PHOSPHORUS LOADING ASSESSMENT FOR LAKE GARFIELD, MONTEREY, MASSACHUSETTS

Project 2016-01/604

Prepared by Water Resource Services, Inc.

Final Report
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AND

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List of Acronyms

ARC	Aquatic Restoration Consulting
BMP	Best Management Practices
COC	Chain-of-Custody
DP	Dissolved Phosphorus
(MA)DEP	Massachusetts Department of Environment Protection
DO	Dissolved Oxygen
GIS	Geographic Information System
LID	Low Impact Development (techniques)
LLRM	Lake Loading Response Model
N	Nitrogen
P	Phosphorus
QA/QC	Quality Assurance/Quality Control
QAPP	Quality Assurance Project Plan
RPD	Relative Percent Difference
SOP	Standard Operating Procedure
TMDL	Total Maximum Daily Load
TP	Total Phosphorus
USEPA	U.S. Environmental Protection Agency
WRS	Water Resource Services

Introduction

Lake Garfield is a Great Pond under the laws of the Commonwealth of Massachusetts and a major public recreational resource in Monterey (Figures 1 and 2). The lake covers 97.6 ha (244 ac), although the pond west of Tyringham Road is included in some estimates, increasing total area to 105 ha (262 ac). It provides habitat and recreational opportunity, with a town beach and public boat launch at the northwest end and many private residences along its shore. Lake Garfield achieves a maximum depth of 9.4 m (31 feet) (Figure 3), with a total volume estimated at 4.34 million m3 (3518 ac-ft).

There is concern over perceived increases in phosphorus (P) levels, low oxygen in deeper water, and blooms of cyanobacteria. P concentrations are not extreme (range of 10-30 µg/L in surface water from past monitoring) and cyanobacteria blooms have not yet been severe or frequent. Yet problems have been sufficient to put Lake Garfield on the impaired waters list for P and oxygen. Lake Garfield has a diverse aquatic plant community, which includes one endangered species (Vasey's pondweed) and one invasive species (Eurasian watermilfoil), and hosts a well-regarded fishery. Lake Garfield is an important natural resource for the Town of Monterey and Berkshire County.

The watershed (Figure 4) is largely forested and covers only about 8 times the lake area. It is drained by multiple small tributaries that have not been quantitatively assessed. Internal loading of P was estimated to be quite high in one past study (AES 1990), but that estimate was not properly substantiated. Internal release is not expected to be high in this region, where calcium is the dominant P binder, but site-specific investigation is warranted. Some combination of watershed and in-lake methods may indeed be necessary to meet water quality goals, but Lake Garfield has not been the subject of a complete diagnostic/feasibility study. Ground water seepage is generally not thought to be a large contributor to water or P inputs to Berkshire lakes, given soils dominated by glacial till with relatively low permeability and high clay content. Water moves slowly underground in this area unless there are fractures, and P is readily adsorbed onto clay particles. A past study of Lake Garfield (Fugro 1994) suggested that P loading to Lake Garfield was indeed low, but concern over the impact of on-site waste water disposal systems within the watershed prompted a supplemental investigation during the period of the 604b grant.

Low oxygen is a natural condition in many lakes, but one that is often exacerbated by human actions. Knowledge of the actual oxygen demand allows consideration of whether reduced internal production (based mainly on P control) can counter the demand or if other measures (most often oxygenation) will be needed to meet the oxygen standard in deep water. This study will facilitate an assessment of oxygen conditions, demand, and possible remediation actions.

Lake Garfield is not listed as impaired by cyanobacteria, but past monitoring in the last half of summer has included observations of cyanobacterial particles in the water, and on calm mornings surface scums can form. The lake has never been posted for cyanobacteria as a threat to humans or pets, but algae monitoring has not been regular and observations are anecdotal. As a supplement to the 604b project, plankton samples have been collected during 2017 and offer more insight into lake condition.

The 604b grant affords the opportunity to investigate P loading, evaluate current conditions, and consider management alternatives. This report provides the results of all water quality investigations conducted during 2017, with a focus on P loading from all possible sources and resulting lake condition.

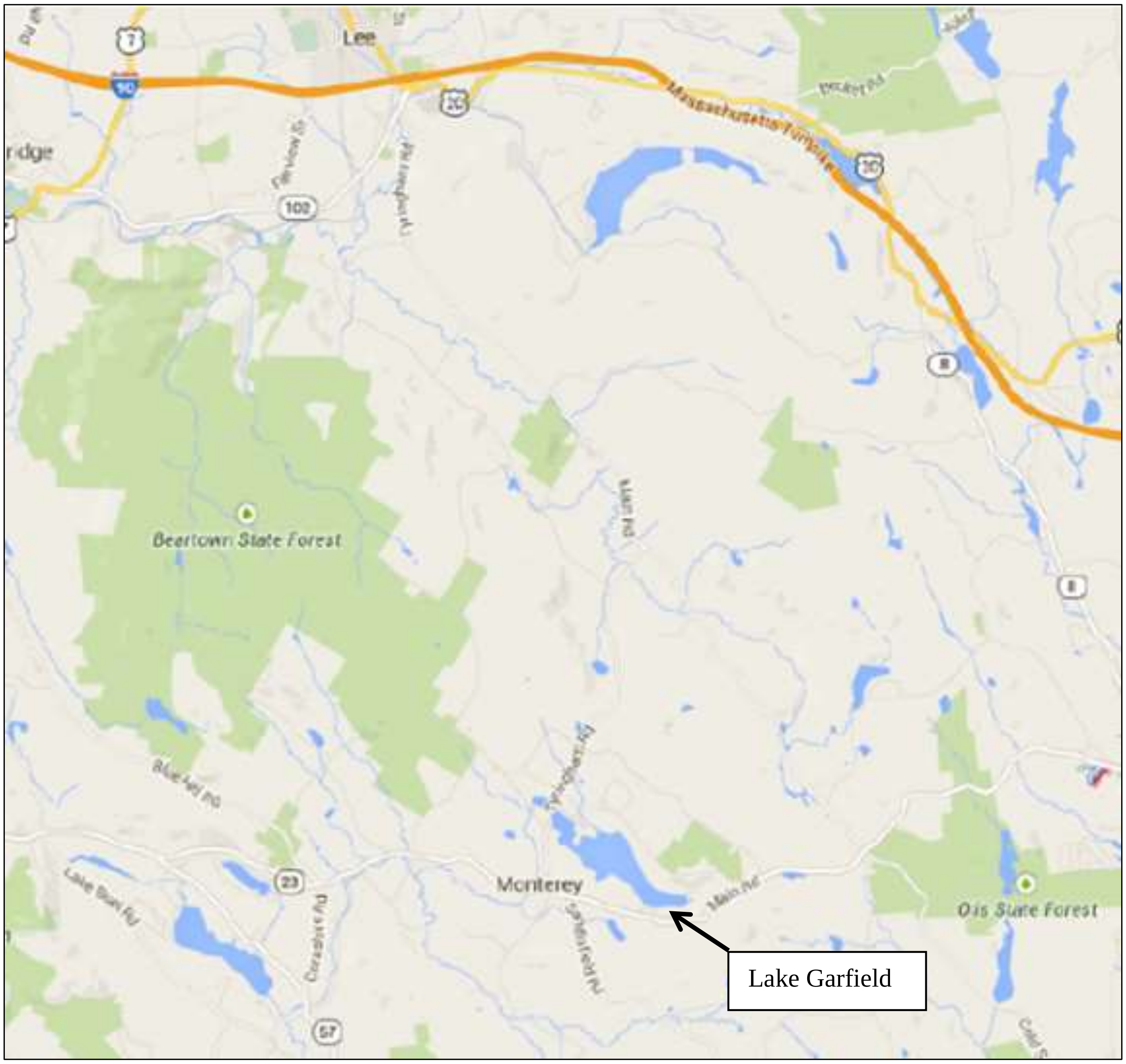


Figure 1. Lake Garfield location in Monterey.

