorus load it represents. For South Pond, a large portion of the total phosphorus load comes from Middle Pond and is counted as external load, as it is not generated within South Pond. Yet much of that external load was generated in Middle Pond as internal load there, so while the actual internal load to South Pond is only 40% of the total, internal loading for the whole 3 pond system impacts South Pond to a greater degree. And because of the seasonal distribution of that load, internal loading is more important to algae production in South Pond that its percentage would indicate.

The nitrogen loading situation is quite different than that for phosphorus, as nitrogen is less easily trapped by soils. Even though the internal load of nitrogen is estimated at three times the phosphorus load, there is so much more nitrogen coming from the watershed that internal nitrogen loading does not appear as important. The only caveat there is that most of the internal load will be as ammonium during periods of low oxygen at the bottom, and some portion of that ammonium will be present as un-ionized ammonia, which can be toxic to fish. Values from 1981 through 2015 for deep water sampling indicated ammonium levels often in excess of 1 mg/L and maximum values of 4 mg/L, 2.9 mg/L and 4 mg/L in North, Middle and South Ponds, respectively (Appendix). That deep water is likely to be toxic to fish and some toxicity might be expected if mixing is rapid.

Loads of nitrogen are much higher than those for phosphorus, but algae and plants use nitrogen in much greater quantity than phosphorus (at a ratio of about 15:1 on average). The N:P ratios suggest that phosphorus is in shorter supply relative to nitrogen most of the time, although deep water N:P ratios are routinely lower than surface water ratios and do indicate nitrogen shortage some of the time. Lower N:P ratios favor cyanobacteria, which can also use dissolved gaseous nitrogen that is not included in the ratio and unavailable to other algae. While both nitrogen and phosphorus are important to algae production, we tend to view phosphorus as controlling the quantity of algae produced and nitrogen as determining which types of algae will be most abundant, by virtue of the ratio to phosphorus.

Table 8. Loading Summary for Current Conditions in North Pond

| Loading Source | Water (Cu.M/Yr) | Water (%) | P (Kg/Yr) | P (%) | N (Kg/Yr) | N (%) |
|---|---|---|---|---|---|---|
| Atmospheric | 229900 | 28.1 | 3.8 | 12.3 | 123.5 | 21.9 |
| Internal | 0 | 0.0 | 5.3 | 17.1 | 15.8 | 2.8 |
| Waterfowl | 0 | 0.0 | 2.0 | 6.5 | 9.5 | 1.7 |
| Septic Systems | 0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Watershed Load | 587781 | 71.9 | 19.8 | 64.2 | 414.1 | 73.6 |
| | | | | | | |
| Total Load to Pond | 817681 | 100.0 | 30.9 | 100.0 | 562.9 | 100.0 |

Table 9. Loading Summary for Current Conditions in Middle Pond

| Loading Source | Water (Cu.M/Yr) | Water (%) | P (Kg/Yr) | P (%) | N (Kg/Yr) | N (%) |
|---|---|---|---|---|---|---|
| Atmospheric | 1356410 | 24.5 | 22.4 | 5.8 | 728.7 | 12.9 |
| Internal | 0 | 0.0 | 213.1 | 55.0 | 639.4 | 11.3 |
| Waterfowl | 0 | 0.0 | 12.0 | 3.1 | 57.0 | 1.0 |
| Septic Systems | 31950 | 0.6 | 25.6 | 6.6 | 575.1 | 10.2 |
| Watershed Load | 4141850 | 74.9 | 114.2 | 29.5 | 3650.3 | 64.6 |
| | | | | | | |
| Total Load to Pond | 5530210 | 100.0 | 387.3 | 100.0 | 5650.4 | 100.0 |

Table 10. Loading Summary for Current Conditions in South Pond

| Loading Source | Water (Cu.M/Yr) | Water (%) | P (Kg/Yr) | P (%) | N (Kg/Yr) | N (%) |
|---|---|---|---|---|---|---|
| Atmospheric | 717530 | 11.2 | 11.9 | 3.4 | 385.5 | 8.0 |
| Internal | 0 | 0.0 | 140.0 | 40.5 | 420.0 | 8.7 |
| Waterfowl | 0 | 0.0 | 6.0 | 1.7 | 28.5 | 0.6 |
| Septic Systems | 17100 | 0.3 | 13.7 | 4.0 | 307.8 | 6.4 |
| Watershed Load | 5693216 | 88.6 | 174.1 | 50.4 | 3662.0 | 76.2 |
| | | | | | | |
| Total Load to Pond | 6427846 | 100.0 | 345.7 | 100.0 | 4803.8 | 100.0 |

For the most part, loads from the atmosphere, birds and septic systems are relatively minor components of the total phosphorus and nitrogen loads. As none of these components was directly measured, these values are somewhat speculative, but they make sense in light of known conditions in the Congamond Lakes. Atmospheric loads are rarely high, the number of birds on the water is not extreme, and most of the Massachusetts portion of the watershed has been sewered. The Connecticut portion of the Middle Pond watershed is fairly high above the pond, allowing for much phosphorus adsorption onto soil particles before effluents can reach the pond. Systems in the Connecticut portion of the South Pond watershed are mostly as lower elevation, but the estimated contribution is not large relative to the internal load.

Other watershed sources are varied, but fertilizers are generally recognized as major contributors of phosphorus and nitrogen. Fertilizer companies are eliminating high phosphorus lawn fertilizers as some states have banned them, so this problem should diminish over time, at least from residential areas. The substantial crop land in the Middle Pond watershed is likely a significant contributor, but given its distance to the pond and lack of a direct flow tributary, this source is expected to be largely attenuated before reaching the pond. The directly contributing watersheds of each pond are not especially large relative to the area of each pond, but as water flows from North to Middle to South Pond, each pond in that series gets contributions from the watersheds of the upgradient ponds. Still, the internal phosphorus load is the largest factor in at least Middle and South Ponds, and is significant in North Pond as well.

One factor not considered in this model is the Great Brook drainage (Figure 2). While originally an outlet near the south end of Middle Pond, Great Brook was blocked off so that water would flow though the canal at the south end of South Pond to support commerce and has not been an active outlet for many years. However, infilling in the absence of constant flow from Middle Pond has allowed Great Brook to flow back toward Middle Pond during storms, functioning as an inlet. There is potentially a very large area, much of it developed or agricultural, that could contribute loading to Middle Pond. We do not know the extent of the contributory area, the frequency with which flow reversal occurs, or the magnitude of loading to Middle Pond from Great Brook. It is unlikely to be substantial compared to the internal load, but remains an unquantified factor that may require future investigation and management.

Comparing water loads (Tables 8-10) and pond volumes (Table 2), the rate of flushing and detention time for each pond can be determined. North Pond flushes 1.1 times per year, equating to a detention time of 332 days. Middle Pond flushes 0.9 times per year, a detention time of 406 days. South Pond flushes 2.5 times per year, a detention time of 146 days. These are average values, with variation possible by season and year depending on the amount of precipitation, but all values are long enough to allow the full effect of nutrient loading to be manifest in the ponds. Any thought of improving flushing rate as a management action should be abandoned; there is simply not enough water to be had to make a meaningful difference in the detention time in these ponds. Circulation might be improved, but that is a different management approach than altering the overall flushing rate.

Once the model for each pond is calibrated to the extent that results reasonably match actual data without setting model variable values outside the realm of reason, the potential impact of various possible management options can be evaluated. We can set all developed or agricultural land uses to forested land and minimize internal loading to estimate "background" conditions in the ponds prior to major human interaction with the ponds and their watersheds. We can leave land use alone and just reduce the internal load, an action that can be achieved by dredging, oxygenation, circulation or phosphorus inactivation. We can apply best management practices in the watershed to reduce pollutant loading, with maximum reduction based on an extensive USEPA database reflected in the model as the best reduction we can expect. Note that many best management practices (BMPs) have already been implemented in the Massachusetts portion of the watersheds of the Congamond Lakes under various programs, mostly notably Section 319 of the Clean Water Act as administered by the MA DEP. Past projects may limit the maximum extent of additional benefit achievable with future projects through the model. These loading scenarios (Tables 11-13) provide insight into probable background conditions and the efficacy of different management options.

For all three ponds, the projected original background phosphorus level was well under 10 µg/L and productivity was low. Sometimes, especially with large watersheds and shallow lakes, the background condition is not pristine at all, while for smaller watersheds and deeper lakes, the condition is often highly desirable, and that appears to be the case with all three Congamond ponds. The ponds have not experienced conditions like those predicted for background watershed conditions since long before any monitoring commenced, and restoration to such conditions is not a realistic expectation.

Working from the current conditions and reducing the internal load by 90%, something that can be achieved by multiple means, North Pond is predicted to have an average phosphorus concentration of 15 µg/L, Secchi transparency of 4.6 m (15.2 feet), and chlorophyll in excess of 10 µg/L about 6.3% of the time. These conditions represent an improvement over the current condition and involve just control of internal loading of phosphorus, but the change is not dramatic. As North Pond is in the best condition of the three ponds, it does not need as great a level of improvement, and the key to improving North Pond is not really a change in average phosphorus level. Rather, we need to prevent cyanobacteria blooms, which relates to either accumulation at the thermocline or growth on nutrient-rich bottom sediments where phosphorus levels are considerably higher than the pondwide average. The methods intended to reduce internal loading are appropriate to this task, and may do more than the simple model predicts.

For Middle or South Ponds, a 90% reduction in internal load represents a major shift in pond condition. The internal load is a primary component of phosphorus concentrations and the dominant source during summer when blooms are most likely to occur. Average phosphorus in Middle Pond would be 13 µg/L with a single treatment of surficial sediment in water deep enough to experience anoxia, while treatment of just South Pond would lower its phosphorus concentration to 19 µg/L. If both Middle and South Pond were treated, South Pond would experience an average phosphorus level of 11 µg/L, since so much of its phosphorus load comes from Middle Pond. With treatment of both Middle and South Ponds, clarity would rise to at least 4 m (>13 feet) and chlorophyll would exceed 10 µg/L <3% of the time. This would meet perceived goals for these ponds with no other management action.

**Table 11. Resulting Pond Features for North Pond Management Scenarios**

| NORTH POND SUMMARY TABLE FOR SCENARIO TESTING | Existing Conditions | | Background Conditions | 90% Internal Load Reduction | All Feasible BMPs | 90% Load Reduction and All Feasible BMPs |
|---|---|---|---|---|---|---|
| Pond Feature | Calibrated Model Value | Actual Data | Model Value | Model Value | Model Value | Model Value |
| Phosphorus (ppb) | 19 | 19 | 7 | 15 | 17 | 13 |
| Nitrogen (ppb) | 436 | 432 | 287 | 425 | 372 | 361 |
| Mean Chlorophyll (ug/L) | 6.8 | 7.9 | 1.8 | 5.3 | 5.8 | 4.3 |
| Peak Chlorophyll (ug/L) | 23.7 | 23.0 | 7.1 | 18.6 | 20.4 | 15.4 |
| Mean Secchi (m) | 4.3 | 4.2 | 5.6 | 4.6 | 4.5 | 4.7 |
| Bloom Probability | | | | | | |
| Probability of Chl >10 ug/L | 15.6% | 21.4% | 0.0% | 6.3% | 9.2% | 2.7% |
| Probability of Chl >15 ug/L | 3.4% | 9.5% | 0.0% | 1.0% | 1.6% | 0.3% |
| Probability of Chl >20 ug/L | 0.8% | 4.8% | 0.0% | 0.2% | 0.3% | 0.0% |
| Probability of Chl >30 ug/L | 0.1% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% |
| Probability of Chl >40 ug/L | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% |

**Table 12. Resulting Pond Features for Middle Pond Management Scenarios**

| MIDDLE POND SUMMARY TABLE FOR SCENARIO TESTING | Existing Conditions | | Background Conditions | 90% Internal Load Reduction | All Feasible BMPs | 90% Load Reduction and All Feasible BMPs |
|---|---|---|---|---|---|---|
| Pond Feature | Calibrated Model Value | Actual Data | Model Value | Model Value | Model Value | Model Value |
| Phosphorus (ppb) | 32 | 31 | 7 | 13 | 30 | 11 |
| Nitrogen (ppb) | 561 | 551 | 343 | 510 | 522 | 469 |
| Mean Chlorophyll (ug/L) | 13.9 | 19.2 | 1.8 | 4.4 | 12.7 | 3.5 |
| Peak Chlorophyll (ug/L) | 46.4 | 45.6 | 6.9 | 15.6 | 42.6 | 12.7 |
| Mean Secchi (m) | 2.7 | 2.4 | 5.5 | 4.0 | 2.8 | 4.3 |
| Bloom Probability | | | | | | |
| Probability of Chl >10 ug/L | 65.9% | 74.0% | 0.0% | 2.9% | 59.1% | 0.9% |
| Probability of Chl >15 ug/L | 34.4% | 62.0% | 0.0% | 0.3% | 28.0% | 0.1% |
| Probability of Chl >20 ug/L | 16.4% | 36.0% | 0.0% | 0.1% | 12.4% | 0.0% |
| Probability of Chl >30 ug/L | 3.7% | 18.0% | 0.0% | 0.0% | 2.5% | 0.0% |
| Probability of Chl >40 ug/L | 0.9% | 6.0% | 0.0% | 0.0% | 0.5% | 0.0% |

## Table 13. Resulting Pond Features for South Pond Management Scenarios

| SOUTH POND SUMMARY TABLE FOR SCENARIO TESTING | Existing Conditions | | Background Conditions | 90% Internal Load Reduction: SP Only | 90% Internal Load Reduction: MP & SP | All Feasible BMPs | 90% Load Reduction and All Feasible BMPs |
|---|---|---|---|---|---|---|---|
| Pond Feature | Calibrated Model Value | Actual Data | Model Value | Model Value | Model Value | Model Value | Model Value |
| Phosphorus (ppb) | 33 | 32 | 8 | 19 | 11 | 29 | 10 |
| Nitrogen (ppb) | 511 | 612 | 306 | 473 | 445 | 441 | 401 |
| Mean Chlorophyll (ug/L) | 14.4 | 21.0 | 2.3 | 7.1 | 3.3 | 11.9 | 2.8 |
| Peak Chlorophyll (ug/L) | 48.1 | 60.2 | 8.9 | 24.6 | 12.2 | 40.0 | 10.5 |
| Mean Secchi (m) | 2.6 | 2.0 | 5.0 | 3.4 | 4.4 | 2.8 | 4.6 |
| Bloom Probability | | | | | | | |
| Probability of Chl >10 ug/L | 68.6% | 81.1% | 0.1% | 17.6% | 0.7% | 53.8% | 0.3% |
| Probability of Chl >15 ug/L | 37.2% | 56.6% | 0.0% | 4.1% | 0.1% | 23.7% | 0.0% |
| Probability of Chl >20 ug/L | 18.4% | 50.9% | 0.0% | 1.0% | 0.0% | 9.8% | 0.0% |
| Probability of Chl >30 ug/L | 4.3% | 17.0% | 0.0% | 0.1% | 0.0% | 1.8% | 0.0% |
| Probability of Chl >40 ug/L | 1.1% | 9.4% | 0.0% | 0.0% | 0.0% | 0.4% | 0.0% |

Altering the current conditions to reflect maximum application of BMPs in the watershed but no in-lake work to control internal load, phosphorus would be expected to average 17 µg/L in North Pond, 30 µg/L in Middle Pond, and 29 µg/L in South Pond. Water clarity would be acceptable in North Pond most of the time, but then it is acceptable now most of the time. In 2015, with a very dry spring, North Pond phosphorus averaged 12 µg/L near the surface. This is generally acceptable for all pond uses, and clarity was high all summer. However, phosphorus in deeper water averaged 50 µg/L and algae apparently grew near the thermocline causing a bloom when stratification broke down in the fall. Bloom control appears to be a function of internal load control in North Pond, and watershed BMPs will not provide that control.

With extended watershed management, clarity would average 2.8 m (9.2 feet) in Middle and South Ponds and chlorophyll would exceed 10 µg/L more than half the time; this is unlikely to meet user preferences. Getting such a reduction would be a slow and expensive process, based on experience elsewhere and literature examples. This scenario extends the type of work that has been done over the last two decades in the watershed, and would represent application of similar techniques to more locations at increasing cost. Watershed management should be part of any long-term plan to protect a lake, but diminishing returns on investment, ongoing maintenance needs, and practical limits to application generally prevent reductions of more than 50%. As substantial effort has already been put into watershed management for Congamond Lakes, the expected reduction in this case is much less than 50%. Additionally, watershed management will not control internal loading, and this is reflected in the predicted outcome within the lake.

Combining the maximum feasible watershed approach and internal load reduction, the model predicts an average phosphorus level of 13 µg/L in North Pond, 11 µg/L in Middle Pond, and 10 µg/L in South Pond. While higher than the postulated background concentrations, these would be highly advantageous values in these ponds. However, these phosphorus levels are not appreciably lower than what can be achieved with just internal load control, which will provide immediate relief from blooms and will be less expensive than maximum watershed management. Watershed management and internal load control are basically additive in this case; watershed management would limit the build-up of phosphorus reserves that fuel the internal load, but there is already so much phosphorus present that the linkage is not tight at all and internal load control is necessary to control algae blooms. Certainly ongoing watershed management is worthwhile, but discerning choices should be made about where to expend funds, getting the largest loading reductions possible per dollar spent.

## Target Phosphorus Concentration

The choice of a target phosphorus level is partly subjective; greater water clarity is obviously desirable, but increased rooted plant management needs are to be expected as clarity increases. The primary goal with regard to algae control would seem to be the elimination of cyanobacteria blooms, as these include floating scums and can produce taste and odor and toxicity. Moderate water clarity with other algae dominating would enhance the food web and improve fishing.

Predicted background conditions are unlikely to be achieved with current watershed land use, but lowering fertility to those levels may not be desirable from a fish and wildlife management viewpoint anyway. Phosphorus criteria established by the USEPA for New England suggest values near 10 µg/L to avoid algae blooms and use impairment, but this has proven inadequate in cases where the algae grow at the sediment-water interface and then float upward to cause blooms in lakes with relatively low water column phosphorus levels. And in many cases it is cyanobacteria that use this bloom formation mode, suggesting that both water column phosphorus concentration and surficial sediment phosphorus availability must be lowered. This strongly favors including a technique to address internal loading in the management plan.

Some discussion among interested town parties and members of the Lake Management Committee is warranted, but we can approximate a threshold by considering desired clarity and acceptable frequency of algae nuisances. As algae abundance is a distribution as described above, having no nuisances is not really a legitimate goal in almost any lake, but minimizing the frequency of algae above some target level is appropriate. A chlorophyll level of 10 µg/L is often applied, as values below 10 µg/L are associated with adequate clarity for most contact recreation but still represent adequate fertility to support a desirable fishery. However, as the ratio of chlorophyll to biomass is variable among algal groups, 10 µg/L of cyanobacteria chlorophyll may be a lot more biomass than for 10 µg/L of green algae chlorophyll. No threshold is perfect, but if chlorophyll remains under 10 µg/L at least 90% of the time, serious blooms will be unlikely.

Clarity relates directly to algae and the phytopigment chlorophyll for the Congamond Lakes, albeit with variability. Incoming storm water carries substantial non-algal particles, but settling appears rapid and most turbidity in the lake appears to be from algae. Secchi transparency is usually used as an indication of lake suitability for contact recreation. Values under about 1.2 m (4 feet) are usually associated with algae blooms and undesirable visibility, and used to be grounds for closing beaches in Massachusetts. Values >2 m (6.7 ft) are considered adequate for swimming, but most people prefer clarity near 3 m (10 feet). Values approaching 5 m (16.5 feet) are rare in Massachusetts lakes in summer, and would be considered outstanding clarity. The current averages for the ponds exceed 2 m, but values in summer are often closer to 1 m in Middle and South Ponds. An average value >3 m would seem like an appropriate target for Secchi transparency.

Translating a $90^{th}$ percentile chlorophyll value of 10 µg/L and an average Secchi transparency of 3 m into a target phosphorus concentration with the model suggests a value between 15 and 17 µg/L for each pond. Any phosphorus concentration within or below that range should provide desirable conditions based on the proposed criteria. If different levels of clarity and chlorophyll are preferred, a different phosphorus threshold can be calculated, but this seems to be a reasonable starting point for discussion.

Referring to Tables 11-13, a 90% reduction of internal loading meets the goal in each case, while maximum practical application of BMPs will only achieve the desired conditions in North Pond. The combination of more BMPs in the watershed plus internal load reduction is more than adequate, so control of internal loading with continued attention to watershed loading as opportunities present themselves seems advisable.

## Management Options Review

There is a wide variety of options for managing algae, ranging from watershed techniques to limit the input of nutrients to physical techniques such as flushing or dredging to chemical techniques such as algaecides or phosphorus inactivation to biological techniques such as food web manipulations to maximize grazing (Table 14). These techniques attempt to either reduce growth or increase loss of algae, and the duration of benefits varies widely by technique. If external loading is low, dredging or phosphorus inactivation may provide decades of improved conditions. If external loading is high, repetitive algaecide applications or near constant input of air or oxygen may be necessary to minimize blooms. The key is to match techniques to the situation with consideration of technical feasibility and expected results, economic affordability and longer term sustainability, and institutional acceptability, which includes everything from acceptance by the Town and lake users to permits from state agencies.

High algal productivity is not necessarily undesirable; if algae are efficiently processed in the food web, biomass will not build up and more fish will be produced while acceptable water clarity is maintained. However, it is rare to have a biological structure that can process as much algae as can be produced at high nutrient levels, so keeping nutrient levels in a low to moderate range is usually a primary goal. Lowering the overall fertility of the system is preferred where feasible.