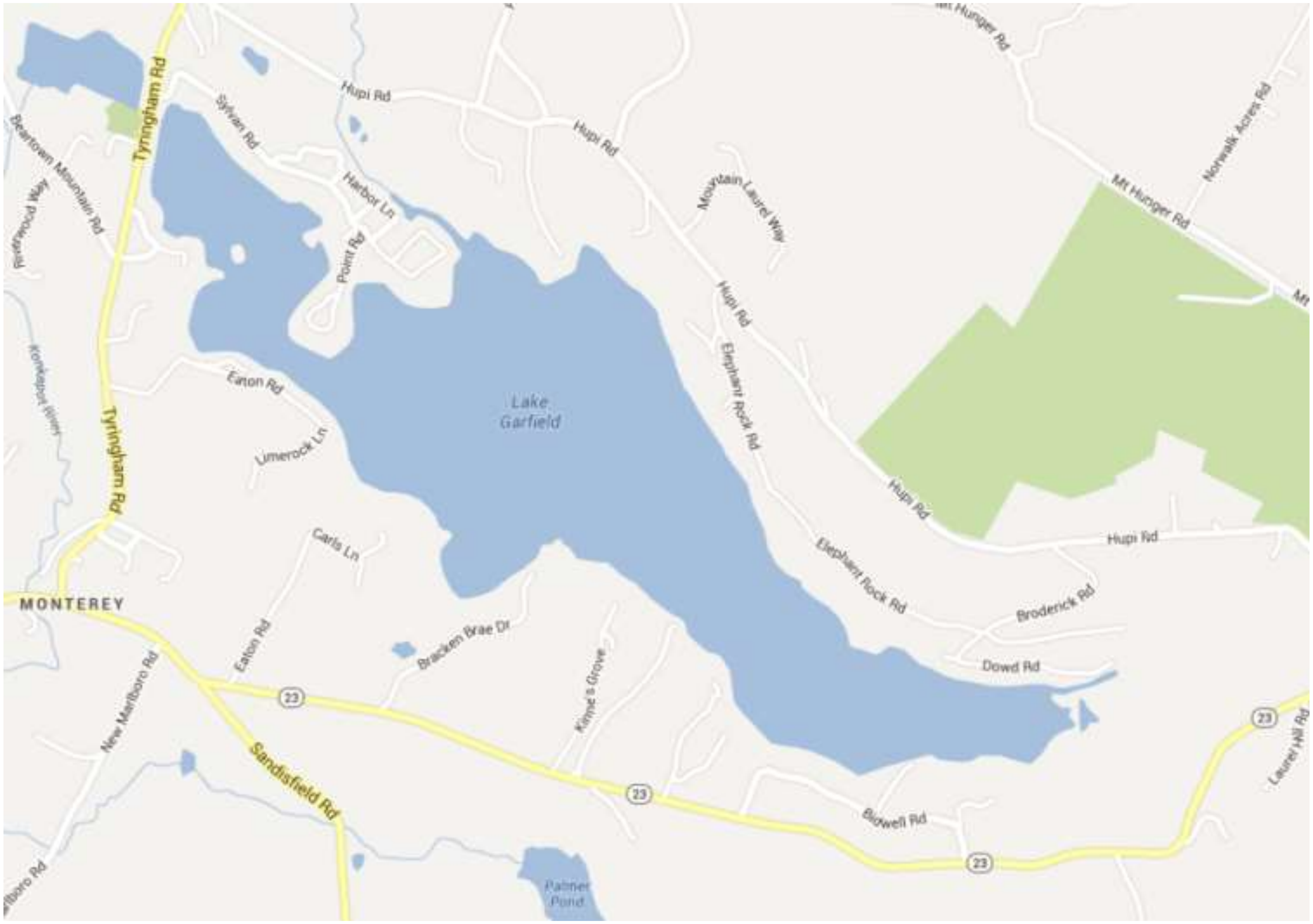


Figure 2. Lake Garfield and surrounding streets

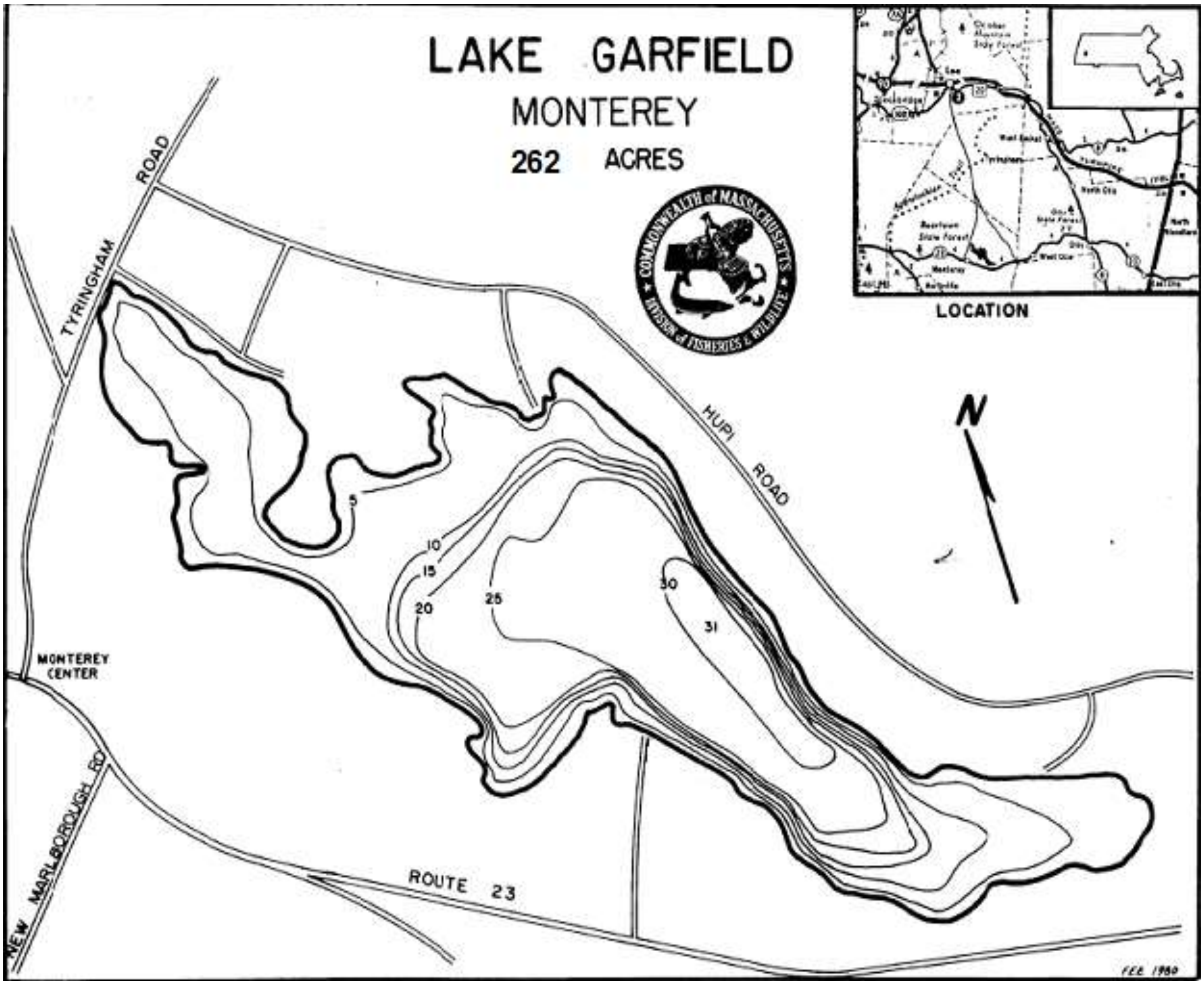


Figure 3. Lake Garfield bathymetry.

Figure 4. Lake Garfield watershed.

Project Approach

The goal of this study is to assess phosphorus (P) loading to Lake Garfield, facilitating evaluation of lake remediation options to improve the water quality, although the focus of the 604b grant is on data collection and load estimation. There are many points of potential runoff around the lake, and with increased residential development and more year-round use, inputs from these areas could be significant. Sampling of base flows and runoff provides data to characterize watershed loading of phosphorus (P) to Lake Garfield and allows calibration of a model of watershed inputs to the lake.

Anoxia occurs in the deepest water of Lake Garfield during summer, which represents slightly less than one third of the lake area. P bound to iron could be released into the water column, but there are no data for deep water P concentrations or the Fe-P content of the surficial sediments in deep areas. Lake Garfield is in the Berkshire County limestone belt, and much P may be bound to calcium, limiting release under anoxia. The amount of available P and its release into the water need to be quantified for a complete picture of P loading to the lake, and involves both oxygen and sediment P measurement.

Oxygen demand can be estimated with temperature/dissolved oxygen (T/DO) profiles collected while oxygen concentrations are >2 mg/L. It is harder for decay processes to remove oxygen as it approaches 0 mg/L, leading to less oxygen loss over the measurement period and underestimation of actual oxygen demand when available oxygen is low. Reasonably accurate estimates of oxygen demand can be obtained from spring profiles when stratification is setting up but oxygen depletion has not yet occurred. Sediment P is assessed by a series of extractions from samples collected over the area of the lake bottom likely to be exposed to anoxia.

With the watershed and internal loads assessed, we can calibrate a model of the lake and estimate the acceptable load of P to the lake based on target levels of water clarity and bloom probability. Model scenarios can be run to evaluate the levels of watershed management and internal load control needed to reach acceptable conditions, and the efficacy of specific possible actions (i.e., non-structural or structural storm water controls, P inactivation for sediment, oxygenation).

The needs expressed above led to formulation of a program of investigation. Specific tasks to be accomplished include the development of a Quality Assurance Project Plan (QAPP) as Task 1. The QAPP has been provided as a separate document (WRS 2017). The following additional tasks were completed in accordance with the QAPP:

2. Obtaining in-lake oxygen profiles in the deepest part of the lake from spring through summer
3. Measuring in-lake phosphorus concentrations in the epilimnion and hypolimnion during stratification
4. Pre-, first flush and post-storm water sampling for 3 storms to capture inflowing phosphorus concentrations from identified input locations
5. Assessment of available P in surficial sediment in areas subject to anoxia
6. Assess nutrient loading using the Lake Loading Response Model (LLRM):
    - 6.1 Calibration of the model
    - 6.2 Determination of target P loading to meet water quality objectives
    - 6.3 Testing of watershed and in-lake management scenarios to determine how target loading can be met

Additionally, the Town of Monterey and Friends of Lake Garfield asked WRS Inc. to evaluate plankton and ground water seepage as supplements to the 604b study. The result was two additional tasks:

7. Collect and analyze phytoplankton and zooplankton samples on the same dates as in-lake P assessments
8. Assess the seepage of ground water into the lake, including the quantity and quality of inflow

As a final task we prepared a comprehensive report of the results and management implications of the above tasks.

Oxygen Profiles

BACKGROUND:

Oxygen has been measured as part of a volunteer monitoring program for over a decade, plus various professional monitoring events prior to that, but the recent measurements tend to cease at a depth close to 6 m (20 feet), and usually there is still substantial oxygen at that point. The lake has a maximum depth of 9.4 m (31 feet), with about one third of the lake deeper than 6 m, so a substantial area and moderate volume are not addressed by many measurements. For those profiles that extend deeper than 6 m, the thermocline is between 6 and 7 m and anoxia can occur just below that depth. Gaining an understanding for the timeframe of loss of oxygen in deeper water is important to understanding possible internal P loading. Measuring oxygen demand is important to estimating whether P control can ameliorate that demand or additional actions will be needed to meet the state standard.

APPROACH:

Oxygen status will be assessed at the deepest location with a Hach DS5 that measures oxygen, temperature, pH, conductivity, turbidity and chlorophyll-a. The instrument is calibrated in the office each day prior to deployment in the field, in accordance with the QAPP. Measurements were made at 1 m intervals from surface to bottom, with the deepest measurements collected near the sediment-water interface. Assessment occurred in April, May, June, July, August and September. Oxygen profiles were assessed along a horizontal gradient in August to determine the extent of anoxia laterally from the deep hole area (Figure 5).

These measurements allow both calculation of oxygen demand (which properly must occur in the spring before deep water values drop below 2 mg/L and the kinetics of oxygen uptake are altered – see Standard Methods, 22nd edition, page 2-88) and estimation of the bottom area exposed to anoxia during the period of stratification. The areal and temporal extent of low oxygen factor into the calculation of P loading from surficial sediment.

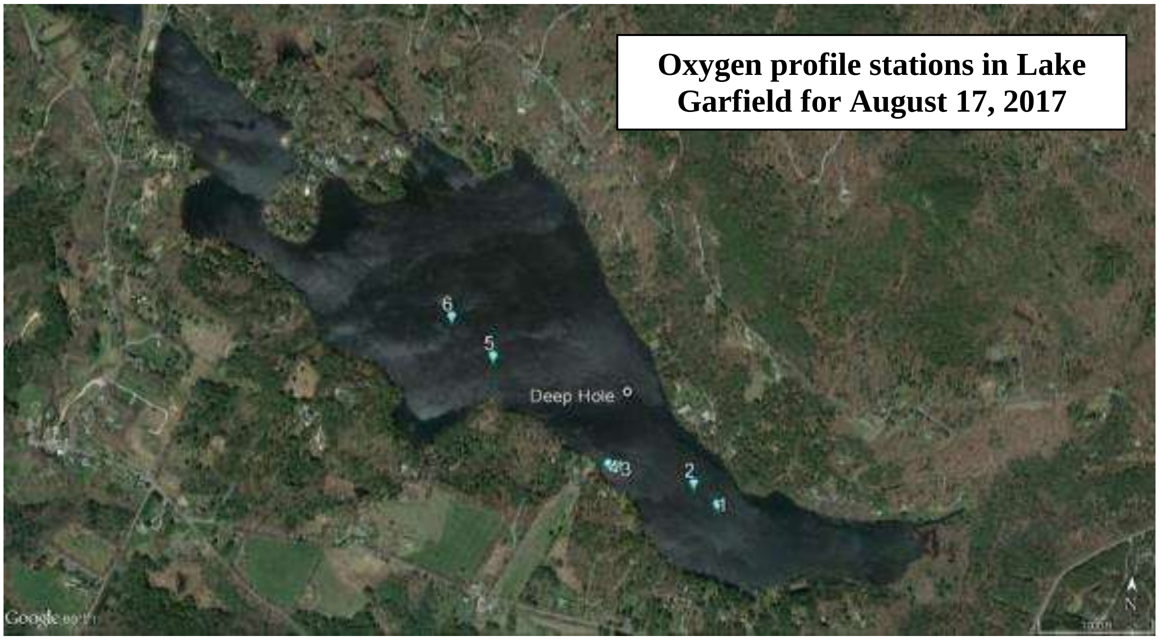


Figure 5. Oxygen profile stations in Lake Garfield for August 17, 2017.