**Table 14. Options for Control of Algae and Floating Plants (Adapted from Wagner 2001)**

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| **WATERSHED CONTROLS** | | | |
| 1) Management for nutrient input reduction | ♦ Includes wide range of watershed and lake edge activities intended to eliminate nutrient sources or reduce delivery to lake<br>♦ Essential component of algal control strategy where internal recycling is not the dominant nutrient source, and desired even where internal recycling is important | ♦ Acts against the original source of algal nutrition<br>♦ Creates sustainable limitation on algal growth<br>♦ May control delivery of other unwanted pollutants to lake<br>♦ Facilitates ecosystem management approach which considers more than just algal control | ♦ May involve considerable lag time before improvement observed<br>♦ May not be sufficient to achieve goals without some form of in-lake management<br>♦ Reduction of overall system fertility may impact fisheries<br>♦ May cause shift in nutrient ratios which favor less desirable algae |
| 1a) Point source controls | ♦ More stringent discharge requirements<br>♦ May involve diversion<br>♦ May involve technological or operational adjustments<br>♦ May involve pollution prevention plans | ♦ Often provides major input reduction<br>♦ Highly efficient approach in most cases<br>♦ Success easily monitored | ♦ May be very expensive in terms of capital and operational costs<br>♦ May transfer problems to another watershed<br>♦ Variability in results may be high in some cases |
| 1b) Non-point source controls | ♦ Reduction of sources of nutrients<br>♦ May involve elimination of land uses or activities that release nutrients<br>♦ May involve alternative product use, as with no phosphate fertilizer | ♦ Removes source<br>♦ Limited ongoing costs | ♦ May require purchase of land or activity<br>♦ May be viewed as limitation of "quality of life"<br>♦ Usually requires education and gradual implementation |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 1c) Non-point source pollutant trapping | ♦ Capture of pollutants between source and lake<br>♦ May involve drainage system alteration<br>♦ Often involves wetland treatments (det./infiltration)<br>♦ May involve storm water collection and treatment as with point sources | ♦ Minimizes interference with land uses and activities<br>♦ Allows diffuse and phased implementation throughout watershed<br>♦ Highly flexible approach<br>♦ Tends to address wide range of pollutant loads | ♦ Does not address actual sources<br>♦ May be expensive on necessary scale<br>♦ May require substantial maintenance |
| **IN-LAKE PHYSICAL CONTROLS** | | | |
| 2) Circulation and destratification | ♦ Use of water or air to keep water in motion<br>♦ Intended to prevent or break stratification<br>♦ Generally driven by mechanical or pneumatic force | ♦ Reduces surface build-up of algal scums<br>♦ May disrupt growth of blue-green algae<br>♦ Counteraction of anoxia improves habitat for fish/invertebrates<br>♦ Can eliminate localized problems without obvious impact on whole lake | ♦ May spread localized impacts<br>♦ May lower oxygen levels in shallow water<br>♦ May promote downstream impacts |
| 3) Dilution and flushing | ♦ Addition of water of better quality can dilute nutrients<br>♦ Addition of water of similar or poorer quality flushes system to minimize algal build-up<br>♦ May have continuous or periodic additions | ♦ Dilution reduces nutrient concentrations without altering load<br>♦ Flushing minimizes detention; response to pollutants may be reduced | ♦ Diverts water from other uses<br>♦ Flushing may wash desirable zooplankton from lake<br>♦ Use of poorer quality water increases loads<br>♦ Possible downstream impacts |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 4) Drawdown | ♦ Lowering of water over autumn period allows oxidation, desiccation and compaction of sediments<br>♦ Duration of exposure and degree of dewatering of exposed areas are important<br>♦ Algae are affected mainly by reduction in available nutrients. | ♦ May reduce available nutrients or nutrient ratios, affecting algal biomass and composition<br>♦ Opportunity for shoreline clean-up/structure repair<br>♦ Flood control utility<br>♦ May provide rooted plant control as well | ♦ Possible impacts on non-target resources<br>♦ Possible impairment of water supply<br>♦ Alteration of downstream flows and winter water level<br>♦ May result in greater nutrient availability if flushing inadequate |
| 5) Dredging | ♦ Sediment is physically removed by wet or dry excavation, with deposition in a containment area for dewatering<br>♦ Dredging can be applied on a limited basis, but is most often a major restructuring of a severely impacted system<br>♦ Nutrient reserves are removed and algal growth can be limited by nutrient availability | ♦ Can control algae if internal recycling is main nutrient source<br>♦ Increases water depth<br>♦ Can reduce pollutant reserves<br>♦ Can reduce sediment oxygen demand<br>♦ Can improve spawning habitat for many fish species<br>♦ Allows complete renovation of aquatic ecosystem | ♦ Temporarily removes benthic invertebrates<br>♦ May create turbidity<br>♦ May eliminate fish community (complete dry dredging only)<br>♦ Possible impacts from containment area discharge<br>♦ Possible impacts from dredged material disposal<br>♦ Interference with recreation or other uses during dredging |
| 5a) "Dry" excavation | ♦ Lake drained or lowered to maximum extent practical<br>♦ Target material dried to maximum extent possible<br>♦ Conventional excavation equipment used to remove sediments | ♦ Tends to facilitate a very thorough effort<br>♦ May allow drying of sediments prior to removal<br>♦ Allows use of less specialized equipment | ♦ Eliminates most aquatic biota unless a portion left undrained<br>♦ Eliminates lake use during dredging |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 5b) "Wet" excavation | ♦ Lake level may be lowered, but sediments not substantially exposed<br>♦ Draglines, bucket dredges, or long-reach backhoes used to remove sediment | ♦ Requires least preparation time or effort, tends to be least cost dredging approach<br>♦ May allow use of easily acquired equipment<br>♦ May preserve aquatic biota | ♦ Usually creates extreme turbidity<br>♦ Normally requires intermediate containment area to dry sediments prior to hauling<br>♦ May disrupt ecological function<br>♦ Use disruption |
| 5c) Hydraulic removal | ♦ Lake level not reduced<br>♦ Suction or cutterhead dredges create slurry which is hydraulically pumped to containment area<br>♦ Slurry is dewatered; sediment retained, water discharged | ♦ Creates minimal turbidity and impact on biota<br>♦ Can allow some lake uses during dredging<br>♦ Allows removal with limited access or shoreline disturbance | ♦ Often leaves some sediment behind<br>♦ Cannot handle coarse or debris-laden materials<br>♦ Requires sophisticated and more expensive containment area |
| 6) Light-limiting dyes and surface covers | ♦ Creates light limitation | ♦ Creates light limit on algal growth without high turbidity or great depth<br>♦ May achieve some control of rooted plants as well | ♦ May cause thermal stratification in shallow ponds<br>♦ May facilitate anoxia at sediment interface with water |
| 6.a) Dyes | ♦ Water-soluble dye is mixed with lake water, thereby limiting light penetration and inhibiting algal growth<br>♦ Dyes remain in solution until washed out of system. | ♦ Produces appealing color<br>♦ Creates illusion of greater depth | ♦ May not control surface bloom-forming species<br>♦ May not control growth of shallow water algal mats<br>♦ Altered thermal regime |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 6.b) Surface covers | ♦ Opaque sheet material applied to water surface | ♦ Minimizes atmospheric and wildlife pollutant inputs | ♦ Minimizes atmospheric gas exchange<br>♦ Limits recreation |
| 7) Mechanical removal | ♦ Filtering of pumped water for water supply purposes<br>♦ Collection of floating scums or mats with booms, nets, or other devices<br>♦ Continuous or multiple applications per year usually needed | ♦ Algae and associated nutrients can be removed from system<br>♦ Surface collection can be applied as needed<br>♦ May remove floating debris<br>♦ Collected algae dry to minimal volume | ♦ Filtration requires high backwash and sludge handling capability<br>♦ Labor and/or capital intensive<br>♦ Variable collection efficiency<br>♦ Possible impacts on non-target aquatic life |
| 8) Selective withdrawal | ♦ Discharge of bottom water which may contain (or be susceptible to) low oxygen and higher nutrient levels<br>♦ May be pumped or utilize passive head differential | ♦ Removes targeted water from lake efficiently<br>♦ May prevent anoxia and phosphorus build up in bottom water<br>♦ May remove initial phase of algal blooms which start in deep water<br>♦ May create coldwater conditions downstream | ♦ Possible downstream impacts of poor water quality<br>♦ May promote mixing of remaining poor quality bottom water with surface waters<br>♦ May cause unintended drawdown if inflows do not match withdrawal |
| 9) Sonication | ♦ Sound waves disrupt algal cells | ♦ Supposedly affects only algae (new technique)<br>♦ Applicable in localized areas | ♦ Unknown effects on non-target organisms<br>♦ May release cellular toxins or other undesirable contents into water column |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 10) Hypolimnetic aeration or oxygenation | ♦ Addition of air or oxygen provides oxic conditions<br>♦ Maintains stratification<br>♦ Can also withdraw water, oxygenate, then replace | ♦ Oxic conditions reduce P availability<br>♦ Oxygen improves habitat<br>♦ Oxygen reduces build-up of reduced cpds | ♦ May disrupt thermal layers important to fish community<br>♦ Theoretically promotes supersaturation with gases harmful to fish |
| **IN-LAKE CHEMICAL CONTROLS** | | | |
| 11) Algaecides | ♦ Liquid or pelletized algaecides applied to target area<br>♦ Algae killed by direct toxicity or metabolic interference<br>♦ Typically requires application at least once/yr, often more frequently | ♦ Rapid elimination of algae from water column , normally with increased water clarity<br>♦ May result in net movement of nutrients to bottom of lake | ♦ Possible toxicity to non-target species<br>♦ Restrictions on water use for varying time after treatment<br>♦ Increased oxygen demand and possible toxicity<br>♦ Possible recycling of nutrients |
| 11a) Forms of copper | ♦ Cellular toxicant, disruption of membrane transport<br>♦ Applied as wide variety of liquid or granular formulations | ♦ Effective and rapid control of many algae species<br>♦ Approved for use in most water supplies | ♦ Possible toxicity to aquatic fauna<br>♦ Accumulation of copper in system<br>♦ Resistance by certain green and blue-green nuisance species<br>♦ Lysing of cells releases nutrients and toxins |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 11b) Peroxides | ♦ Disrupts most cellular functions, tends to attack membranes<br>♦ Applied as a liquid or solid.<br>♦ Typically requires application at least once/yr, often more frequently | ♦ Rapid action<br>♦ Oxidizes cell contents, may limit oxygen demand and toxicity | ♦ Much more expensive than copper<br>♦ Limited track record<br>♦ Possible recycling of nutrients |
| 11c) Synthetic organic algaecides | ♦ Absorbed or membrane-active chemicals which disrupt metabolism<br>♦ Causes structural deterioration | ♦ Used where copper is ineffective<br>♦ Limited toxicity to fish at recommended dosages<br>♦ Rapid action | ♦ Non-selective in treated area<br>♦ Toxic to aquatic fauna (varying degrees by formulation)<br>♦ Time delays on water use |
| 12) Phosphorus inactivation | ♦ Typically salts of aluminum, iron or calcium are added to the lake, as liquid or powder<br>♦ Phosphorus in the treated water column is complexed and settled to the bottom of the lake<br>♦ Phosphorus in upper sediment layer is complexed, reducing release from sediment<br>♦ Permanence of binding varies by binder in relation to redox potential and pH | ♦ Can provide rapid, major decrease in phosphorus concentration in water column<br>♦ Can minimize release of phosphorus from sediment<br>♦ May remove other nutrients and contaminants as well as phosphorus<br>♦ Flexible with regard to depth of application and speed of improvement | ♦ Possible toxicity to fish and invertebrates, especially by aluminum at low pH<br>♦ Possible release of phosphorus under anoxia or extreme pH<br>♦ May cause fluctuations in water chemistry, especially pH, during treatment<br>♦ Possible resuspension of floc in shallow areas<br>♦ Adds to bottom sediment, but typically an insignificant amount |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 13) Sediment oxidation | ♦ Addition of oxidants, binders and pH adjustors to oxidize sediment<br>♦ Binding of phosphorus is enhanced<br>♦ Denitrification is stimulated | ♦ Can reduce phosphorus supply to algae<br>♦ Can alter N:P ratios in water column<br>♦ May decrease sediment oxygen demand | ♦ Possible impacts on benthic biota<br>♦ Longevity of effects not well known<br>♦ Possible source of nitrogen for blue-green algae |
| 14) Settling agents | ♦ Closely aligned with phosphorus inactivation, but can be used to reduce algae directly too<br>♦ Lime, alum or polymers applied, usually as a liquid or slurry<br>♦ Creates a floc with algae and other suspended particles<br>♦ Floc settles to bottom of lake<br>♦ Re-application typically necessary at least once/yr | ♦ Removes algae and increases water clarity without lysing most cells<br>♦ Reduces nutrient recycling if floc sufficient<br>♦ Removes non-algal particles as well as algae<br>♦ May reduce dissolved phosphorus levels at the same time | ♦ Possible impacts on aquatic fauna<br>♦ Possible fluctuations in water chemistry during treatment<br>♦ Resuspension of floc possible in shallow, well-mixed waters<br>♦ Promotes increased sediment accumulation |
| 15) Selective nutrient addition | ♦ Ratio of nutrients changed by additions of selected nutrients<br>♦ Addition of non-limiting nutrients can change composition of algal community<br>♦ Processes such as settling and grazing can then reduce algal biomass | ♦ Can reduce algal levels where control of limiting nutrient not feasible<br>♦ Can promote non-nuisance forms of algae<br>♦ Can improve productivity of system without increased standing crop of algae | ♦ May result in greater algal abundance through uncertain biological response<br>♦ May require frequent application to maintain desired ratios<br>♦ Possible downstream effects |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| **IN-LAKE BIOLOGICAL CONTROLS** | | | |
| 16) Enhanced grazing | ♦ Manipulation of biological components of system to achieve grazing control over algae<br>♦ Typically involves alteration of fish community to promote growth of grazing zooplankton | ♦ May increase water clarity by changes in algal biomass or cell size without reduction of nutrient levels<br>♦ Can convert unwanted algae into fish<br>♦ Harnesses natural processes | ♦ May involve introduction of exotic species<br>♦ Effects may not be controllable or lasting<br>♦ May foster shifts in algal composition to even less desirable forms |
| 16.a) Herbivorous fish | ♦ Stocking of fish that eat algae | ♦ Converts algae directly into potentially harvestable fish<br>♦ Grazing pressure can be adjusted through stocking rate | ♦ Typically requires introduction of non-native species<br>♦ Difficult to control over long term<br>♦ Smaller algal forms may be benefited and bloom |
| 16.b) Herbivorous zooplankton | ♦ Reduction in planktivorous fish to promote grazing pressure by zooplankton<br>♦ May involve stocking piscivores or removing planktivores<br>♦ May also involve stocking zooplankton or establishing refugia | ♦ Converts algae indirectly into harvestable fish<br>♦ Zooplankton response to increasing algae can be rapid<br>♦ May be accomplished without introduction of non-native species<br>♦ Generally compatible with most fishery management goals | ♦ Highly variable response expected; temporal and spatial variability may be high<br>♦ Requires careful monitoring and management action on 1-5 yr basis<br>♦ Larger or toxic algal forms may be benefitted and bloom |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 17) Bottom-feeding fish removal | ♦ Removes fish that browse among bottom deposits, releasing nutrients to the water column by physical agitation and excretion | ♦ Reduces turbidity and nutrient additions from this source<br>♦ May restructure fish community in more desirable manner | ♦ Targeted fish species are difficult to control<br>♦ Reduction in fish populations valued by some lake users (human/non-human) |
| 18) Microbial competition | ♦ Addition of microbes, often with oxygenation, can tie up nutrients and limit algal growth<br>♦ Tends to control N more than P | ♦ Shifts nutrient use to organisms that do not form scums or impair uses to same extent as algae<br>♦ Harnesses natural processes<br>♦ May decrease sediment | ♦ Minimal scientific evaluation<br>♦ N control may still favor cyanobacteria<br>♦ May need aeration system to get acceptable results |
| 19) Pathogens | ♦ Addition of inoculum to initiate attack on algal cells<br>♦ May involve fungi, bacteria or viruses | ♦ May create lakewide "epidemic" and reduction of algal biomass<br>♦ May provide sustained control through cycles<br>♦ Can be highly specific to algal group or genera | ♦ Largely experimental approach at this time<br>♦ May promote resistant nuisance forms<br>♦ May cause high oxygen demand or release of toxins by lysed algal cells<br>♦ Effects on non-target organisms uncertain |
| 20) Competition and allelopathy by plants | ♦ Plants may tie up sufficient nutrients to limit algal growth<br>♦ Plants may create a light limitation on algal growth<br>♦ Chemical inhibition of algae may occur through substances released by other organisms | ♦ Harnesses power of natural biological interactions<br>♦ May provide responsive and prolonged control | ♦ Some algal forms appear resistant<br>♦ Use of plants may lead to problems with vascular plants<br>♦ Use of plant material may cause depression of oxygen levels |

| OPTION | MODE OF ACTION | ADVANTAGES | DISADVANTAGES |
|---|---|---|---|
| 20a) Plantings for nutrient control | ♦ Plant growths of sufficient density may limit algal access to nutrients<br>♦ Plants can exude allelopathic substances which inhibit algal growth<br>♦ Portable plant "pods" , floating islands, or other structures can be installed | ♦ Productivity and associated habitat value can remain high without algal blooms<br>♦ Can be managed to limit interference with recreation and provide habitat<br>♦ Wetland cells in or adjacent to the lake can minimize nutrient inputs | ♦ Vascular plants may achieve nuisance densities<br>♦ Vascular plant senescence may release nutrients and cause algal blooms<br>♦ The switch from algae to vascular plant domination of a lake may cause unexpected or undesirable changes |
| 20b) Plantings for light control | ♦ Plant species with floating leaves can shade out many algal growths at elevated densities | ♦ Vascular plants can be more easily harvested than most algae<br>♦ Many floating species provide waterfowl food | ♦ Floating plants can be a recreational nuisance<br>♦ Low surface mixing and atmospheric contact promote anoxia |
| 20c) Addition of barley straw | ♦ Input of barley straw can set off a series of chemical reactions which limit algal growth<br>♦ Release of allelopathic chemicals can kill algae<br>♦ Release of humic substances can bind phosphorus | ♦ Materials and application are relatively inexpensive<br>♦ Decline in algal abundance is more gradual than with algaecides, limiting oxygen demand and the release of cell contents | ♦ Success appears linked to uncertain and potentially uncontrollable water chemistry factors<br>♦ Depression of oxygen levels may result<br>♦ Water chemistry may be altered in other ways unsuitable for non-target organisms |

Running through the list of options in Table 14, some techniques are less applicable than others. Dilution or flushing would require a tremendous amount of water with no obvious source; detention time must be kept under 3 weeks for flushing to work, and a supply of very clean water for dilution is absent. Additionally, internal load would be expected to overcome any dilution effect.

Drawdown is not feasible in this system. Surface covers would greatly restrict use of the lake, and dyes are permitted like algaecides; concentrations high enough to make a difference would not likely be allowed to flow downstream, and surface cyanobacteria scums may not be prevented. Mechanical removal is not practical for microscopic algae. There is a portable diffused air flotation unit in development, and it might allow removal of microscopic algae, but no practical demonstration has yet occurred. Sonication works on some algae but not others, and a large number of units would be needed to maintain control. Although solar options exist, these units are normally run by direct electricity and running power for all those units in the lake would be a major issue.

Sediment oxygenation could be beneficial, but this has not been a practiced technique for many years following field trials in the 1980s. Likewise, addition of settling agents for algae could work, but is not nearly as beneficial as aluminum, which inactivates phosphorus as well as settling algae. Additionally, adding a large mass of algae to the bottom would increase oxygen demand and may accelerate internal loading.

Selective nutrient addition is a fascinating area of study. It is possible to prolong a spring diatom dominated assemblage into summer by adding silica, and addition of nitrate could prevent cyanobacteria from becoming dominant. While having other algae dominant would represent an improvement and may help the food web in the lake, there would still be algae blooms and clarity would be low. Better options exist.

Removal of bottom feeding fish may provide some benefits, but would not impact internal or external loading to a great degree in this case. Microbial competition is a controversial discipline with literally no peer reviewed papers upon which to base rational conclusions. There are success stories, but there are also major failures. Enhanced microbial action may digest more organic sediment, and this generally requires oxygenation and some enzymes to break down certain hydrocarbons, but again there is no clear track record of success. Addition of pathogens has enjoyed two bursts of research over the last 4 decades, but no commercial products are currently available.

The use of plant or plant matter to discourage algae blooms is another interesting area of study. Decaying barley straw produces natural algaecides that seem to be most effective on cyanobacteria, but it is not a registered algaecidal product, so no commercial applicator can use it for that purpose. Abundant vascular plants, especially those with floating leaves, can restrict algae growth through shading, nutrient competition, or release of allelopathic substances, but problems with plant biomass would be every bit as troublesome as algae blooms.