In-lake Phosphorus Levels

BACKGROUND:

Phosphorus has been measured in Lake Garfield on individual spring and summer dates up to 3 times per year since 2003, but only at the surface or mid-depth, not in deep water. Values have ranged from below the detection limit (and desirable value) of 10 µg/L to 30 µg/L, with an average of 21 µg/L, although this average sets values below the detection limit at that limit, so the actual average is undoubtedly lower. This average is, however, at the threshold for expected unacceptable algae bloom frequency (between 20 and 25 µg/L). This means there will be good years and bad years, probably mostly weather dependent, and high runoff or warm conditions that promote internal recycling will figure strongly in P availability, algae production and water clarity. Water clarity measurements are more frequent than the P testing, with a range of 2.0 to 5.0 m (6.6 to 16.5 feet) and an average of 3.7 m (12.2 feet). This does not indicate severe impairment, but is low among Berkshire Lakes not experiencing nutrient loading problems and suggests a decline from limited values from the 1970s or earlier (unpublished data). Clarity in Lake Garfield is not likely a function of suspended non-living solids and will vary with algae abundance and particle size distribution. Clarity should therefore be tied to surface P concentrations unless cyanobacteria are absorbing P at the sediment surface and then rising in the water column, a known mode of bloom generation that could be at work in Lake Garfield. Either way, P control is the key to algae control, and P levels in Lake Garfield need to be better understood.

APPROACH:

Water samples were collected at the deepest point in the lake (Figure 6) from near the surface, at the thermocline when the lake is stratified (6-7 m), and close to the bottom (9 m) in April, May, June, July, August and September to observe the change in P levels and any build-up in the bottom waters. Testing included total and dissolved P. This facilitates two methods of internal loading calculation (hypolimnetic accumulation and hypolimnetic vs. epilimnetic concentration), as well as corroborating estimated release rates for P from sediment and allowing comparison of epilimnetic P mass with watershed inputs. In late June 5 surface locations (Figure 6) were sampled to characterize variability of P concentration over a horizontal gradient in the lake.

Samples were collected with a 2 L horizontal alpha bottle that was lowered to the target depth and closed with a messenger weight slid down the rope. The sampler was rinsed with lake water between samples and moved around at the target depth to promote exchange of water at the target depth. Samples were placed in dedicated plastic bottles without preservative, kept on ice in the dark, and delivered to the lab (which is nearby) within 2 hours. Samples were processed in the lab in accordance with standard methods. A trip blank of distilled water and at least one duplicate sample was collected on each date, with the duplicate selected randomly prior to the start of sampling. If tributary sampling occurred on the same date as in-lake sampling, it is possible that the duplicate sample was a tributary sample.

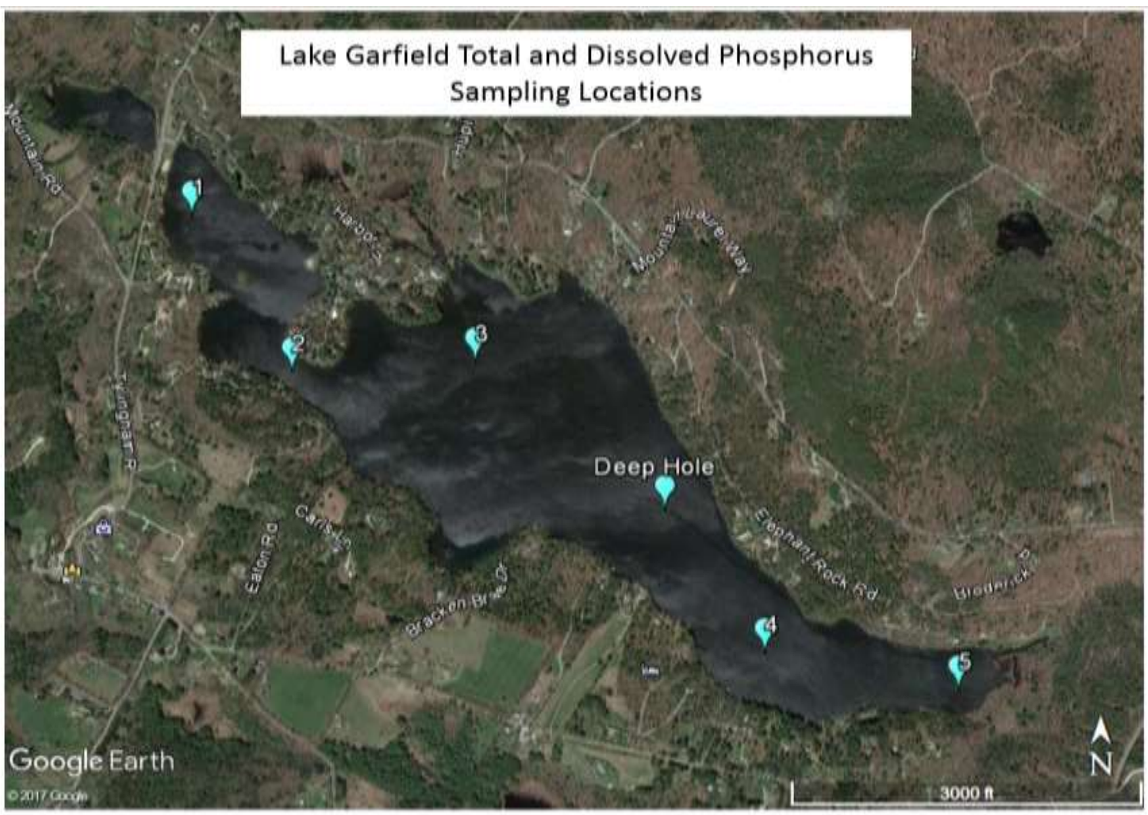


Figure 6. Phosphorus sampling stations in Lake Garfield on June 29, 2017

Inflowing Phosphorus Concentrations

BACKGROUND:

The only available inflow data indicate that P concentrations in the most permanent tributary to Lake Garfield (station H in Table 1, Figure 7) are not appreciably different than concentrations in the lake itself, but these are dry weather data for only a part of the watershed. External loading may or may not be a monthly to seasonal force in determining lake conditions, but will be the ultimate source of most P in the lake. Understanding the level of inputs associated with storm water inputs is likely to be important in this system, with steep slopes and soils of limited permeability. Additionally, the lake undergoes a drawdown each winter and is refilled in early spring, so both snow melt and spring runoff are likely to have a disproportionate impact on water quality. Surface water inputs require assessment to appropriately characterize P loading to Lake Garfield.

APPROACH:

Possible inflow locations were identified during a wet weather field survey (Table 1, Figure 7). The intention was to sample as many of these stations as could be accessed (some are on private property) for total and dissolved P just prior to, during, and near the end of multiple storm events. Shortly before an expected storm event, passive samplers were set out to collect first flush samples, and pre-storm samples were collected where there was flow at that time. The passive samplers were retrieved later in the storm, after peak flows but before baseflow conditions resumed if possible, and additional samples were collected at that time. Thus there could be 3 samples associated with each storm event, although small runoff sites could have no flow prior to or after a storm.

Nine (9) inflow locations were initially identified through a wet weather field survey, but the list was expanded to 13 possible sampling sites with additional field investigation (Table 1, Figure 7). Two small streams combined just upstream of the lake (F+G) and were sampled as one system. Two sites (A and M1) appeared to be mainly ground water directed into piped flow, and flow was detected only early in the program (March/April) when soils were saturated. Several inflows (C, J, K, L, M2) had no flow during dry weather and are mainly storm water drainage systems. One drainage ditch (E) never had any flow that reached the lake, either from snow melt or storm water runoff, but runoff occurs within the associated drainage basin. Remaining sites (B, D, F+G, H and I) appeared to be permanent tributaries, at least with the average to slightly wet weather of 2017, but some could dry up under prolonged drought as occurred in 2016. Station C was on private property and permission to sample it after the initial investigation was not granted.

The first flush sampling was performed using simple automated grab samplers (passive samplers) as described in the QAPP. Multiple samplers were sometimes deployed and in most cases samples were composited, but in a few cases separate samples were submitted to the lab to assess variability. Samples for each station were collected in (or transferred to) dedicated plastic bottles, stored in the dark on ice, and delivered to the nearby lab within a few hours in nearly all cases. However, where weekend collection was necessary, samples were preserved with sulfuric acid and kept in a refrigerator until delivery within a few days. Samples were tested for total and dissolved P, except for acid-preserved samples, which were tested only for total P.

Table 1. Inflow sampling stations for Lake Garfield

ID	Closest Residential Address	Notes
A	55 Eaton	Underdrains through wall supporting road, flows with saturated soil, but no channel or evidence of overland flow to lake.
B	Private Road, Bracken Brae Farm	Small stream through mostly wooded drainage area with some farm influence, fed partly by clean spring used for water supply, no easy access (private property)
C	28 Bidwell	Small runoff system, mostly road drainage. Sampled as close to lake as possible on Wolff property.
D	617 Main	Culvert under road follows ravine to the lake, small, steep tributary. Sampled down slope about half way to lake.
E	22-24 Elephant Rock Rd.	Across from tennis/basketball court. Small drainage channel. Other similar channels in area, but no flow ever observed, even during substantial storms.
F+G	179 Hupi	Two stream branches each cross Hupi Rd then combine into one stream near lake. Beyond cottages at 179 Hupi. F is NW branch, G is SE branch. Sampled together as F/G.
H	12 Brewers Ln	Stream N of Brewers Lane, largest inlet to lake, sampled just below beaver dam but upstream of lake backwater influence.
I	End of Dowd Rd.	Stream crosses Hupi Rd, splits just upstream of lake due to accumulated sediment in wooded wetland, smaller part enters defined arm of lake, larger part enters to W in wetland area. Sampled together upstream of split.
J	17 Limerock	Road drainage, pipe discharges by Putrine property.
K	611 Main	Large storm drainage system, crosses Rt 23 and discharges to slope that runs to lake, sampled near discharge downstream of Rt 23.
L	16 Bidwell	Small stream crosses Bidwell from wet area, picks up additional road drainage, sampled below mixing zone.
M1	3rd house from end of Dowd Rd (no # evident)	Small drainage pipe from residential property, sampled at discharge, close to lake. Appears to be mainly ground water seepage.
M2	Community beach area	Drainage channel that runs along Dowd Rd, crosses at bottom of hill, and runs along edge of grassy area used as community lake access

Figure 7. Locations of inflow sampling around Lake Garfield 2017

Phosphorus in Surficial Sediment

BACKGROUND:

Under anoxic conditions, P is often released from Fe-P compounds and enters the water column above. This is the primary means of P release from sediment, although not the only mechanism. We measured the actual accumulation rate of P in the hypolimnion, but if inactivation is needed, the amount of Fe-P in the sediment must be known to calculate an appropriate dose of inactivator, usually aluminum. Further, the amount of Fe-P in the surficial sediments (upper 10 cm) can be used to provide an independent estimate of P release. In geographic areas where calcium is abundant, it is also possible that Fe-P is low, in which case accumulation may be a function of other processes, such as settling of particles from above or decomposition. Knowledge of Fe-P in surficial sediments is important to a more complete understanding of loading.