This narrows the field of viable candidate management methods to:

1. Watershed management for reduced phosphorus input: This involves source controls and/or pollutant trapping that lowers the external load to the lake. The LLRM application for Middle and South Ponds suggests that even the most aggressive but feasible effort toward watershed management will not result in the proposed level of improvement in the ponds. This is partly a function of past successes in managing storm water and septic systems near these ponds, which limits the potential future reductions achievable through further watershed management. Continued emphasis on proper watershed management is encouraged, but the level of phosphorus reduction necessary to prevent blooms is not realistically achievable from watershed management alone.

   Watershed management should indeed be pursued, but will not be sufficient by itself. At the cost of needed improvements, local leadership will need to be selective and opportunistic to make any measureable difference in the incoming water quality. Certainly management of septic systems within about 300 feet of the ponds is warranted; most remaining septic systems are in the Connecticut portion of the watershed. Structural storm water drainage system improvements including detention, infiltration and various treatment chambers represent valuable protection for the ponds from developed or agricultural areas; many drainage areas have already been addressed, but there are more to be improved. However, watershed management represents protection of the ponds from further degradation, while in-lake management represents necessary loading reductions to restore the ponds to the desired condition.

   Protective watershed measures that are worthy of further pursuit include:

   - Installation of weir gates at the Berkshire Avenue Great Brook culvert to replace wooden boards that have been used for decades to prevent backflow of nutrient-laden water into Middle Pond. Until the drainage situation in Great Brook that allows backflow into Middle Pond is solved, this is a fairly simple protective measure for the pond.
   - Address remaining storm water outfalls leading to Congamond Lakes. The work done to date has helped, but there are more drainage systems in need of retrofitting with particle capture devices, increased detention and/or infiltration systems.
   - Continue work to educate property owners regarding the importance of reducing phosphorus inputs to the ponds. Proper disposal of yard waste, minimizing fertilizer use, and cleaning up aquatic vegetation that washed along shore are all relevant homeowner actions.
   - Evaluate farm runoff and encourage sound management practices. This is a delicate area, as farmers often operate with low profit margins and high risk, but there are programs that can help farmers minimize impact and even pay for improvements. Farm land is decreasing in the watershed, but remaining farms should be evaluated and runoff should be managed to the best possible level.
   - Enforce Town Bylaws, state regulations and federal mandates as relate to water pollution control. Particularly relevant are provisions of the National Pollutant Discharge Elimination System for runoff from land under development. Such parcels often contribute disproportionately to sediment loads and the laws are in place to control such degradation of streams and lakes.

2. Dredging: Removal of soft sediment would reduce the internal load and remove resting stages of both algae and rooted plants. On a large scale, dredging could set the lake back in time and greatly enhance conditions. The areas that need to be dredged for algae control do not completely overlap with areas in need of rooted plant control, and the cost would be very high to dredge adequately to make a difference. Focusing on algae control, the primary purpose of this assessment, and considering the results of the sediment testing, at least 10 cm (4 inches) of sediment would have to be removed (and probably much more) to substantially reduce internal loading.

   Over the 157 acre area over three ponds that experiences summer anoxia, that equates to a minimum of 83,000 cubic yards. At a low end cost of $30/cy, the cost would be at least $2.5 million, and it would not be surprising for the cost to be 3-5 times as much based on additional quantity or additional disposal cost. Just the necessary testing, engineering, and permitting for a dredging project can expected to cost in excess of $100,000 in a case like this. As depth recovery is not really an issue in deeper water, the cost does not justify the expense when other options exist.

   Dredging may well be desirable for navigation and outflow in the canal leaving South Pond, and dredging Great Brook to reduce backflow to Middle Pond seems desirable, but these actions are not strongly related to controlling algae in the Congamond Lakes. Likewise, dredging shallower areas of the ponds to control rooted plant growth could be advantageous, and would also at least temporarily remove algae spores from targeted shallow areas, but primary control of algae blooms in this case requires action in deeper water where dredging would be unlikely to occur. Any dredging of the outlet canal and shallow areas would not interfere with algae control options directed at deep water and is encouraged for the overall health of the ponds, but shallow water dredging will not ultimately control algae blooms in this lake system.

3. Enhanced grazing on algae: At the levels of phosphorus observed in the Congamond Lakes, restructuring the biological community to maximize consumption of algae does have merit. The key is to promote the largest possible population of a large bodied Daphnia, a grazing zooplankter that can clear the water of most algae. Some of the cyanobacteria will be difficult to graze due to particle size and possible toxicity, but encouraging Daphnia is appropriate. However, Daphnia appear to be minimal in these ponds and concerted management of the fish community would be needed to foster a larger population.

   Enhancement would involve altering the fish community to favor Daphnia and patiently awaiting a response. Stocking gamefish or conducting a laborious panfish removal project would be necessary, both at substantial cost and with no guarantee of results. Biological approaches tend to carry substantial variability and require vigilance and maintenance to provide continued benefits. While improvement of the fish community with a possible beneficial cascading effect on zooplankton is highly desirable, it might be better to control algae more directly and monitor for natural improvement of biological structure before attempting manipulations.

4. Algaecides: Application of algaecides can be effective and despite potential non-target impacts, they remain a valuable tool for algae control. Even the best management for nutrient control may be inadequate on occasion, as with catastrophic events (floods or fire) or the impact of climate change (very warm summer periods). In such cases a well-timed algaecide treatment can be effective at reasonable cost. The main issue is timing; monitoring must be adequate to detect the early stages of a bloom, so that algaecide treatment prevents a bloom, not destroys it once formed. The 2015 treatment was very successful due to a combination of vigilance and rapid response. Maintaining that level of responsiveness can be difficult.

   Killing off a major bloom carries risks of higher oxygen demand, toxin release at dangerous levels, and other water quality impacts that have resulted in regulation of cyanobacteria treatment in Massachusetts. Copper and peroxide are the main algaecides in use, with copper less expensive. Having this option available is worthwhile, but it should not be the first line of defense in the Congamond Lakes.

5. Circulation: Keeping the water in motion from top to bottom in a lake helps satisfy the oxygen demand, can limit nutrient release from sediment, and may disrupt the growth of some algae, notably the buoyant cyanobacteria that cause many of the blooms in the Congamond Lakes. There are several limitations to this approach, however. It is difficult to mix to the sediment-water interface without stirring up sediment, so a small anoxic zone may persist and allow release of phosphorus that may then be moved upward by the circulation system and aid algae growth. It is hard to overcome the input of heat from the sun during a prolonged summer dry spell, so circulation systems must either be greatly overpowered to handle all situations or will fail during those dry spells. Whatever nutrients are in the lake will be continually mixed and made available to algae, and with continued watershed loading, this may be enough to promote blooms even if the internal load is reduced.

   If the water can be mixed to a depth of at least 30 feet (9 m), the time spent in darkness may reduce growth, but much of North and Middle Ponds is not 30 feet deep, and none of South Pond is that deep. Loss of the hypolimnion through mixing may have ecological consequences, although the current lack of oxygen during summer minimizes the habitat value of this zone in Middle Pond. Circulation may be a viable strategy for limiting cyanobacteria in South Pond, but will probably not prevent other algae from blooming without some form of internal load control. With internal load control, it is not certain that any other in-lake management will be needed.

   The capital cost is not minor (expect up to $2000 per acre addressed, with 157 acres to be considered among 3 ponds), and there is an ongoing operational cost for power and maintenance that is not trivial (expect $100-$200 per acre per year). While potentially beneficial, this approach has too many drawbacks in this case to be recommended.

6. Oxygenation: Adding oxygen to deep water is not always necessary, but almost always provides water quality benefits. As long as anoxia can be prevented, release of iron-bound phosphorus can be minimized and a variety of other undesirable interactions between sediment and water can be limited. Certainly the fishery of each pond would be benefitted.

Aside from circulation to distribute oxygenated surface waters to near the bottom, there are four ways to add oxygen directly to target waters without breaking stratification. The most efficient method, diffused oxygen, is best applied to a hypolimnion of at least 20 feet (6 m) in thickness to allow enough vertical distance for the oxygen to be absorbed and avoid destratifying the lake. None of the ponds have a hypolimnion with an average thickness of 20 feet. While benefits could be expected, the ponds would probably be destratified by diffused oxygen injection.

The use of hypolimnetic aeration chambers, which circulate air in a chamber to oxygenate water that is then released back into the hypolimnion, can be effective but is very inefficient; oxygen transfer from air is slow, so a lot of it never gets transferred and more water and air have to be moved to achieve the desired results at substantial power cost.

Speece cones use pure oxygen in a submerged chamber, increasing efficiency at a large capital cost, but would be applicable. The main issue would be creating a platform for the equipment on the very soft sediment in deep water; some dredging may be necessary, greatly increasing cost and permitting delay.

Most intriguing is the sidestream supersaturation approach, in which water is pumped from the lake to a land-based pressurized chamber where pure oxygen is add to get a supersaturated solution that is put back into the hypolimnion. Less water has to be moved, although the cost is still substantial. The sidestream supersaturation approach appears most appropriate for the Congamond Lakes, although diffused oxygen would probably be least expensive if some amount of destratification was acceptable.

7. Phosphorus inactivation: Inactivation of phosphorus has been practiced in New England for over 30 years, and this technique has been refined and advanced considerably over those decades. Presentations as recently as November 2015 at the NALMS conference broke new ground on understanding key processes and advancing treatment effectiveness, so those considering phosphorus inactivation need to be up to date on the associated science.

   There are 3 ways to use phosphorus inactivation:

   - Treatment of surficial sediment with larger doses to prevent release of phosphorus, mainly from iron subjected to anoxia
   - Treatment of the water column with lower doses to inactivate and settle phosphorus and algae
   - Treatment of inflows with lower doses, mainly during storms, to inactivate incoming phosphorus and settle associated solids

   Sediment treatments can involve calcium, aluminum or lanthanum, with aluminum having the longest and best documented track record. Sediment treatments in deeper lakes tend to provide benefits for about 20 years, as iron-bound phosphorus is effectively neutralized and that is the main source of phosphorus through internal loading. Termination of benefits is largely linked to upward migration of iron-bound phosphorus through the inactivation zone (usually the upper 4-10 cm of sediment).

One alternative to aluminum that may be worth investigating is Phoslock, which is a mix of bentonite clay and lanthanum. The lanthanum binds well with phosphorus and the clay helps seal the surficial sediment. While the track record is too short to make a firm prediction, the potential to both reduce phosphorus availability and lower oxygen demand by covering the organic sediment with a thin clay layer is appealing. It seems unlikely that the normal dose will be adequate to cover the soft sediment, and any dose may eventually sink into that very loose sediment, but the company that markets Phoslock could conduct tests at limited cost. However, the cost of Phoslock is several times that of aluminum application.

Application of lower doses of aluminum to the water column can reduce available phosphorus and at least temporarily lower algae biomass. Water column treatment has been used elsewhere with varied but temporary results. This is entirely consistent with the assumed mode of internal loading, whereby a very large supply of sediment phosphorus can be released and replace inactivated water column phosphorus within the same summer season. Higher doses or more frequent treatments may provide better or longer lived results; certainly low dose treatments would fare better if the sediment was first treated with a larger dose, but that one large dose to inactivate phosphorus in surficial sediment may be all that is needed in the Congamond Lakes.

Inactivation of incoming storm water phosphorus involves a dosing station delivering an approximate dose whenever it rains. This can be done manually or automatically at slightly higher cost, and has been very successful in many cases in Florida and at the only active installation in Massashusetts (Morses Pond in Wellesley). A dose of 1-3 mg/L is targeted, with pumps and chemical storage sized accordingly. Systems are commercially available or can be custom made at a cost on the order of $100,000 per station. Treatment is generally conducted in the spring to ensure that conditions are optimal going into summer, but continued summer treatment can be conducted if warranted. The problem with inflow dosing at the Congamond Lakes is the diffuse nature of that inflow. Many injection sites would be needed to have a substantial impact, with great cost and operational management need.

Application of a dose sufficient to inactivate iron-bound phosphorus in the surficial sediment would greatly reduce internal loading in the Congamond Lakes, which based on model results, could achieve the desired conditions in the lake with no other action.

Considering all the available options, there seems little doubt that inactivation of the internal phosphorus load is the most advantageous approach to improving Middle and South Ponds, and would also alleviate the less frequent blooms in North Pond. Certainly the cost of treating North Pond is small compared to the other two ponds, and would not add appreciably to a project involving Middle and South Ponds. It seems important to treat Middle Pond before South Pond, as South Pond receives a substantial portion of its phosphorus load from Middle Pond. At an estimated treatment area <200 acres, such a project could be completed in less than a month once funding and permits are in place.

## Phosphorus Inactivation Potential

Phosphorus inactivation involves the binding of phosphorus by added compounds that make it unavailable for uptake by algae. Treatment of incoming water, lake water or lake sediment is possible and applicable to Congamond Lakes, but the primary need appears to be inactivation of iron-bound phosphorus in surficial sediments that provides a substantial internal load. Treatment of incoming storm water or the lake water column could be considered, but each would involve repetitive treatments over years, while a single adequate dose to the sediment over an area of about 188 acres could minimize internal loading and control algae blooms for up to 20 years.

Aluminum has been the phosphorus binder of choice in Massachusetts for the last 30 years. Aluminum sulfate can be applied by itself where alkalinity is high, but in most cases in Massachusetts sodium aluminate is applied with the aluminum sulfate to keep the pH stable. Alkalinity ranges from 22 to 104 mg/L in the Congamond Lakes, is mostly between 30 and 50 mg/L, and averages 40 mg/L. Some buffering is likely to be needed at anticipated doses. Polyaluminum chloride is gaining popularity for inflow and water column treatments and requires no buffering at typical doses, but the combination of aluminum sulfate and sodium aluminate, applied separately but at the same time, is still the standard treatment mode for high aluminum doses. There is another binder that has been used in recent years called Phoslock, which is bentonite clay modified with lanthanum. It may do a superior job capturing phosphorus from the water column, and may or may not do as good a job on surficial sediments, but at a cost of about five times the cost of an appropriate aluminum dose. Given the cost factor and experience with aluminum, dosing sediments with aluminum is recommended.