APPROACH:

Surficial sediment was sampled at 5 locations in Lake Garfield (Figure 8), plus a duplicate at one station, covering the area known to be exposed to anoxia. Only one sampling is needed to characterize the sediment, and sampling can occur any time of year. Samples were collected with a universal gravity corer a described in the QAPP and only the upper 10 cm of sediment were collected. Samples were packed with freezer packs and overnighted to the laboratory. Samples were tested for percent solids/percent water, total organic carbon (representing organic content), TP, loosely bound P, Fe-P, Al-P, Ca-P, biogenic P (the more available fraction of organic P) and organic P.

Figure 8. Locations of sediment sampling in Lake Garfield on July 6, 2017

Plankton

BACKGROUND:

Algae and planktonic animals are important links in the aquatic food web. Knowledge of the types and density of each in a lake over time helps with interpretation of the impact of phosphorus loading on the aquatic system. Blooms of cyanobacteria are of particular concern, and have been reported from Lake Garfield in the past, although such blooms have been uncommon and unquantified. Zooplankton represent energy flow between algae and small fish, and the abundance and mean length of crustacean zooplankton can be a reflection of the fish community.

APPROACH:

Phytoplankton samples were collected from just below the surface of the lake at the deep hole station used for water quality sampling between April and September 2017. Whole water samples were collected in 250 mL bottles and preserved with glutaraldehyde to a concentration of 0.5%. Samples were settled in the lab and concentrated before quantitative examination under phase contrast optics at 200-400X. The final multiplication factor for cells observed to cells/mL of raw sample was <25 in all cases.

Zooplankton samples were collected by towing a net with 80 um mesh through 30 m of water. With a net diameter of 5 inches, this results in 380 L of water being filtered. Samples were preserved with glutaraldehyde at a concentration of 2%, settled in the lab, and quantitatively examined under phase contrast optics at 100X magnification. Final multiplication factors for converting observed specimens to density per liter were <1 in all cases.

Ground Water

BACKGROUND:

Ground water is one source of water to lakes, along with direct precipitation, runoff, and any permitted discharges. In the Berkshire region of Massachusetts, it is usually assumed that ground water flow is a minor input, as the soils are “tight”, affording only slow lateral movement of ground water. However, fissures in rock or cracks in clay can allow greater flow, and direct measurement can be insightful. During past drawdowns of Lake Garfield, breakout of ground water was noticed in some areas, although major inseepage is still not expected. Even at lower flow rates, however, there is potential for on-site wastewater inputs to affect seepage quality and add significantly to the phosphorus load to the lake. Therefore, assessment of ground water quantity and quality is worthwhile, even if just to confirm that this is not a major source.

APPROACH:

Ground water inseepage was measured with seepage meters after the method of Mitchell et. al (1988). Measurements were made at 15 stations (Figure 9). Plastic seepage meters which are the ends of 55 gallon cider drums with spouts installed, are inserted into the sediment in shallow water, such that any inseepage will be captured in a bag attached to the spout, and any outseepage will pull water (100 mL loaded at the start) out of the bag. Seepage meters are left in place for 3-4 hours, after which the change in bag volume is measured. Measurements are converted to liters of seepage (in or out) per square meter per day, and those values can be multiplied by the area of expected seepage represented by the meter. A total of 15 locations (Figure 9) were assessed, with six locations having duplicate seepage measurements. The seepage area represented was calculated as half the distance to the next closest seepage meter and out to the 6 meter (20 feet) depth contour.

Seepage quality was assessed by the method of Mitchell et al. (1988, 1989), using Littoral Interstitial Porewater (LIP) samplers, at the same stations for which seepage quantity was assessed. The sampler is functionally a miniature well. The sampler is pushed into the sediment near where the seepage meter had been, usually to a depth of at least 0.2 m. The outer sleeve is pulled up to reveal a screened set of holes, a hand pump is attached to a flask that acts as a trap, which is attached to the upper end of the mini-well with tubing. The hand pump generates pressure and pulls porewater into the flask. The first 100 mL are used as a rinse, and the next several hundred mL are kept as the sample. Samples were collected at the same locations as seepage meters, except when there was no measured seepage (one location). Final samples were field filtered through 0.45 um glass fiber filters and were delivered to the lab the same day for testing for total phosphorus, Kjeldahl nitrogen, and nitrate nitrogen, all of which were dissolved fractions due to the field filtering, which removed any particles that were captured by the sampling but would not be expected to travel into the lake with ground water.

Total estimated seepage values were multiplied by phosphorus and nitrogen concentrations from associated quality samples to get loading estimates for each established seepage zone. Those values were summed to get a total load to the lake from ground water.

Figure 9. Locations of seepage meters and LIP samplers in Lake Garfield in 2017

Assess Nutrient Loading Using LLRM

BACKGROUND and APPROACH:

For supplemental information regarding the use of the Lake Loading and Response Model (LLRM), please refer to the LLRM users guide and QAPP (AECOM 2009). Three separate subtasks have been identified for this task.

Set up LLRM

BACKGROUND:

The measurements made in previous tasks are very useful in understanding current loading, but to make predictions one needs a model. The Lake Loading and Response Model, LLRM, is a public domain model used in multiple TMDL efforts, including projects in NH, MA and CT. It is a fairly simple spreadsheet model that generates P loads from land uses and areas in the watershed, routes them to the lake with attenuation that can be corroborated with real data at inlet points, and predicts in-lake conditions including P concentration, water clarity, and the probability of chlorophyll at any desired level based on depth, flushing, and inputs. LLRM has a user manual and a QAPP already in place (AECOM 2009).

APPROACH:

LLRM was set up for Lake Garfield and its watershed. The model was calibrated with inputs and in-lake data. Once set up to appropriately model reality, the model can be used to back-calculate desirable loading and test management scenarios.

Determine Target Loading

BACKGROUND:

Desirable levels of water clarity and chlorophyll concentration probability can be set based on user preferences and literature linking uses to conditions. These levels can be translated into P concentration through the model, and in turn into acceptable loading levels.

APPROACH:

The model was used to determine what load of P is tolerable to reach desired levels of water clarity and chlorophyll concentration. The MA DEP and the Friends of Lake Garfield have been offered input in setting target conditions. A TP concentration close to 10 µg/L and water clarity averaging about 4 m appears to be appropriate, although there is an additional desire that cyanobacteria blooms be minimized. We used the model to determine what load will meet the TP and clarity goals. Minimizing cyanobacteria may be more complicated, involving nitrogen to phosphorus ratios (nitrogen was not assessed in this study) and growth of cyanobacteria at the sediment water interface with rise into the water column to form blooms, independent of surface P concentrations.

Evaluate Possible Load Reduction Scenarios

BACKGROUND:

Models are used to test "what if" scenarios of environmental management to determine probable impacts on a resource and whether or not desired conditions can be achieved by specific actions. Once a target load has been determined, various management scenarios can be tested for resultant reduction in P load. Options such as structural changes in storm water management (such as detention or infiltration that affect selected drainage areas within the model) or reducing the internal load (which can be accomplished with oxygenation or inactivation) can be incorporated and the result on steady state lake conditions can be examined.

APPROACH:

LLRM was used to assess watershed and in-lake measures for potential reductions in P loading to Lake Garfield and possible achievement of target conditions.

Results

QA/QC Results

There were no problems with the field instruments during this study. All calibration events went smoothly, with minimal adjustment needed. The Hach DS5 is used extensively during the summer half of the year and is kept in excellent condition to avoid missing data. Oxygen calibration usually involves very minor adjustment, with values holding between $\pm$10% of the correct value for weeks at a time. Calibration needs for pH are more often significant, but that measurement was not an essential feature of this sampling program.

All 15 blank samples (distilled water placed in sample bottles at the start of the sampling day and processed like environmental samples) returned lab values below the detection limit for total or dissolved phosphorus (Table 2). The 16 duplicate samples that were actually pairs split from the same sample returned percent difference values between 0.0 and 27.3%, with an average of 10.8% (Table 3, Figure 10). Higher values were mostly related to small differences when paired samples contained P near the detection limit. The percent difference for paired dissolved P samples ranged from 0.0 to 48.1% with an average of 6.4%. Many dissolved P values were below the detection limit. QA/QC objectives were met with regard to laboratory water testing results.

Two samples in Table 3 are not true duplicates, but are rather from separate storm water samplers that filled during the same storm. The timing of the filling of each sampler is unknown and while the samples represent the same station in the same storm, they are not true duplicates. They do, however, provide some insight into the variability that can be encountered in water quality during a storm event. Both involved elevated TP values, but the difference was 4.7% for one pair of samples and 145.7% for the other. Clearly, there is high variation in storm water, and while more storm sampling was performed in this study than any other form of sampling, the results must be viewed with the knowledge that any given sample may not adequately represent storm water quality.

A duplicate sediment sample was collected and results compared favorably with the paired sample (Table 4), with percent difference among 10 paired tests ranging from 0.0 to 18.2%. Additionally, the laboratory runs its own QA/QC samples, and reported relative percent differences of 1.3 to 10.4% for duplicate tests and values for blank samples all below the detection limit. Spike recovery values ranged from 92.5 to 105.1%. Laboratory quality objectives for sediment were met.

Seepage measurements were duplicated for 6 locations, with paired measurements exhibiting percent differences of 0.0 to 77.8% with an average of 24.6% (Table 5). All seepage values were low, and higher percent differences were a function of small differences with a very low denominator. Differences of 100% would not have significantly affected conclusions drawn from seepage measurements.

The biggest QA problem was completeness, as storm water samplers do not always fill, storms occur at night and on weekends, and the field crew apparently misunderstood the need for a lake sample near the thermocline on several occasions. Additional storm water sampling was conducted to meet the targeted quantities of samples and additional lake sampling was performed in July and August to get thermocline samples in those months. Overall goals of the sampling program were met, but required follow up sampling in some cases.

Table 2. QA/QC results for blank samples

Blanks		
Date	**Parameter**	**Blank Value (mg/L)**
4/4/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
4/11/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
4/21/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
5/18/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
05/26/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
6/27/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
6/29/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
6/29/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
7/3/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
7/10/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
7/14/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
7/26/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
8/17/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
8/22/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106
09/15/17	Total Phosphorus	<0.106
	Dissolved Phosphorus	<0.106

Table 3. QA/QC data for duplicate samples

(Note: Values <MDL, which is 0.0106 mg/L, are reported here as the rounded MDL for purposes of comparison)

Duplicates						
Date	**Station**	**Parameter**	**Station Value (mg/L)**	**Duplicate Value (mg/L)**	**Difference in Values (mg/L)**	**Percent Difference**
4/4/17	H	Total Phosphorus	0.050	0.047	0.003	6.2%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
4/4/17	I	Total Phosphorus	0.030	0.032	0.002	7.0%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
4/4/17	K	Total Phosphorus	0.021	0.017	0.004	20.2%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
4/11/17	LG-S	Total Phosphorus	0.018	0.014	0.004	23.8%
		Dissolved Phosphorus	0.015	0.011	0.004	28.9%
4/12/17	D	Total Phosphorus	0.011	0.012	0.001	13.2%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
4/21/17	H	Total Phosphorus	0.017	0.021	0.004	23.5%
		Dissolved Phosphorus	0.017	0.017	0.000	0.0%
5/18/17	LG-S	Total Phosphorus	0.013	0.012	0.001	8.6%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
5/26/17	H	Total Phosphorus	0.016	0.016	0.000	0.0%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
5/26/17	L	Total Phosphorus	0.116	0.143	0.027	23.3%
		Dissolved Phosphorus	0.057	0.030	0.028	48.1%
6/27/17	F+G	Total Phosphorus	0.012	0.015	0.003	27.4%
		Dissolved Phosphorus	0.012	0.014	0.002	17.9%
6/29/17	Blank	Total Phosphorus	0.011	0.011	0.000	0.0%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
7/3/17	I	Total Phosphorus	0.011	0.011	0.000	0.0%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
7/3/17	K	Total Phosphorus	0.635	1.560	0.925	145.7%
		Second first flush sample - separate sampler from same storm				
7/3/17	L	Total Phosphorus	0.408	0.427	0.019	4.7%
		Second first flush sample - separate sampler from same storm				
7/14/17	Blank	Total Phosphorus	0.011	0.011	0.000	0.0%
		Dissolved Phosphorus	0.011	0.011	0.000	0.0%
7/26/17	LG-B	Total Phosphorus	0.200	0.177	0.023	11.5%
		Dissolved Phosphorus	0.017	0.016	0.001	6.5%
8/17/17	LG-S	Total Phosphorus	0.005	0.005	0.000	0.0%
		Dissolved Phosphorus	0.005	0.005	0.000	0.0%
08/22/17	LG-S	Total Phosphorus	0.005	0.005	0.000	0.0%
		Dissolved Phosphorus	0.005	0.005	0.000	0.0%
09/15/17	LG-S	Total Phosphorus	0.013	0.014	0.001	7.8%
		Dissolved Phosphorus	0.005	0.005	0.000	0.0%

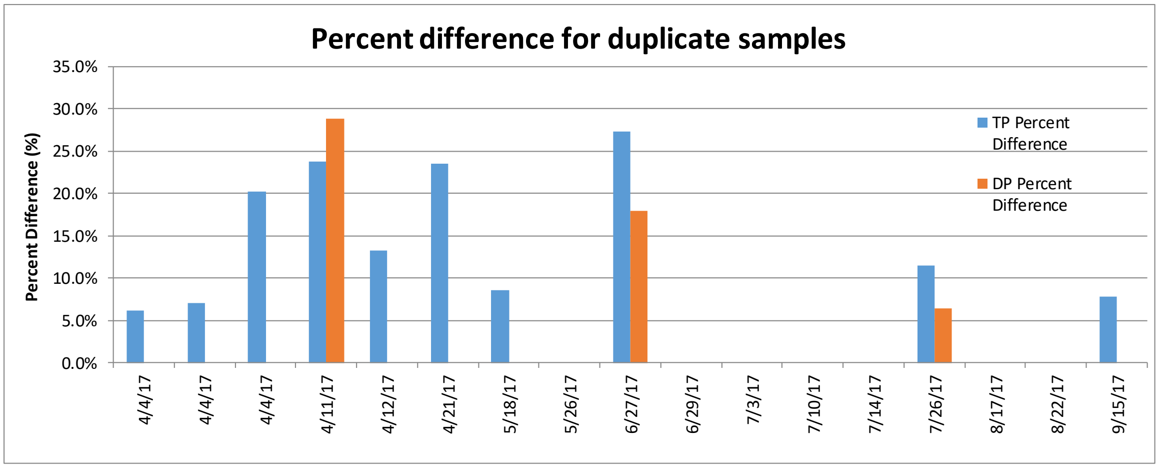


Figure 10. Percent difference for duplicate samples from Lake Garfield or tributaries

Table 4. Duplicate sediment sample results

Sample ID	Solids	Water	Total Organic Carbon	Total P	Loosely Bound P (NH4Cl extr)	Fe Bound P (Dithionate extr)	Al Bound P (NaOH extr)	Ca Bound P (HCl extr)	Biogenic P	Organic P
	%	%	%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
SED C	10.1	89.9	10.7	1227	1	148	370	255	307	454
SED F (C dupl)	10.3	89.7	11.6	1406	1	175	425	270	357	535
% Difference	2.0	0.2	8.4	14.6	0.0	18.2	14.9	5.9	16.3	17.8

Table 5. Duplicate seepage meter measurements

Lake Seepage				
Station	Total Time In Lake (hr)	Net Gain Volume (mL)	Seepage (L/sq.m/day)	% Difference
7a	3.0	175	5.6	31.4
7b	3.0	120	3.9	
8a	3.0	155	4.9	12.9
8b	3.0	135	4.3	
9a	3.2	45	1.4	77.8
9b	3.2	80	2.4	
10a	3.3	150	4.4	0.0
10b	3.3	150	4.4	
11a	3.0	200	6.4	7.5
11b	3.0	185	5.9	
15a	3.9	203	5.1	18.3
15b	4.4	190	4.1	
			Avg % Diff	**24.6**

Oxygen Profiles and Demand

Temperature has affects the oxygen content of water and was measured concurrently with oxygen at all stations on all dates where oxygen assessment was performed, along with other water quality features that could be assessed by a multi-probe sonde (Hach DS5) (Table 6, Figures 11 and 12). Temperature followed a typical seasonal pattern, with cold and relatively uniform values top to bottom in April, followed by warming and greater variation over depth. Stratification was complete by late June and was breaking down by mid-September. Defining the thermocline as the point of greatest inflection in the temperature curve, it was found at about 5.5 m in July and August, but there was a fairly gradual decline in temperature between 4 and 7 m, representing a substantial mid-depth zone (metalimnion). The upper layer, or epilimnion, extended from the surface to 4 m, while the bottom layer, or hypolimnion, was mainly the volume deeper than 7 m.

Oxygen concentrations exhibited just a slight decline with depth in April, but showed a more pronounced decline in May, although oxygen was still >2 mg/L at the bottom in the deepest part of the lake. By the end of June, oxygen was nearly depleted below a depth of 7 m and there was a strong decline between depths of 5 and 6 m, with values <2 mg/L at 6 m and values <1 m at 7 m. The pattern of strong oxygen decline with depth was maintained through the September sampling, despite the onset of thermal destratification.

Measurement of oxygen at multiple stations in August (Table 7, Figures 5 and 13) revealed a similar vertical pattern of oxygen distribution at all stations, which were at least 6 m deep. Oxygen was depressed (<2 mg/L) at 6 m and nearly depleted (<1 mg/L) by 8 m, but there was little difference between values at 6 and 7 m, suggesting some mixing on that date. The gradual decline in temperature below depths of 4 m suggests that disturbance of the deeper water is plausible, such that substances in deeper water may be transported vertically. Mixing of water much deeper than 4 m with the surface layer will be less likely, as the temperature differential (and therefore resistance to mixing) is much greater. However, light penetrates well below the 4 m depth, so algae are likely to be able to take advantage of available nutrients in deeper water and still have enough light to grow.

Considering the relation between depth and either area or volume for Lake Garfield (Figure 14), low oxygen conditions affect about 29 ha (71.5 ac) of the lake, the area deeper than 7 m (23 feet), representing 29.3% of the total lake area. At the greatest vertical extent of anoxia (defined as oxygen <1 mg/L), about 481,000 m^3 of lake volume are impacted, representing 11.1% of total lake volume. Anoxia near the sediment water interface appears to occur between mid- to late June and mid- to late September, suggesting an exposure time of about 90 days, typical for stratified Massachusetts lakes.

Spring oxygen profiles can be used to estimate oxygen demand, as describe in the QAPP. The difference between April and May profiles will be affected to some degree by vertical mixing even in deep water, as there was minimal thermal gradient early in this period. This will cause underestimate of the actual demand. Comparison of the May and June profiles represents less mixing, but by the end of the period the oxygen in deep water was <2 mg/L, at which point oxygen demand is less easily expressed and again we expect underestimation of actual demand. The results of those comparisons indicates a rate of oxygen depletion of 0.33 (April vs May) and 0.48 (May vs June) $g/m^2/d$. From experience in other Massachusetts lakes, these will represent about half the actual oxygen demand, suggesting an average of 0.8 $g/m^2/d$. Values <0.5 $g/m^2/d$ are considered fairly low, while values >1 $g/m^2/d$ are elevated and values >2 $g/m^2/d$ are very high. Lake Garfield therefore has a moderate oxygen demand.