As noted above, aluminum has been used in Massachusetts for over 30 years. It is not a new approach for the Congamond Lakes either, as some portion of this system was treated in 1989. Treatment approaches were primitive by today's standards, and dose determination was very rudimentary. About 30,500 gallons of aluminum sulfate and 17,200 gallons of sodium aluminate were applied from a tank mounted on a barge (Figure 29), translating into 16.5 million grams of aluminum. The exact treatment area does not appear to be available, but if it covered just the anoxic areas in Middle and South Ponds, the dose would have been about 27 $g/m^2$. This would be a low dose compared to the estimated current need, and may explain why the results did not last as long as hoped. If a smaller area was treated at a larger dose, substantial contributory area would have been left untreated, potentially limiting even the initial results.

Modern treatment uses a variety of delivery vessels designed for that purpose, and chemicals are precisely metered to provide precise ratios of aluminum sulfate to sodium aluminate and tight pH control as well as accurate dosing. Barges typically traverse GPS guided paths for accurate delivery of aluminum to target areas and floc formation and related water chemistry are carefully monitored (Figure 30). Dose determination is made from sediment tests and confirmed with laboratory assays. Aluminum treatments may not be an exact science, but they are fairly reliable and have produced excellent results when properly conducted on appropriate subject lakes.

Successful aluminum treatment is a function of supplying an adequate dose to the appropriate treatment area. It is generally acknowledged that the targeted treatment area should be at least the area of sediment that can experience anoxia, which facilitates the release of phosphorus bound by iron (Fe-P). Treating a slightly larger area where algae may grow on the surficial sediment then float upward in response to change in light and/or temperature is often advisable. The necessary dose is a matter of both the Fe-P concentration and other sediment constituents that may compete with Fe-P for binding sites on the applied aluminum compounds. This is an area of current study that has some degree of uncertainty attached to it.

The aluminum to phosphorus ratio (Al:P) necessary for effective inactivation varies inversely with Fe-P concentration, as lower Fe-P levels mean that other constituents are abundant and compete for binding sites (James and Bischoff 2015). When Fe-P is high it tends to occupy more of the binding sites and the necessary ratio of Al to P increases as Fe-P declines. The range of Al:P ratios for successful treatments tends to range from 10 to 150, and the range of aluminum doses has been 10 to about 200 $g/m^2$, although treatments at >100 g/m2 have not been needed in MA and most Al:P ratios have been near the low end of the known range. The applied dose range in MA is 10 to 100 $g/m^2$, with the average close to 50 $g/m^2$. Treated lakes with higher Fe-P had a lower Al:P ratio, while those with lower Fe-P required a higher Al:P ratio.

Fe-P levels in the Congamond Lakes would be considered moderate (Table 4), with averages of 372 mg/kg in North Pond (ignoring the one lower value outside the expected treatment zone), 455 mg/kg in Middle Pond, and 472 mg/kg in South Pond. However, the solids content of the sediment varies, such that there is actually more Fe-P per square meter in North Pond (4.09 g) than in the other two ponds (3.45 g in Middle, 3.27 g in South). Functionally the Fe-P concentration in all three ponds would be considered similar.

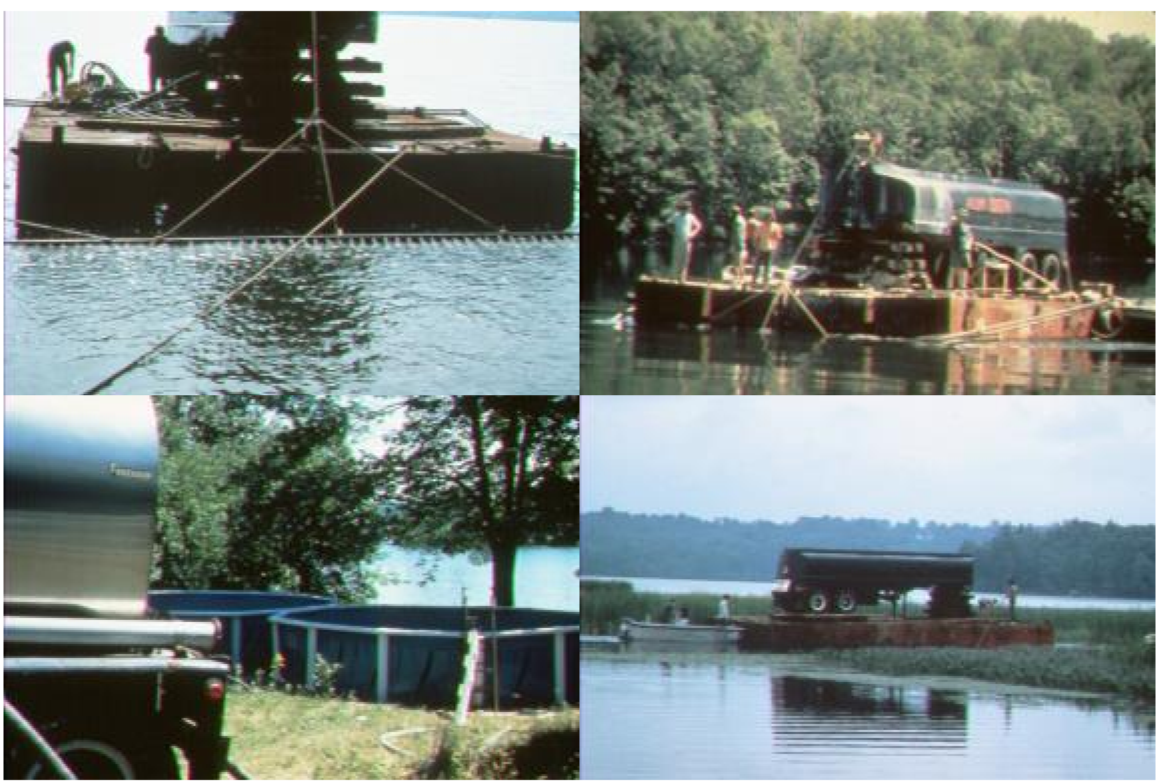


**Figure 29. Photographs from the 1989 Application of Aluminum to Congamond Lakes**

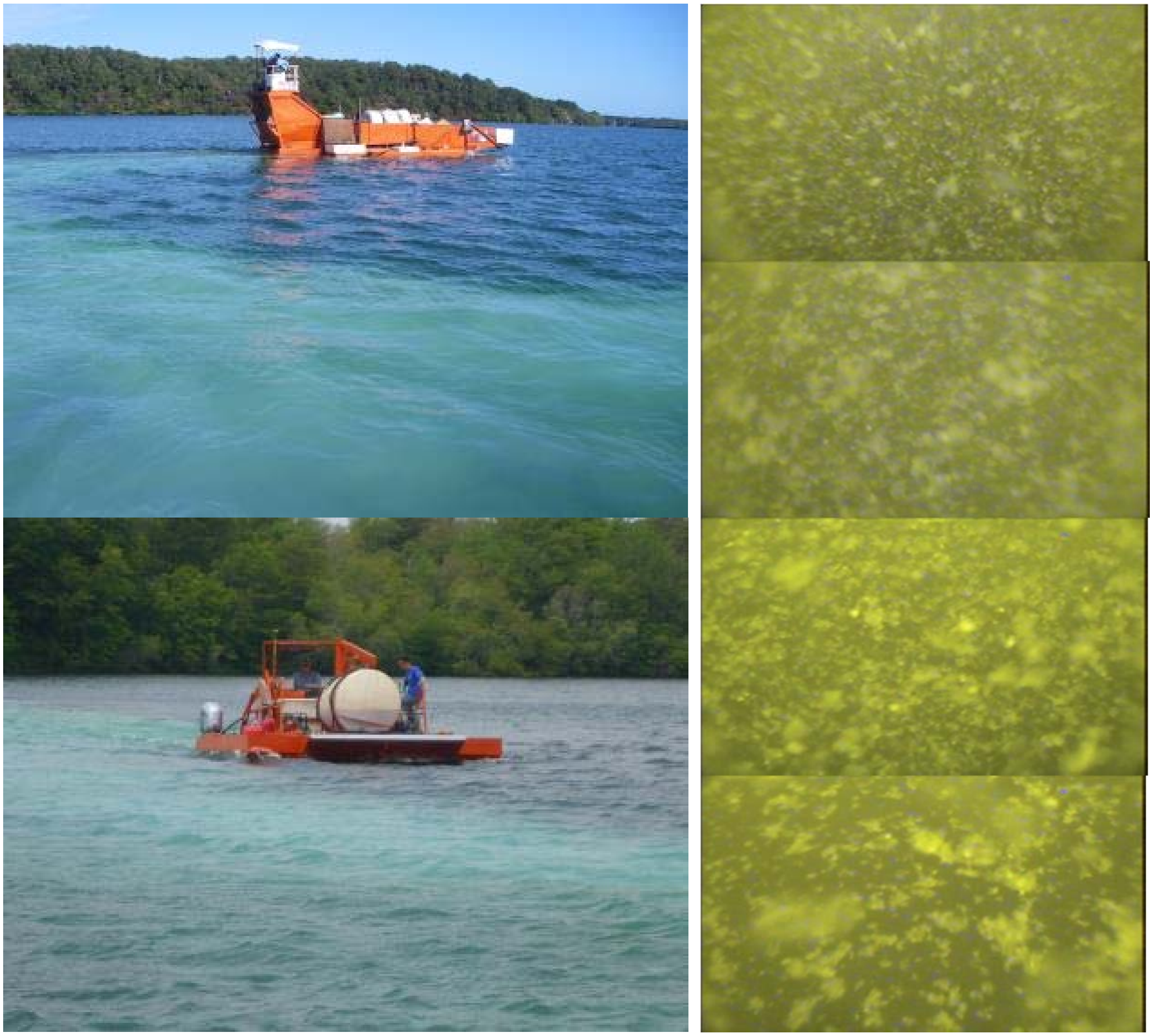


**Figure 30. One Current Approach to Application of Aluminum**
**(barge applying aluminum on the left, vertical gradient of floc formation on the right)**

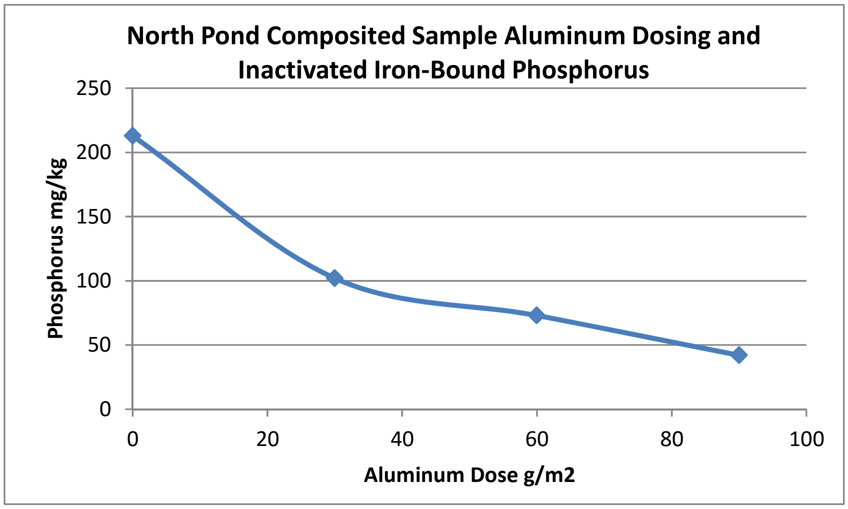
Based on equations from James and Bischoff (2015), the average Fe-P concentration suggests a 40:1 Al:P ratio, which would yield a dose of 131 $g/m^2$ in South Pond, 138 $g/m^2$ in Middle Pond, and 164 $g/m^2$ in North Pond, much higher than any MA treatment to date. Dose determination by stoichiometry calculations suggests doses of 65 to 82 $g/m^2$. The Al:P ratio is 20:1 in those calculations, toward the low end of the recommended range but typical for Massachusetts lakes.

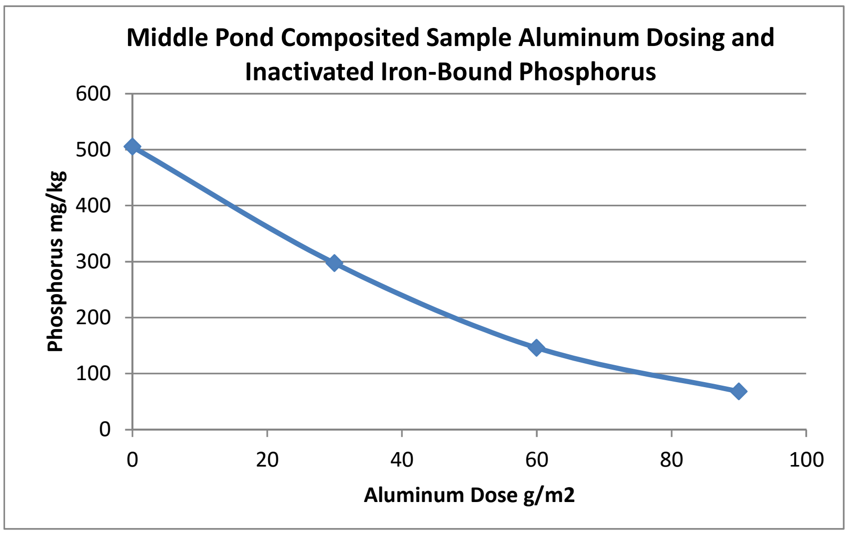
Out of concern over the variability and uncertainty of calculated doses, we developed a laboratory aluminum assay about a decade ago. In those assays, small amounts of sediment are treated with aluminum in the lab then retested for Fe-P. As the dose rises, remaining Fe-P declines, but almost never in a linear pattern. This allows visualization of how well Fe-P can be inactivated and the point at which diminishing returns drive costs to an intolerable level. The aluminum assays for a composite sample from each pond (Figure 31) provided useful insights. Phosphorus availability declined in a nearly linear fashion, rather than the more asymptotic pattern normally observed. There is not a clear inflection point beyond which benefits decline rapidly while needed aluminum (and associated cost) rises quickly. Ideally, we would like to drive Fe-P below 100 mg/kg and preferably below 50 mg/kg, but the reduction in internal load is roughly proportional to the percent reduction in Fe-P, so major reductions can be achieved even when Fe-P remains above the detection limit.