Table 6. Field water quality data for Lake Garfield in 2017

Date	Depth	Temp	DO	DO	SpCond	pH	Turbidity	CHL-a	Secchi
M/D/YY	meters	°C	mg/l	% Sat	µS/cm	Units	NTU	µg/l	m
4/11/17	0.1	6.6	9.8	81.4	151	7.3	4.3	1.7	3.9
4/11/17	1.1	5.4	10.0	79.8	151	7.2	4.4	9.4	
4/11/17	2.0	5.3	9.9	79.2	151	7.2	4.5	11.1	
4/11/17	3.0	5.0	9.8	77.5	151	7.2	4.8	14.1	
4/11/17	4.0	4.8	9.7	76.6	152	7.1	5.0	14.1	
4/11/17	5.0	4.8	9.7	76.9	151	7.1	5.3	15.0	
4/11/17	6.0	4.7	9.6	75.9	152	7.1	5.7	15.6	
4/11/17	7.0	4.6	9.6	75.4	151	7.0	6.9	15.1	
4/11/17	8.0	4.2	9.1	70.6	156	6.8	7.0	18.2	
4/11/17	8.2	4.2	9.2	71.4	155	6.7	7.0	17.6	
4/11/17	8.6	4.2	8.8	68.7	156	6.8	8.2	21.6	
5/18/17	0.1	17.3	8.7	92.2	155	7.5	4.3	2.7	3.9
5/18/17	1.0	17.2	8.7	91.4	156	7.4	4.6	3.4	
5/18/17	2.1	15.0	8.6	86.6	153	7.3	4.6	4.1	
5/18/17	3.0	13.7	8.5	82.9	153	7.2	4.7	6.0	
5/18/17	4.1	13.2	8.0	77.2	153	7.1	4.8	8.1	
5/18/17	5.0	12.8	7.8	74.6	153	7.0	4.9	7.2	
5/18/17	6.0	12.4	6.9	65.3	153	7.0	5.0	5.9	
5/18/17	7.0	11.9	6.1	57.3	153	6.9	5.1	5.2	
5/18/17	8.0	11.7	5.4	50.5	154	6.8	5.0	5.3	
5/18/17	9.1	11.0	2.8	25.6	158	6.9	8.5	4.3	
5/18/17	8.8	11.0	2.8	25.7	158	6.9	7.2	3.5	
6/29/17	0.1	22.8	9.2	108.5	131	8.1	4.2	0.1	5.5
6/29/17	1.0	22.7	9.3	108.6	131	8.0	4.3	1.4	
6/29/17	2.0	22.5	9.3	108.2	131	7.9	4.6	2.3	
6/29/17	3.0	22.5	9.2	107.1	132	7.8	4.9	2.7	
6/29/17	4.0	21.5	9.9	113.5	130	7.6	5.6	2.3	
6/29/17	5.0	17.0	9.3	97.1	128	7.3	6.9	6.1	
6/29/17	6.0	14.3	3.1	30.5	130	7.5	6.4	9.9	
6/29/17	7.0	13.2	0.9	8.4	133	6.9	15.4	14.7	
6/29/17	8.0	11.9	0.7	6.5	137	6.7	21.2	9.7	
6/29/17	8.5	11.5	0.7	6.2	147	6.9	11.7	8.8	
6/29/17	9.1	11.2	0.6	5.9	156	6.8	14.7	5.2	
7/10/17	0.1	24.6	8.9	108.1	166	7.6	3.7	1.0	4.6
7/10/17	1.0	24.5	9.0	108.8	165	7.5	3.8	1.4	
7/10/17	2.0	24.4	8.9	108.3	165	7.4	4.0	1.4	
7/10/17	3.0	24.1	8.9	107.7	165	7.1	4.9	2.0	
7/10/17	4.0	23.3	9.2	109.8	164	6.8	5.6	2.9	
7/10/17	5.0	18.7	9.2	99.4	163	6.5	6.6	9.6	
7/10/17	6.0	15.2	2.4	24.1	165	6.2	5.7	7.0	
7/10/17	7.0	13.6	0.8	8.1	166	6.1	3.2	6.9	
7/10/17	8.0	12.2	0.7	6.3	171	5.9	19.2	11.2	
7/10/17	9.3	11.1	0.6	5.7	221	5.7	16.4	7.4	

Table 6 (continued). Field water quality data for Lake Garfield in 2017

Date	Depth	Temp	DO	DO	SpCond	pH	Turbidity	CHL-a	Secchi
M/D/YY	meters	°C	mg/l	% Sat	µS/cm	Units	NTU	µg/l	m
7/26/17	0.2	23.1	8.5	100.7	151	7.4	5.2	2.1	3.4
7/26/17	1.0	22.9	8.5	100.2	150	7.4	5.6	2.3	
7/26/17	2.0	22.7	8.4	98.7	150	7.3	6.5	3.4	
7/26/17	3.0	22.6	8.3	96.9	150	7.2	7.5	3.7	
7/26/17	4.0	22.4	8.3	96.4	151	7.0	8.9	3.6	
7/26/17	5.0	20.6	6.8	77.1	153	7.0	11.3	6.5	
7/26/17	6.0	16.0	1.9	19.8	154	6.9	14.8	5.7	
7/26/17	7.0	14.1	1.2	11.5	153	6.9	22.0	8.0	
7/26/17	8.0	12.4	0.6	5.9	170	6.8	30.1	9.5	
7/26/17	9.0	11.4	0.6	5.7	210	6.6	28.7	11.7	
8/17/17	0.1	23.7	8.7	103.7	136	8.2	3.8	1.4	4.8
8/17/17	1.0	23.4	8.7	103.1	136	8.1	4.0	6.8	
8/17/17	2.0	23.3	8.6	102.1	136	7.7	4.9	2.1	
8/17/17	3.0	23.2	8.5	100.5	136	7.5	5.6	2.3	
8/17/17	4.0	22.9	8.3	97.9	136	7.0	6.9	2.4	
8/17/17	5.0	20.6	6.6	74.0	137	6.5	9.1	6.7	
8/17/17	6.0	17.0	2.4	24.7	138	6.4	11.2	5.1	
8/17/17	7.0	14.2	2.3	23.1	140	6.4	15.6	10.5	
8/17/17	8.0	12.1	0.6	5.6	184	6.2	15.4	4.3	
8/17/17	8.8	11.7	0.6	5.6	204	6.2	32.0	15.1	
9/15/2017	0.2	21.2	9.9	112.5	155	7.5	9.7	1.8	3.1
9/15/2017	1.0	20.8	9.7	109.5	155	7.2	11.4	2.8	
9/15/2017	2.0	19.8	9.9	109.3	154	6.9	14.2	6.6	
9/15/2017	3.0	19.2	10.1	110.9	153	7.4	10.3	6.7	
9/15/2017	4.0	18.9	8.6	93.5	153	6.7	15.9	8.2	
9/15/2017	5.0	18.7	8.2	88.6	153	6.6	17.5	8.4	
9/15/2017	6.0	18.4	5.7	61.2	153	6.4	23.7	5.4	
9/15/2017	7.0	17.2	0.9	9.1	156	6.3	28.7	5.6	
9/15/2017	8.0	15.2	0.8	8.2	178	6.2	36.0	3.9	
9/15/2017	9.0	12.9	0.8	7.8	251	6.3	44.6	3.9	