For North Pond, the curve crosses the 100 mg/kg mark at a dose of 30 g/$m^2$, representing a 73% decrease in the internal load. Run back through the LLRM, this level of reduction (instead of the 90% reduction assumed in the original model runs) produces an expected average phosphorus concentration of 17 µg/L, within the range considered to provide the desired conditions in the pond. For Middle Pond, the curve crosses the 100 mg/kg line at a dose of 75 g/$m^2$, representing a 78% reduction in internal loading. LLRM predicts a corresponding phosphorus concentration of 16 µg/L, an acceptable value. For South Pond, the curve crosses the 100 mg/kg threshold at 85 g/$m^2$, representing a 79% decrease in internal loading. Assuming Middle Pond is also treated, LLRM predicts a South Pond phosphorus level of 16 µg/L, an acceptable value.

Additional testing would be warranted prior to actual treatment, but this set of tests sets reasonable bounds for phosphorus inactivation with aluminum for the Congamond Lakes and facilitates further treatment planning and costing. While the laboratory aluminum assay is not a perfect replication of field conditions, these have proven accurate in other MA treatments. Also, treatments are believed to be additive; later application is not compromised by earlier application, so if more treatment is needed, it can be conducted later with no reduction in effectiveness. While it is preferable to treat once and be done with the process, overdosing raises costs without providing clear benefits. Extreme underdosing will not attain the desired benefits, but a dose slightly lower than optimal appears to only affect longevity of benefits, not the degree of initial improvement. Consequently, a lower dose with potential follow up treatment some years later becomes an attractive option when costs are high.

Given the relative condition of the three ponds, doses of 30, 75 and 85 g/$m^2$ are suggested for North, Middle and South Ponds respectively for planning purposes. Further testing prior to treatment can be used to adjust doses slightly, but it is highly unlikely that more than a large change would result. More sampling might suggest different doses in different zones, but the values obtained were fairly consistent within ponds. The dose for North Pond is substantially lower than suggested by some methods of calculation, but is consistent with its condition relative to Middle and South Ponds and the dose is consistent with other lakes in similar condition treated previously. The Middle and South Ponds doses are slightly higher than derived by the stoichiometric approach and lower than those obtained from the Al:P ratio method, but within the range applied in Massachusetts to date.



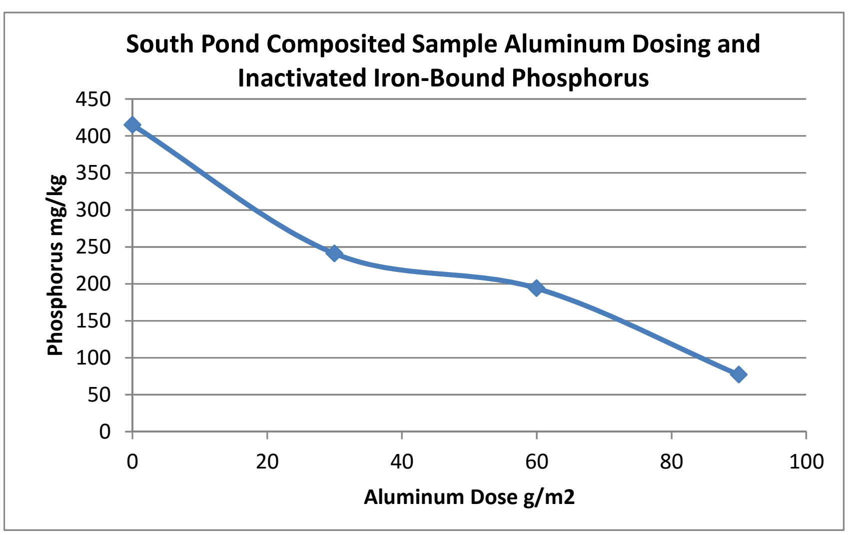




**Figure 31. Aluminum Assay Results for Congamond Lakes Sediment**

Diminishing benefits of treatment are a function of external loading, release of phosphorus from organic decay, and upward P migration through the treated sediment (Huser et al. 2016). External loading affects results for all lakes, and aluminum treatment is not advised when external loading is still the dominant P source. Conditions in the Congamond Lakes suggest that addressing the internal load is the right thing to do, and aluminum treatment should provide relief from algae blooms for many years. Certainly there is ongoing external loading and further reductions will help protect the lake, but the efforts of the last two decades suggest that much progress has been made and that further reductions will be both more difficult and expensive. Export coefficients for phosphorus for the defined drainage areas are near the low end for drainage areas with residential development and agricultural uses. Further, relatively long detention time suggests that any in-lake action will have more immediate benefits that will last for many years.

For unstratified lakes that mix frequently, organic phosphorus is an important source, although phosphorus released from Fe-P during intermittent periods of anoxia can also be an important phosphorus source. Such lakes tend to exhibit improved conditions for about 10 years after proper aluminum treatment. South Pond does not stratify strongly, but is more stratified with more prolonged anoxia at the sediment-water interface than lakes we would consider unstratified. Mixing in response to major storms or wind events is possible, but South Pond tends to behave more like a standard stratified lake. That stratification breaks down more easily and earlier in late summer than for the other two ponds, but appears to set up by late June and be fairly stable in July and August. Decay of organic matter will provide some phosphorus in South Pond, but Fe-P is likely to be the most critical source, especially during summer. Duration of benefits in excess of 10 years is therefore expected.

For lakes that more strongly stratify, the upward migration of Fe-P through the treatment zone is expected to be the dominant influence on duration of benefits. Duration of benefits has averaged about 20 years over a range of treatments in stratified lakes (Huser et al. 2016). This is more along the lines of the expected longevity of the proposed Congamond Lakes treatments. If the dose turns out to be lower than optimal, the initial results should still be quite impressive, but they would not last as long. Current thinking in aluminum treatments is not to overdose, but to monitor sediment cores to assess the rate at which Fe-P is migrating upward to predict when additional treatment might be needed. Additional sampling is advised as part of any aluminum treatment project. The suggested 75 and 85 $g/m^2$ treatment levels for Middle and South Ponds, respectively, represent above average doses for New England lakes where treatments have provided more than 15 years of benefits in the past.

The benefits of phosphorus inactivation should extend to oxygen improvements as well as a decrease in the available phosphorus and algae. Most treatments have reduced but not eliminated anoxia in the deeper water. The reduction in algae translates to less oxygen-demanding organic matter settling into deeper water, but the ongoing oxygen demand of the organic sediments is not appreciably reduced by the treatment, so a thinner anoxic layer that forms later in the summer is often observed.

Cost considerations cannot be ignored when planning aluminum treatments of this magnitude. It is suggested that an area of 10.2 acres be treated in North Pond, the area >5 m (16.5 feet) deep, to address algae growth on sediments as well as the area that is truly anoxic each summer. For Middle and South Ponds, treatment at about 5 m is also appropriate, translating into areas of 127.1 and 50.5 acres, respectively. This is a total of 187.8 acres of treatment area over three ponds, but the dose varies by pond. There are several ways to estimate cost, including summing itemized costs for mobilization, chemicals, application and monitoring and multiplying the dose by average cost per $g/m^2$ per acre treated and the number of targeted acres.

For North Pond, the range is $18,500 to $39,000, with the high end applicable if North Pond was done separately from the other ponds with all the mobilization and monitoring covered by just this small treatment. For Middle Pond, the cost ranges from $513,000 to $574,000. For South Pond the cost ranges from $238,000 to $258,000. Done as a single project, the maximum cost should be no greater than $850,000.

Multiple combinations of dose and treated area could be considered to stay within the limits of available funding. No less than 157 acres, the documented anoxic area in July and August, should be treated. At the suggested dose for each pond, this suggests a minimum treatment cost of about $723,000. Middle Pond should have the highest priority, followed by South Pond and then North Pond, but the North Pond treatment is small compared to the other two in terms of area and dose, so its cost is nominal by comparison.

## Recommendations

Based on the documented condition of the Congamond Lakes and current understanding of the factors contributing to that condition, the most important action that can be taken to improve these ponds is the reduction of internal loading of phosphorus from sediments exposed to anoxia. This can be accomplished by multiple means, but considering cost, flexibility of implementation, and overall expected effectiveness, it is recommended that a single treatment of surficial sediments with aluminum over an approximately 188 acre area below a water depth of 5 m (16.5 feet) be performed. The recommended dose is 30 $g/m^2$ over 10.2 acres in North Pond, 75 $g/m^2$ over 127.1 acres in Middle Pond, and 85 $g/m^2$ over 50.5 acres in South Pond (Figures 32-34). Such a treatment could be conducted at almost any time, but maximum effectiveness and most immediate improvement of the lake would be obtained with a spring treatment. This treatment should cost no more than $850,000. The duration of benefits is expected to be 15 to 20 years.

The proposed treatment is expected to reduce average phosphorus in the upper portion of the water column of each pond to about 16 µg/L, resulting in average water clarity >3.0 m (10 feet) and algal chlorophyll levels that exceed 10 µg/L less than 10% of the time. No severe cyanobacteria blooms should develop, and conditions are likely to be better than predicted since more of the internal load targeted for inactivation may be actively used by algae than presumed in the loading analysis. While interested parties are encouraged to discuss goals for the ponds and translate those into quantifiable objectives, discussion to date suggests that the proposed program will establish conditions acceptable to all users.

Figure 32. Proposed treatment area in North Pond

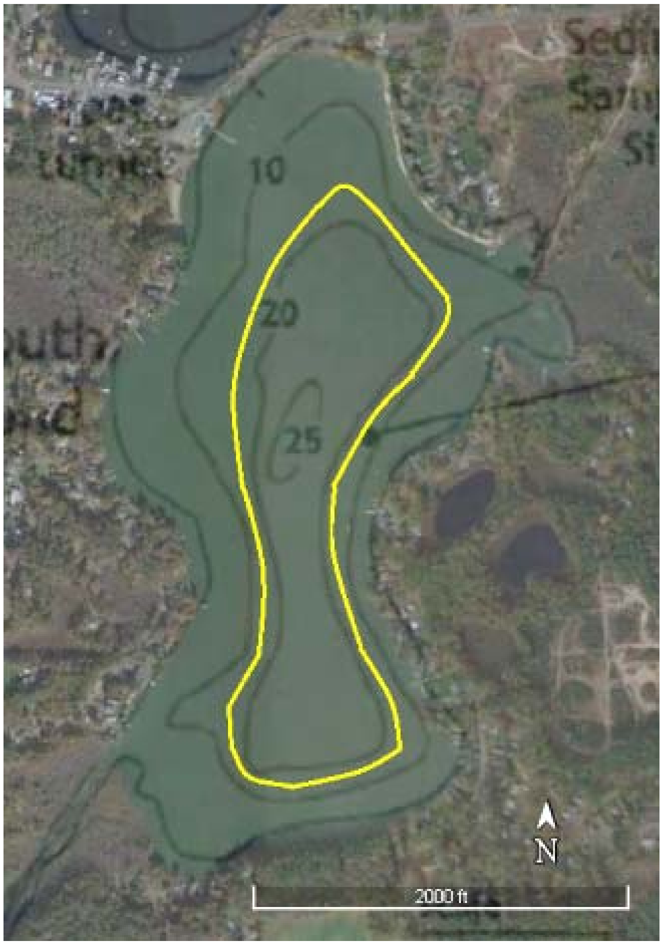


Figure 33. Proposed treatment area in South Pond

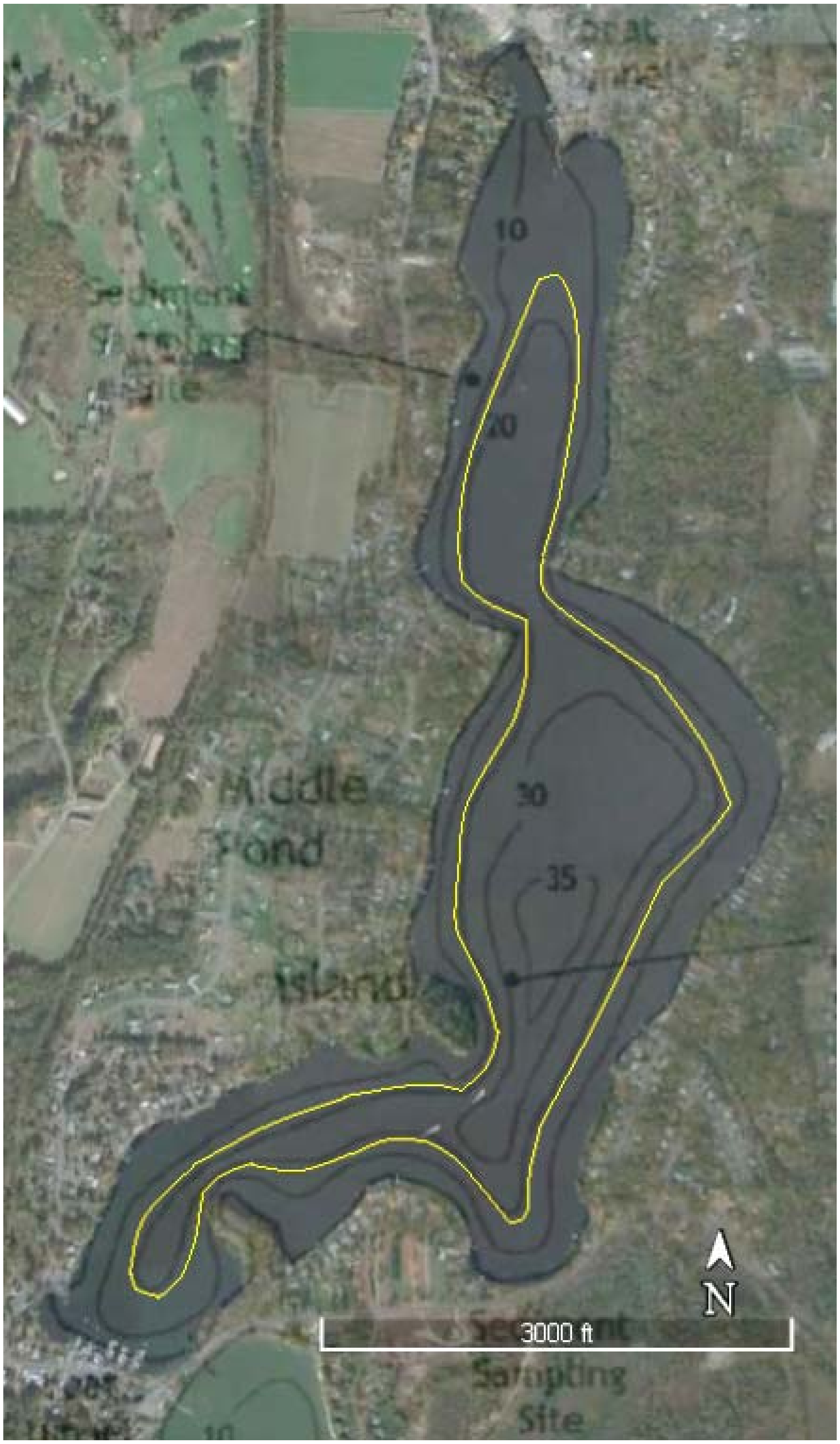



**Figure 34. Proposed treatment area in Middle Pond**

Oxygen in deep water should increase, although some anoxia is still expected after about mid-summer until fall turnover in the deepest areas. Recreational lake use and suitability for fish and wildlife habitat will be greatly enhanced. The increased clarity is likely to promote increased rooted plant growth, and plans should be made to apply physical or chemical means to address potential plant nuisances.