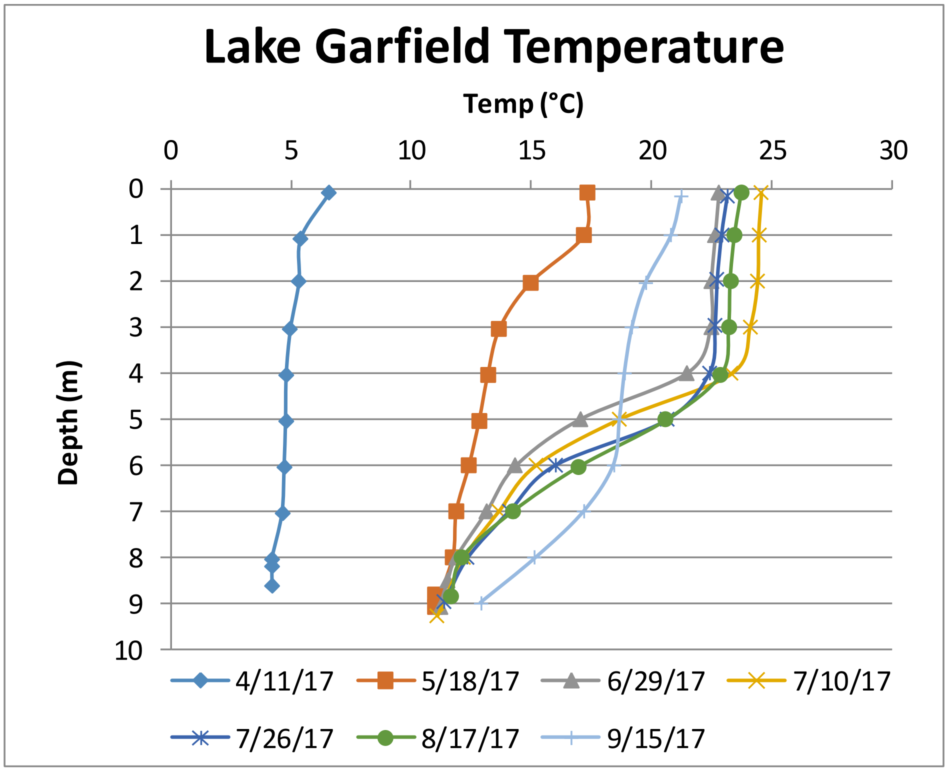


Figure 11. Temperature profiles for Lake Garfield in 2017

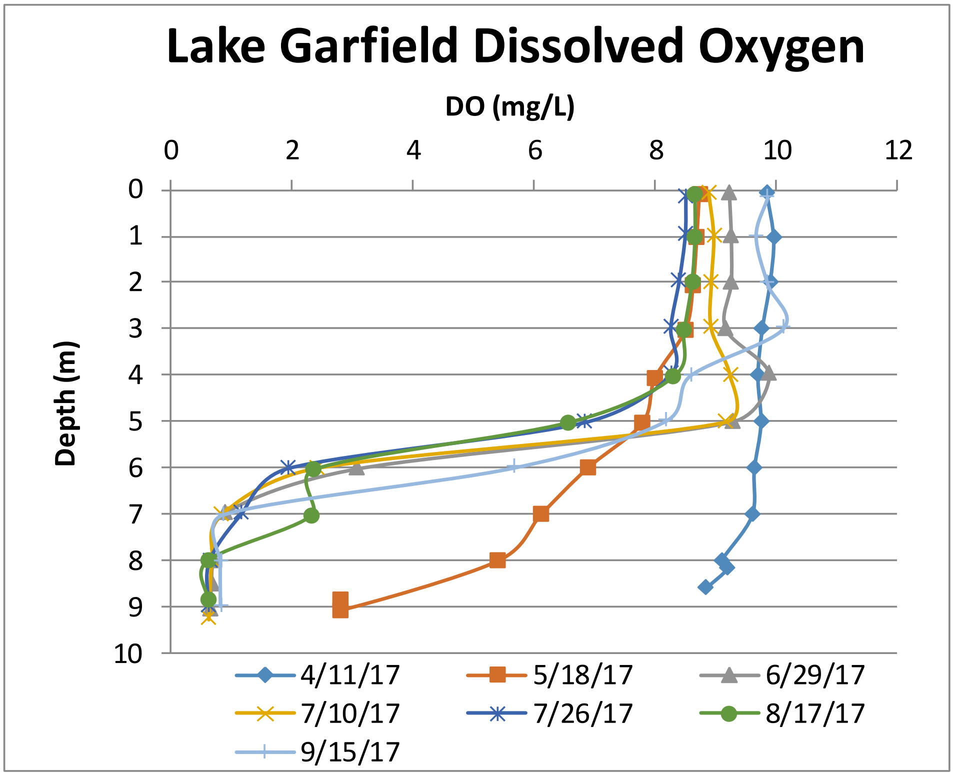


Figure 12. Oxygen profiles for Lake Garfield in 2017

Table 7. Field water quality data for Lake Garfield on August 17, 2017

	Date	Depth	Depth	Temp	DO	DO	SpCond	pH	Turbidity	CHL-a
Station	M/D/YY	meters	feet	°C	mg/l	% Sat	µS/cm	Units	NTU	µg/l
Oxy 1	8/17/17	0.0	0.0	24.3	8.5	102.8	137	8.3	2.0	1.3
	8/17/17	2.0	6.7	23.4	8.4	99.5	136	8.1	2.2	1.6
	8/17/17	4.0	13.2	23.2	7.3	86.4	136	7.8	2.3	2.7
	8/17/17	6.0	19.7	18.4	3.2	34.7	138	7.5	2.9	3.8
	8/17/17	6.5	21.3	16.7	1.0	10.7	139	7.6	4.7	3.9
Oxy 2	8/17/17	0.0	0.0	24.6	8.5	103.3	136	8.0	5.2	2.0
	8/17/17	2.0	6.7	23.4	8.4	100.0	136	7.8	6.3	1.5
	8/17/17	4.0	13.2	23.2	7.3	86.1	138	7.3	8.4	2.2
	8/17/17	6.0	19.8	17.9	2.1	22.8	139	7.0	11.9	5.2
	8/17/17	7.0	23.1	14.8	1.7	16.9	140	7.1	16.5	5.4
	8/17/17	7.5	24.5	13.6	0.7	6.4	136	7.1	18.9	14.8
	8/17/17	7.8	25.7	12.5	0.7	6.2	182	7.1	19.3	5.8
Oxy 3	8/17/17	0.0	0.0	24.9	8.6	105.5	137	8.4	2.5	1.1
	8/17/17	2.0	6.7	23.4	8.5	101.3	138	8.3	2.6	1.9
	8/17/17	4.0	13.2	23.2	8.4	99.4	136	8.1	2.9	2.3
	8/17/17	6.0	19.7	18.2	4.2	44.9	138	7.5	4.4	6.7
	8/17/17	7.0	23.0	14.9	4.3	42.9	138	7.5	8.6	7.2
	8/17/17	7.5	24.6	13.1	0.8	8.0	155	7.6	25.4	10.1
Oxy 4	8/17/17	0.0	0.0	25.1	8.6	105.0	137	8.6	2.2	1.2
	8/17/17	2.1	6.7	23.4	8.5	101.3	137	8.5	2.2	2.1
	8/17/17	4.1	13.3	23.1	7.5	88.3	137	8.2	2.5	2.4
	8/17/17	6.1	20.0	18.2	2.6	27.7	139	8.0	2.3	4.9
	8/17/17	6.2	20.4	17.3	2.3	24.3	139	8.1	2.1	5.1
	8/17/17	6.6	21.5	16.8	2.6	26.9	139	8.3	1.5	5.0
Oxy 5	8/17/17	0.0	0.0	24.8	8.7	105.9	137	8.6	2.9	2.3
	8/17/17	2.1	6.9	23.3	8.6	102.2	136	8.5	3.4	2.3
	8/17/17	4.0	13.1	23.1	7.8	92.7	136	8.0	4.7	3.0
	8/17/17	4.0	13.2	23.1	8.3	97.7	136	8.3	4.0	3.5
	8/17/17	6.0	19.8	18.4	4.0	42.7	138	7.8	6.9	5.8
	8/17/17	6.1	19.9	18.4	4.0	43.4	138	7.8	9.4	5.5
	8/17/17	7.0	23.1	14.9	4.1	40.6	139	7.9	15.2	10.7
Oxy 6	8/17/17	0.0	0.1	24.7	8.6	104.9	137	8.6	2.5	1.2
	8/17/17	2.0	6.7	23.4	8.5	101.4	136	8.5	2.7	1.8
	8/17/17	4.0	13.1	23.1	7.9	93.5	136	8.2	3.0	2.4
	8/17/17	6.0	19.8	17.8	2.7	28.7	138	7.9	3.3	4.9
	8/17/17	6.4	21.0	16.4	2.0	20.6	139	8.0	2.6	4.5
	8/17/17	6.7	22.0	15.5	2.7	26.9	139	8.0	3.5	5.1
Deep Hole	8/17/17	0.0	0.0	23.7	8.7	103.7	136	8.2	3.8	1.4
	8/17/17	1.0	3.3	23.4	8.7	103.1	136	8.1	4.0	6.8
	8/17/17	2.0	6.6	23.3	8.6	102.1	136	7.7	4.9	2.1
	8/17/17	3.0	9.9	23.2	8.5	100.5	136	7.5	5.6	2.3
	8/17/17	4.0	13.2	22.9	8.3	97.9	136	7.0	6.9	2.4
	8/17/17	5.0	16.5	20.6	6.6	74.0	137	6.5	9.1	6.7
	8/17/17	6.0	19.8	17.0	2.4	24.7	138	6.4	11.2	5.1
	8/17/17	7.0	23.0	14.2	2.3	23.1	140	6.4	15.6	10.5
	8/17/17	8.0	26.2	12.1	0.6	5.6	184	6.2	15.4	4.3
	8/17/17	8.8	29.0	11.7	0.6	5.6	204	6.2	32.0	15.1

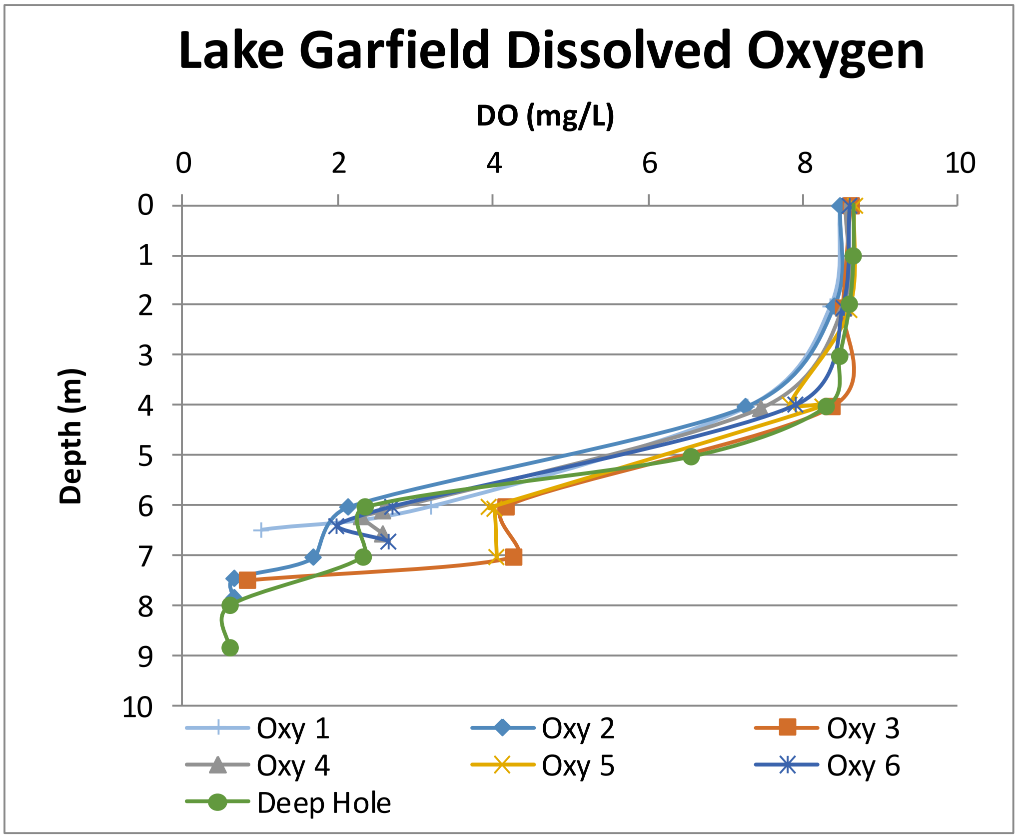


Figure 13. Oxygen profiles for Lake Garfield on August 17, 2017

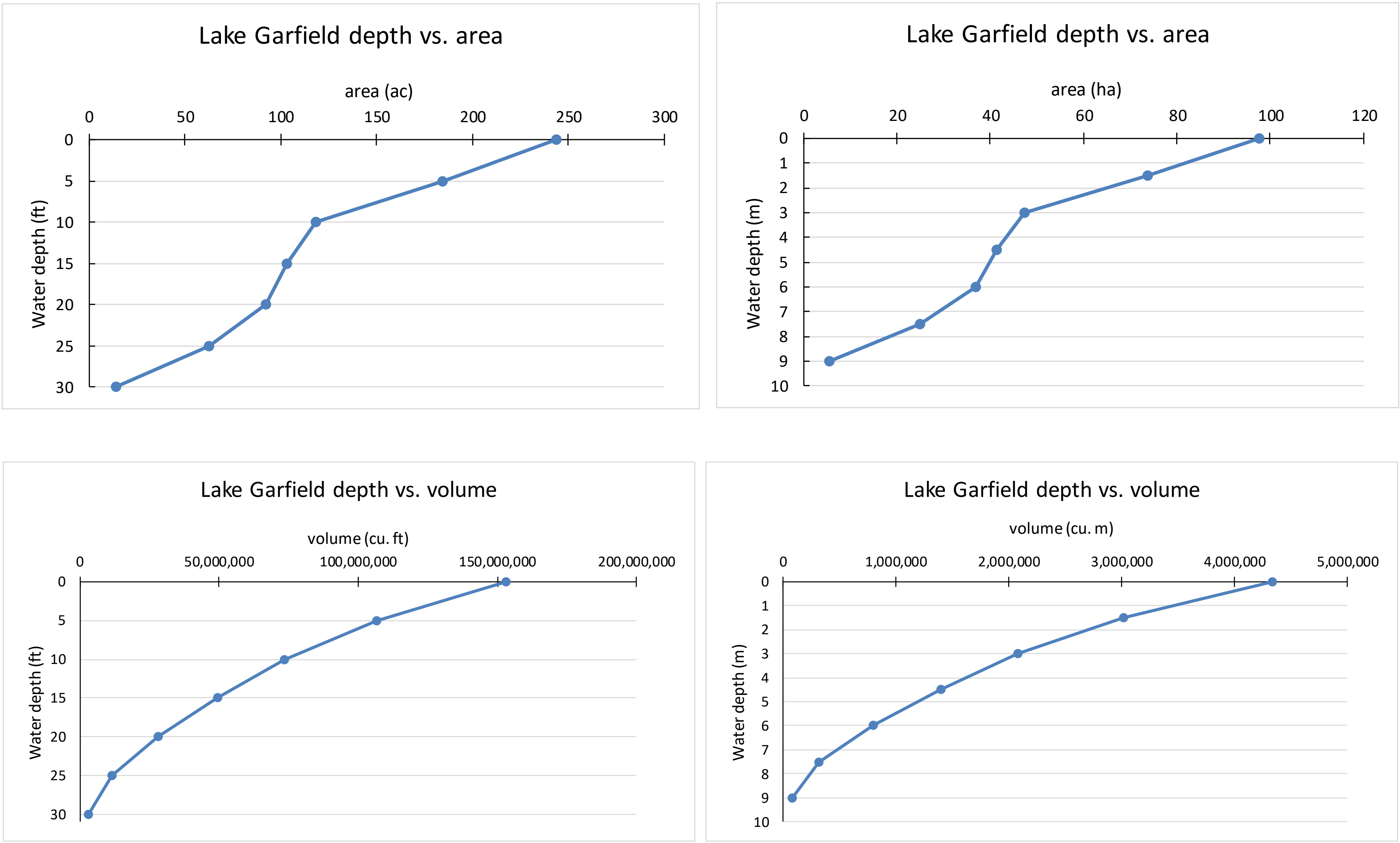


Figure 14. Depth vs area and volume for Lake Garfield

Additional Water Quality Profiles

Water quality features of pH, conductivity, turbidity and chlorophyll-a were assessed when performing temperature-oxygen profile measurements. While not essential to this investigation, they are useful data and are incorporated as background here (Tables 6 and 7, Figures 15-18). Secchi disk transparency is a highly relevant measurement of water clarity and was also measured when performing monthly oxygen assessments. Results are provided in Table 6 and Figure 19.