Additional watershed management efforts are warranted to limit external inputs, mainly as protection for the investment made in in-lake management. Suggested watershed activities encompass source control and pollutant trapping, and specifically include installation of weir gates on Great Brook to prevent backflow into Middle Pond during storms, pollution control retrofits to existing storm water drainage systems not yet addressed, education of property owners regarding management methods to limit nutrient inputs to the ponds, farm runoff management, and enforcement of laws relating to runoff controls for developing land.

Long planned dredging of the outlet canal and shallow areas will aid overall lake health and is encouraged, but dredging in shallow water areas will not provide the level of control needed; the proposed phosphorus inactivation program remains the most reliable means to minimize algae blooms in the Congamond Lakes.

To document project success and provide data to aid additional management planning, a monitoring program such as that conducted in 2015 in support of this management planning exercise should be instituted. Aside from funding limitations, lack of data is the greatest impediment to ongoing management.

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# APPENDIX

# DATA AND SUPPLEMENTAL INFORMATION

# Available Nutrient Data for Congamond Lakes

| Parameters | Station | 8/20/81 | 10/5/81 | 10/15/81 | 2/20/82 | 4/15/82 | 5/7/82 | 6/15/82 | 6/30/82 | 7/15/82 | 7/30/82 | 8/15/82 | 8/30/82 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Ammonium N (mg/L) | North Pond Top | 0.230 | 0.080 | 0.130 | 0.280 | 0.100 | 0.640 | 0.500 | 1.090 | 0.380 | 0.280 | 0.020 | 0.200 |
| Ammonium N (mg/L) | North Pond Bottom | 4.000 | | 1.960 | 0.740 | 0.300 | 1.050 | 0.620 | 1.500 | 0.500 | 0.020 | 0.580 | < 0.050 |
| Ammonium N (mg/L) | Middle Pond Top | 0.350 | 0.500 | 1.310 | | 0.360 | 0.440 | 1.380 | 1.420 | 1.400 | | | 0.070 |
| Ammonium N (mg/L) | Middle Pond Bottom | 2.150 | | 1.970 | 0.280 | 0.290 | 0.600 | 1.340 | 1.560 | 0.160 | 0.620 | 0.920 | 0.070 |
| Ammonium N (mg/L) | South Pond Top | 3.750 | 0.120 | 0.700 | | 0.120 | 0.530 | 0.960 | 1.760 | 0.620 | 0.520 | 0.240 | < 0.050 |
| Ammonium N (mg/L) | South Pond Bottom | 0.350 | | 0.900 | 0.500 | 0.200 | 0.580 | 0.830 | 1.640 | 0.380 | 0.620 | 0.300 | < 0.050 |
| Nitrate N (mg/L) | North Pond Top | < 0.050 | 0.150 | 0.050 | 0.47 | 0.120 | 0.060 | 0.060 | 0.100 | < 0.050 | 0.120 | < 0.050 | 0.020 |
| Nitrate N (mg/L) | North Pond Bottom | 3.360 | | 0.170 | 0.520 | 0.280 | 0.390 | 0.430 | 0.130 | | 0.260 | < 0.050 | 0.470 |
| Nitrate N (mg/L) | Middle Pond Top | < 0.050 | 0.080 | 0.070 | 3.040 | 0.560 | 0.740 | < 0.050 | 0.210 | 0.040 | | | 0.020 |
| Nitrate N (mg/L) | Middle Pond Bottom | 0.980 | | 0.210 | 1.160 | 0.910 | 0.100 | 0.170 | 0.150 | 1.640 | 0.170 | < 0.050 | 0.370 |
| Nitrate N (mg/L) | South Pond Top | < 0.050 | 0.080 | 0.070 | 2.130 | 0.210 | 0.340 | 0.040 | 0.100 | < 0.050 | 0.100 | < 0.050 | 0.210 |
| Nitrate N (mg/L) | South Pond Bottom | < 0.050 | | 0.300 | 0.170 | 0.750 | 0.130 | < 0.050 | 0.210 | 0.910 | 0.210 | 0.080 | 0.650 |
| Organic N (mg/L) | North Pond Top | | | | | | | | | | | | |
| Organic N (mg/L) | North Pond Bottom | | | | | | | | | | | | |
| Organic N (mg/L) | Middle Pond Top | | | | | | | | | | | | |
| Organic N (mg/L) | Middle Pond Bottom | | | | | | | | | | | | |
| Organic N (mg/L) | South Pond Top | | | | | | | | | | | | |
| Organic N (mg/L) | South Pond Bottom | | | | | | | | | | | | |
| TKN (mg/L) | North Pond Top | | | | | | | | | | | | |
| TKN (mg/L) | North Pond Bottom | | | | | | | | | | | | |
| TKN (mg/L) | Middle Pond Top | | | | | | | | | | | | |
| TKN (mg/L) | Middle Pond Bottom | | | | | | | | | | | | |
| TKN (mg/L) | South Pond Top | | | | | | | | | | | | |
| TKN (mg/L) | South Pond Bottom | | | | | | | | | | | | |
| Total Nitrogen (mg/L) | North Pond Top | | | | | | | | | | | | |
| Total Nitrogen (mg/L) | North Pond Bottom | | | | | | | | | | | | |
| Total Nitrogen (mg/L) | Middle Pond Top | | | | | | | | | | | | |
| Total Nitrogen (mg/L) | Middle Pond Bottom | | | | | | | | | | | | |
| Total Nitrogen (mg/L) | South Pond Top | | | | | | | | | | | | |
| Total Nitrogen (mg/L) | South Pond Bottom | | | | | | | | | | | | |
| Total Phosphorus (mg/L) | North Pond Top | 0.020 | 0.020 | 0.021 | 0.010 | 0.030 | 0.020 | 0.050 | 0.030 | 0.020 | 0.010 | 0.040 | 0.020 |
| Total Phosphorus (mg/L) | North Pond Bottom | 0.070 | | 0.090 | | 0.280 | 0.040 | 0.040 | 0.040 | | 0.180 | 0.090 | 0.030 |
| Total Phosphorus (mg/L) | Middle Pond Top | 0.020 | 0.040 | 0.090 | | 0.010 | 0.080 | 0.050 | 0.030 | 0.040 | | | 0.030 |
| Total Phosphorus (mg/L) | Middle Pond Bottom | 0.030 | | 0.210 | 0.040 | < 0.010 | 0.100 | 0.070 | 0.030 | 0.140 | 0.050 | 0.070 | 0.030 |
| Total Phosphorus (mg/L) | South Pond Top | 0.030 | 0.070 | 0.090 | | 0.010 | 0.080 | 0.040 | 0.030 | 0.030 | 0.010 | 0.030 | 0.040 |
| Total Phosphorus (mg/L) | South Pond Bottom | 0.020 | | 0.210 | 0.040 | < 0.010 | 0.070 | 0.040 | 0.450 | 0.310 | 0.040 | 0.100 | 0.080 |
| Dissolved Phosphorus (mg/L) | North Pond Top | | | | | | | | | | | | |
| Dissolved Phosphorus (mg/L) | North Pond Bottom | | | | | | | | | | | | |
| Dissolved Phosphorus (mg/L) | Middle Pond Top | | | | | | | | | | | | |
| Dissolved Phosphorus (mg/L) | Middle Pond Bottom | | | | | | | | | | | | |
| Dissolved Phosphorus (mg/L) | South Pond Top | | | | | | | | | | | | |
| Dissolved Phosphorus (mg/L) | South Pond Bottom | | | | | | | | | | | | |

| | | | | | | | | | | | All Available Data | | | | 2015 Data | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Parameters | Station | 9/9/09 | 10/14/09 | 7/24/13 | 9/3/13 | 5/20/15 | 6/19/15 | 7/16/15 | 8/18/15 | 9/18/15 | 10/14/15 | MIN | MAX | AVERAGE | MEDIAN | MIN | MAX | AVERAGE | MEDIAN |
| Ammonium N (mg/L) | North Pond Top | 0.017 | 0.016 | | | < 0.010 | 0.062 | 0.083 | 0.057 | 0.087 | 0.140 | 0.010 | 1.090 | 0.220 | 0.115 | 0.010 | 0.140 | 0.073 | 0.073 |
| Ammonium N (mg/L) | North Pond Bottom | 0.600 | 0.850 | | 0.005 | 0.061 | 0.061 | 0.320 | 1.100 | 0.120 | 0.870 | 0.005 | 4.000 | 0.765 | 0.590 | 0.061 | 1.100 | 0.422 | 0.220 |
| Ammonium N (mg/L) | Middle Pond Top | 0.285 | 0.212 | | 0.005 | 0.092 | 0.170 | 0.100 | < 0.050 | 0.079 | 0.100 | 0.005 | 1.420 | 0.462 | 0.249 | 0.050 | 0.170 | 0.099 | 0.096 |
| Ammonium N (mg/L) | Middle Pond Bottom | 1.930 | 2.900 | | 1.196 | 0.560 | 1.200 | 2.200 | 2.700 | 2.400 | 2.600 | 0.070 | 2.900 | 1.382 | 1.270 | 0.560 | 2.700 | 1.943 | 2.300 |
| Ammonium N (mg/L) | South Pond Top | 0.050 | 0.198 | | 0.009 | 0.095 | 0.085 | 0.052 | < 0.050 | 0.084 | 0.099 | 0.009 | 3.750 | 0.505 | 0.120 | 0.050 | 0.099 | 0.078 | 0.085 |
| Ammonium N (mg/L) | South Pond Bottom | 4.160 | 0.202 | | 0.010 | 0.160 | 0.820 | 0.140 | 0.200 | 0.081 | 0.130 | 0.010 | 4.160 | 0.613 | 0.325 | 0.081 | 0.820 | 0.255 | 0.150 |
| Nitrate N (mg/L) | North Pond Top | | | | 0.004 | < 0.010 | < 0.010 | < 0.010 | < 0.010 | 0.110 | < 0.050 | 0.004 | 0.470 | 0.079 | 0.050 | 0.010 | 0.110 | 0.033 | 0.010 |
| Nitrate N (mg/L) | North Pond Bottom | | | | 0.005 | < 0.010 | < 0.010 | < 0.010 | < 0.010 | 0.110 | < 0.050 | 0.005 | 3.360 | 0.369 | 0.130 | 0.010 | 0.110 | 0.033 | 0.010 |
| Nitrate N (mg/L) | Middle Pond Top | | | | 0.005 | 0.120 | < 0.050 | < 0.050 | < 0.050 | < 0.050 | < 0.050 | 0.005 | 3.040 | 0.308 | 0.050 | 0.050 | 0.120 | 0.062 | 0.050 |
| Nitrate N (mg/L) | Middle Pond Bottom | | | | 0.005 | 0.140 | < 0.050 | < 0.050 | < 0.050 | < 0.050 | < 0.050 | 0.005 | 1.640 | 0.350 | 0.145 | 0.050 | 0.140 | 0.065 | 0.050 |
| Nitrate N (mg/L) | South Pond Top | | | | 0.008 | 0.064 | < 0.050 | < 0.050 | < 0.050 | 0.130 | < 0.050 | 0.008 | 2.130 | 0.202 | 0.064 | 0.050 | 0.130 | 0.066 | 0.050 |
| Nitrate N (mg/L) | South Pond Bottom | | | | 0.007 | 0.160 | < 0.050 | < 0.260 | < 0.050 | 0.069 | < 0.050 | 0.007 | 0.910 | 0.231 | 0.145 | 0.050 | 0.260 | 0.107 | 0.060 |
| Organic N (mg/L) | North Pond Top | | | | 0.314 | 0.370 | 0.338 | 0.277 | 0.433 | 0.213 | 0.320 | 0.213 | 0.433 | 0.324 | 0.320 | 0.213 | 0.433 | 0.325 | 0.329 |
| Organic N (mg/L) | North Pond Bottom | | | | 0.362 | 0.439 | 0.439 | 0.880 | 0.700 | 0.640 | 0.730 | 0.362 | 0.880 | 0.599 | 0.640 | 0.439 | 0.880 | 0.638 | 0.670 |
| Organic N (mg/L) | Middle Pond Top | | | | 0.374 | 0.408 | 0.300 | 0.360 | 0.490 | 0.241 | 0.370 | 0.241 | 0.490 | 0.363 | 0.370 | 0.241 | 0.490 | 0.362 | 0.365 |
| Organic N (mg/L) | Middle Pond Bottom | | | | 0.422 | 0.270 | 0.700 | 0.700 | 0.700 | 0.600 | 0.400 | 0.270 | 0.700 | 0.542 | 0.600 | 0.270 | 0.700 | 0.562 | 0.650 |
| Organic N (mg/L) | South Pond Top | | | | 0.383 | 0.415 | 0.495 | 0.348 | 0.610 | 0.286 | 0.421 | 0.286 | 0.610 | 0.423 | 0.415 | 0.286 | 0.610 | 0.429 | 0.418 |
| Organic N (mg/L) | South Pond Bottom | | | | 0.362 | 0.460 | 0.380 | 0.540 | 1.000 | 0.409 | 0.390 | 0.362 | 1.000 | 0.506 | 0.409 | 0.380 | 1.000 | 0.530 | 0.435 |
| TKN (mg/L) | North Pond Top | | | | 0.318 | 0.380 | 0.400 | 0.360 | 0.490 | 0.300 | 0.460 | 0.300 | 0.490 | 0.387 | 0.380 | 0.300 | 0.490 | 0.398 | 0.390 |
| TKN (mg/L) | North Pond Bottom | | | | 0.367 | 0.500 | 0.500 | 1.200 | 1.800 | 0.760 | 1.600 | 0.367 | 1.800 | 0.961 | 0.760 | 0.500 | 1.800 | 1.060 | 0.980 |
| TKN (mg/L) | Middle Pond Top | | | | 0.379 | 0.500 | 0.470 | 0.460 | 0.540 | 0.320 | 0.470 | 0.320 | 0.540 | 0.448 | 0.470 | 0.320 | 0.540 | 0.460 | 0.470 |
| TKN (mg/L) | Middle Pond Bottom | | | | 0.427 | 0.830 | 1.900 | 2.900 | 3.400 | 3.000 | 3.000 | 0.427 | 3.400 | 2.208 | 2.900 | 0.830 | 3.400 | 2.505 | 2.950 |
| TKN (mg/L) | South Pond Top | | | | 0.391 | 0.510 | 0.580 | 0.400 | 0.660 | 0.370 | 0.520 | 0.370 | 0.660 | 0.490 | 0.510 | 0.370 | 0.660 | 0.507 | 0.515 |
| TKN (mg/L) | South Pond Bottom | | | | 0.369 | 0.620 | 1.200 | 0.680 | 1.200 | 0.490 | 0.520 | 0.369 | 1.200 | 0.726 | 0.620 | 0.490 | 1.200 | 0.785 | 0.650 |
| Total Nitrogen (mg/L) | North Pond Top | 0.703 | 0.600 | 0.318 | 0.318 | 0.390 | 0.410 | 0.370 | 0.500 | 0.410 | 0.510 | 0.318 | 0.703 | 0.453 | 0.410 | 0.370 | 0.510 | 0.432 | 0.410 |
| Total Nitrogen (mg/L) | North Pond Bottom | 1.700 | 2.510 | 0.372 | 0.372 | 0.510 | 0.510 | 1.210 | 1.810 | 0.870 | 1.650 | 0.372 | 2.510 | 1.151 | 1.040 | 0.510 | 1.810 | 1.093 | 1.040 |
| Total Nitrogen (mg/L) | Middle Pond Top | 0.555 | 1.052 | 0.384 | 0.384 | 0.620 | 0.520 | 0.510 | 0.590 | 0.370 | 0.520 | 0.370 | 1.052 | 0.551 | 0.520 | 0.370 | 0.620 | 0.522 | 0.520 |
| Total Nitrogen (mg/L) | Middle Pond Bottom | 2.470 | 5.875 | 1.623 | 1.623 | 0.970 | 1.950 | 2.950 | 3.450 | 3.050 | 3.050 | 0.970 | 5.875 | 2.701 | 2.710 | 0.970 | 3.450 | 2.570 | 3.000 |
| Total Nitrogen (mg/L) | South Pond Top | 0.780 | 1.109 | 0.400 | 0.400 | 0.574 | 0.630 | 0.450 | 0.710 | 0.500 | 0.570 | 0.400 | 1.109 | 0.612 | 0.572 | 0.450 | 0.710 | 0.572 | 0.572 |
| Total Nitrogen (mg/L) | South Pond Bottom | 5.500 | 0.819 | 0.379 | 0.379 | 0.780 | 1.250 | 0.940 | 1.250 | 0.559 | 0.570 | 0.379 | 5.500 | 1.243 | 0.800 | 0.559 | 1.250 | 0.892 | 0.860 |
| Total Phosphorus (mg/L) | North Pond Top | 0.013 | 0.013 | 0.010 | 0.010 | 0.014 | 0.014 | < 0.010 | 0.013 | < 0.010 | < 0.010 | 0.010 | 0.050 | 0.019 | 0.014 | 0.010 | 0.014 | 0.012 | 0.012 |
| Total Phosphorus (mg/L) | North Pond Bottom | 0.042 | 0.042 | 0.024 | 0.024 | 0.032 | 0.032 | 0.066 | 0.079 | 0.030 | 0.062 | 0.024 | 0.280 | 0.068 | 0.042 | 0.030 | 0.079 | 0.050 | 0.047 |
| Total Phosphorus (mg/L) | Middle Pond Top | 0.022 | 0.025 | 0.017 | 0.017 | 0.021 | 0.029 | 0.014 | 0.019 | 0.019 | 0.013 | 0.010 | 0.090 | 0.031 | 0.022 | 0.013 | 0.029 | 0.019 | 0.019 |
| Total Phosphorus (mg/L) | Middle Pond Bottom | 0.026 | 0.196 | 0.110 | 0.110 | 0.034 | 0.130 | 0.390 | 0.450 | 0.340 | 0.220 | 0.010 | 0.450 | 0.133 | 0.100 | 0.034 | 0.450 | 0.261 | 0.280 |
| Total Phosphorus (mg/L) | South Pond Top | 0.027 | 0.043 | 0.019 | 0.018 | 0.022 | 0.026 | 0.012 | 0.013 | 0.019 | 0.022 | 0.010 | 0.090 | 0.032 | 0.027 | 0.012 | 0.026 | 0.019 | 0.021 |
| Total Phosphorus (mg/L) | South Pond Bottom | 0.252 | 0.047 | 0.030 | 0.320 | 0.028 | 0.081 | 0.036 | 0.088 | 0.048 | 0.023 | 0.010 | 0.450 | 0.111 | 0.048 | 0.023 | 0.088 | 0.051 | 0.042 |
| Dissolved Phosphorus (mg/L) | North Pond Top | | | | | < 0.010 | < 0.010 | < 0.010 | 0.012 | < 0.010 | < 0.010 | 0.010 | 0.012 | 0.010 | 0.010 | 0.010 | 0.012 | 0.010 | 0.010 |
| Dissolved Phosphorus (mg/L) | North Pond Bottom | | | | | < 0.010 | < 0.010 | 0.010 | 0.011 | 0.011 | < 0.010 | 0.010 | 0.011 | 0.010 | 0.010 | 0.010 | 0.011 | 0.010 | 0.010 |
| Dissolved Phosphorus (mg/L) | Middle Pond Top | | | | | < 0.010 | 0.010 | < 0.010 | < 0.010 | < 0.010 | < 0.010 | 0.010 | 0.010 | 0.010 | 0.010 | 0.010 | 0.010 | 0.010 | 0.010 |
| Dissolved Phosphorus (mg/L) | Middle Pond Bottom | | | | | < 0.010 | 0.074 | 0.052 | 0.046 | 0.067 | 0.062 | 0.010 | 0.074 | 0.052 | 0.057 | 0.010 | 0.074 | 0.052 | 0.057 |
| Dissolved Phosphorus (mg/L) | South Pond Top | | | | | < 0.010 | < 0.010 | < 0.010 | < 0.010 | < 0.010 | < 0.010 | 0.010 | 0.010 | 0.010 | 0.010 | 0.010 | 0.010 | 0.010 | 0.010 |
| Dissolved Phosphorus (mg/L) | South Pond Bottom | | | | | < 0.010 | 0.014 | < 0.010 | 0.014 | < 0.010 | < 0.010 | 0.010 | 0.014 | 0.011 | 0.010 | 0.010 | 0.014 | 0.011 | 0.010 |

# Storm Water Quality Data from Congamond Lakes in 2015

| Parameters | Station | < | 9/12/15 | < | 10/3/15 | < | 10/10/15 | | Mean |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | |
| Ammonium N (mg/L) NH3 | North Pond Rd | | 0.270 | | 0.330 | | 0.220 | | 0.27 |
| Ammonium N (mg/L) | Lakemont Street | | 0.700 | | | | | | 0.70 |
| Ammonium N (mg/L) | Veteran Street | | 0.360 | | 0.340 | | 0.210 | | 0.30 |
| Ammonium N (mg/L) | Echo Road | | 0.090 | | 0.097 | | 0.170 | | 0.12 |
| Ammonium N (mg/L) | Island Pond Raod | | 0.320 | | 0.170 | | | | 0.25 |
| Ammonium N (mg/L) | White Street | | 0.200 | | 0.160 | | 0.250 | | 0.20 |
| Ammonium N (mg/L) | Rt 168 Crabby Joes | | 0.280 | | 0.620 | | 0.380 | | 0.43 |
| Ammonium N (mg/L) | Shore Road | | 0.620 | | 0.750 | | 0.200 | | 0.52 |
| | | | | | | | | | |
| Nitrate N (mg/L) NOX | North Pond Rd | | 0.330 | | 0.064 | | 0.072 | | 0.16 |
| Nitrate N (mg/L) | Lakemont Street | | 0.290 | | | | | | 0.29 |
| Nitrate N (mg/L) | Veteran Street | | 0.061 | | 0.067 | < | 0.050 | | 0.06 |
| Nitrate N (mg/L) | Echo Road | < | 0.050 | | 0.066 | < | 0.050 | | 0.06 |
| Nitrate N (mg/L) | Island Pond Raod | | 0.310 | | 0.068 | | | | 0.19 |
| Nitrate N (mg/L) | White Street | | 0.240 | | 0.064 | < | 0.050 | | 0.12 |
| Nitrate N (mg/L) | Rt 168 Crabby Joes | | 0.062 | | 0.066 | < | 0.050 | | 0.06 |
| Nitrate N (mg/L) | Shore Road | < | 0.050 | | 0.077 | < | 0.050 | | 0.06 |
| | | | | | | | | | |
| TKN (mg/L) | North Pond Rd | | 2.100 | | 1.300 | | 1.900 | | 1.77 |
| TKN (mg/L) | Lakemont Street | | 6.900 | | | | | | 6.90 |
| TKN (mg/L) | Veteran Street | | 3.400 | | 1.300 | | 1.400 | | 2.03 |
| TKN (mg/L) | Echo Road | | 0.480 | | 0.230 | | 1.100 | | 0.60 |
| TKN (mg/L) | Island Pond Raod | | 1.300 | | 1.100 | | | | 1.20 |
| TKN (mg/L) | White Street | | 1.100 | | 1.600 | | 1.700 | | 1.47 |
| TKN (mg/L) | Rt 168 Crabby Joes | | 0.990 | | 4.000 | | 1.500 | | 2.16 |
| TKN (mg/L) | Shore Road | | 3.800 | | 5.600 | | 2.200 | | 3.87 |
| | | | | | | | | | |
| Total Nitrogen (mg/L) | North Pond Rd | | 2.430 | | 1.364 | | 1.972 | | 1.92 |
| Total Nitrogen (mg/L) | Lakemont Street | | 7.190 | | | | | | 7.19 |
| Total Nitrogen (mg/L) | Veteran Street | | 3.461 | | 1.367 | | 1.450 | | 2.09 |
| Total Nitrogen (mg/L) | Echo Road | | 0.530 | | 0.296 | | 1.150 | | 0.66 |
| Total Nitrogen (mg/L) | Island Pond Raod | | 1.610 | | 1.168 | | | | 1.39 |
| Total Nitrogen (mg/L) | White Street | | 1.340 | | 1.664 | | 1.750 | | 1.58 |
| Total Nitrogen (mg/L) | Rt 168 Crabby Joes | | 1.052 | | 4.066 | | 1.550 | | 2.22 |
| Total Nitrogen (mg/L) | Shore Road | | 3.850 | | 5.677 | | 2.250 | | 3.93 |
| | | | | | | | | | |
| Total Phosphorus (mg/L) | North Pond Rd | | 0.590 | | 0.220 | | 0.370 | | 0.393 |
| Total Phosphorus (mg/L) | Lakemont Street | | 1.600 | | | | | | 1.600 |
| Total Phosphorus (mg/L) | Veteran Street | | 0.690 | | 0.220 | | 0.320 | | 0.410 |
| Total Phosphorus (mg/L) | Echo Road | | 0.110 | | 0.063 | | 0.240 | | 0.138 |
| Total Phosphorus (mg/L) | Island Pond Raod | | 0.460 | | 0.270 | | | | 0.365 |
| Total Phosphorus (mg/L) | White Street | | 0.380 | | 0.220 | | 0.560 | | 0.387 |
| Total Phosphorus (mg/L) | Rt 168 Crabby Joes | | 0.120 | | 0.450 | | 0.170 | | 0.247 |
| Total Phosphorus (mg/L) | Shore Road | | 0.630 | | 0.660 | | 0.170 | | 0.487 |
| | | | | | | | | | |
| Dissolved Phosphorus (mg/L) | North Pond Rd | | 0.170 | | 0.071 | | 0.057 | | 0.099 |
| Dissolved Phosphorus (mg/L) | Lakemont Street | | 0.120 | | | | | | 0.120 |
| Dissolved Phosphorus (mg/L) | Veteran Street | | 0.140 | | 0.068 | | 0.079 | | 0.096 |
| Dissolved Phosphorus (mg/L) | Echo Road | | 0.080 | | 0.044 | | 0.091 | | 0.072 |
| Dissolved Phosphorus (mg/L) | Island Pond Raod | | 0.410 | | 0.094 | | | | 0.252 |
| Dissolved Phosphorus (mg/L) | White Street | | 0.220 | | 0.120 | | 0.190 | | 0.177 |
| Dissolved Phosphorus (mg/L) | Rt 168 Crabby Joes | | 0.081 | | 0.065 | | 0.024 | | 0.057 |
| Dissolved Phosphorus (mg/L) | Shore Road | | 0.220 | | 0.021 | | 0.017 | | 0.086 |
| | | | | | | | | | |
| Turbidity (FNU) | North Pond Rd | | 4.85 | | 37.7 | | 77.8 | | 40.1 |
| Turbidity (FNU) | Lakemont Street | | 11.4 | | | | | | 11.4 |
| Turbidity (FNU) | Veteran Street | | 5.14 | | 40.6 | | 14.8 | | 20.2 |
| Turbidity (FNU) | Echo Road | | 3.78 | | 5.41 | | 21.7 | | 10.3 |
| Turbidity (FNU) | Island Pond Raod | | 3.55 | | 22.5 | | | | 13.0 |
| Turbidity (FNU) | White Street | | 5.28 | | 15.6 | | 135 | | 52.0 |
| Turbidity (FNU) | Rt 168 Crabby Joes | | 2.63 | | 95.8 | | 72.8 | | 57.1 |
| Turbidity (FNU) | Shore Road | | 6.75 | | 115 | | 20 | | 47.3 |
| | | | | | | | | | |