The pH is a measure of hydrogen ion concentration, expressed as the reciprocal of the logarithm of that concentration, such that lower values mean higher hydrogen concentration, which means that the water is more acidic. Higher pH values mean lower hydrogen concentrations, or more basic conditions, with a pH of 7 considered neutral. Berkshire lakes typically have a slightly basic natural pH, at least near the surface, but stratified lakes can accumulate decomposition products in deeper water, lowering that pH into the acidic range. Lake Garfield was typical, with surface pH values of 7.3 to 8.2 declining to 5.7 to 7.0 near the bottom in the deepest area (Figure 15). The pH can be raised considerably by algae, which increase pH through photosynthetic activity; this happens to some extent in Lake Garfield but there is no clear indication of extreme impact in this regard.

Conductivity represents the dissolved solids in water. It does not indicate the nature of those solids, but the total is linked to electrical conductance and is fairly easy to measure. Conductivity values <100 are considered low, and usually indicate low fertility as well. Values >400 are unusual in Massachusetts and suggest excessive dissolved solids, usually salt of some kind. Values for Lake Garfield ranged from about 130 to 165 µS in surface water, and were fairly consistent to a depth of 7 m (Figure 16). However, at maximum depth during the period of stratification the conductivity increased to >200 µS, suggesting the substances were being released from the sediment under anoxia and into the water column. This could include P as well as other contaminants, and is generally an undesirable situation, although the conductivity values in Lake Garfield are not extreme.

Turbidity is a measure of light scattering by suspended particles in the water, which include algae and non-living particles suspended by wind or other mixing action. Values of <1 NTU are considered low and indicative of very clear water, while values >3 NTU start to indicate potentially undesirable accumulations of solids from a drinking water perspective and values >10 NTU suggest lowered clarity that will be quite noticeable to swimmers and other lake users. Turbidity in Lake Garfield is typically around 5 NTU in the upper waters, but increases with depth and is >10 NTU in water >3 m (10 feet) deep, often >20 NTU (Figure 17). This suggests an accumulation of particles in deeper water, mostly likely settled from upper waters, but not dense enough to settle completely as the deeper water is colder and therefore denser. Such particles are most likely to be algae or organic matter from the watershed (leaf fragments), which have a low specific gravity and do not settle easily; these organic particles contribute substantially to oxygen demand.

Chlorophyll-a is a photosynthetic pigment common to all algae and higher plants. Measured as fluorescence in the lake by the instrument, it is indicative of algae abundance in the water column. Values <4 µg/L are generally considered low, while values >10 µg/L are considered elevated and values >20 µg/L are usually taken as indication of a bloom. However, fluorescence measures are compromised by natural light in shallow water and settled organic matter can fluoresce in the same wavelengths as chlorophyll-a in deep water, so one must exercise caution in the interpretation of field data. For Lake Garfield, there is a substantial increase over depth (Figure 18), but the caveats on shallow and deep measures apply. Chlorophyll-a between 2 and 5

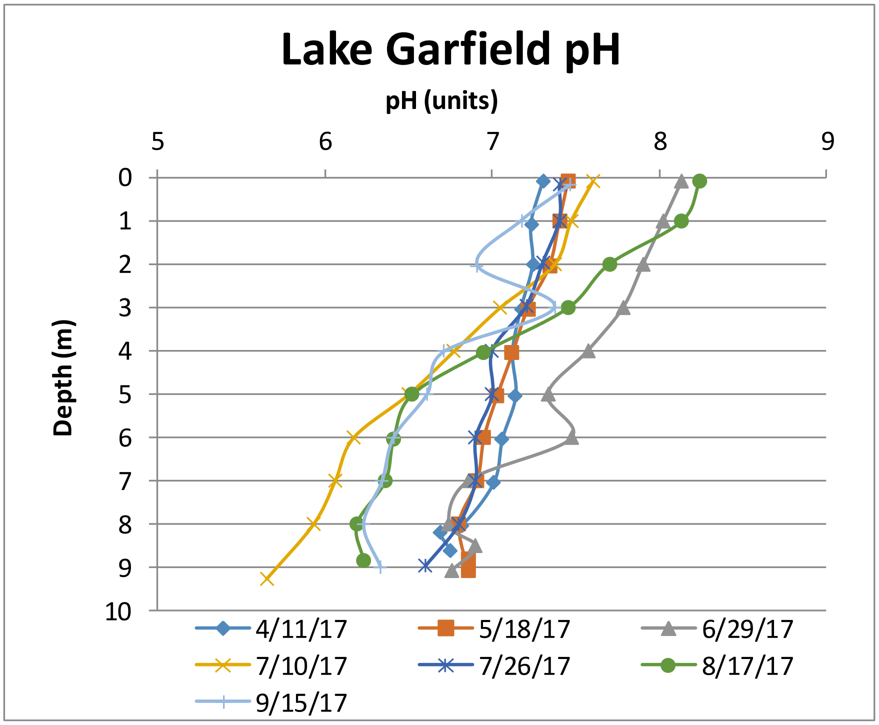


Figure 15. Lake Garfield pH

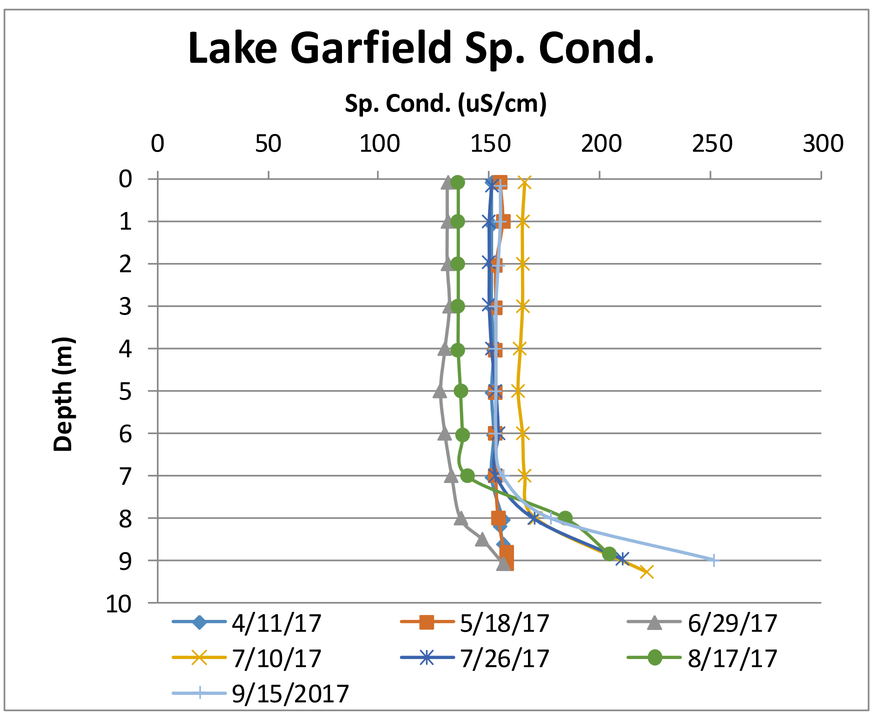


Figure 16. Lake Garfield specific conductivity

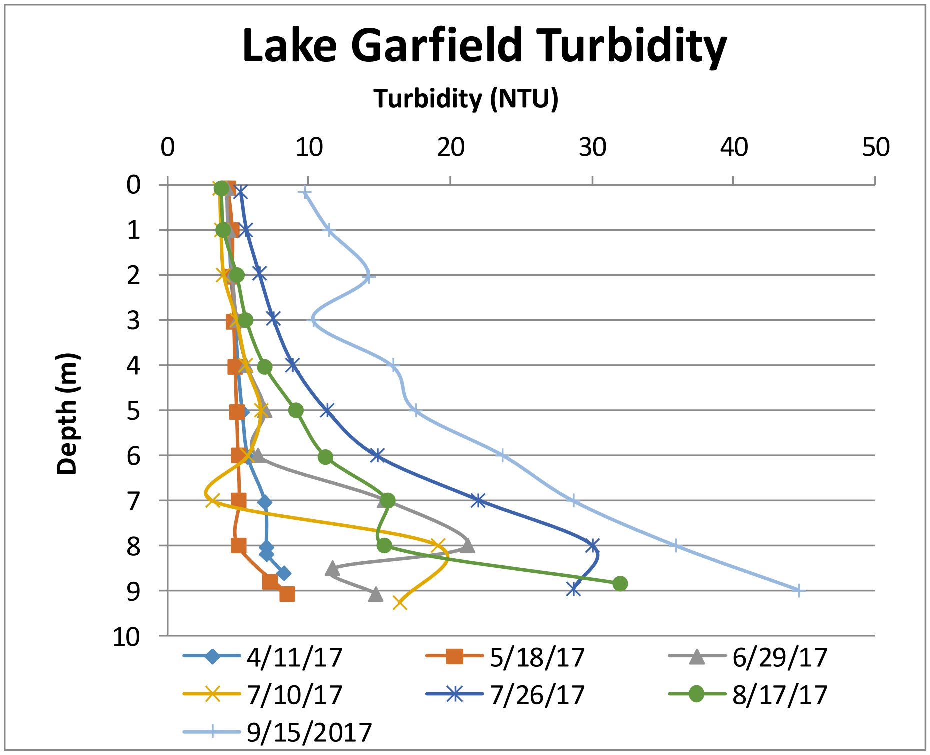


Figure 17. Lake Garfield turbidity

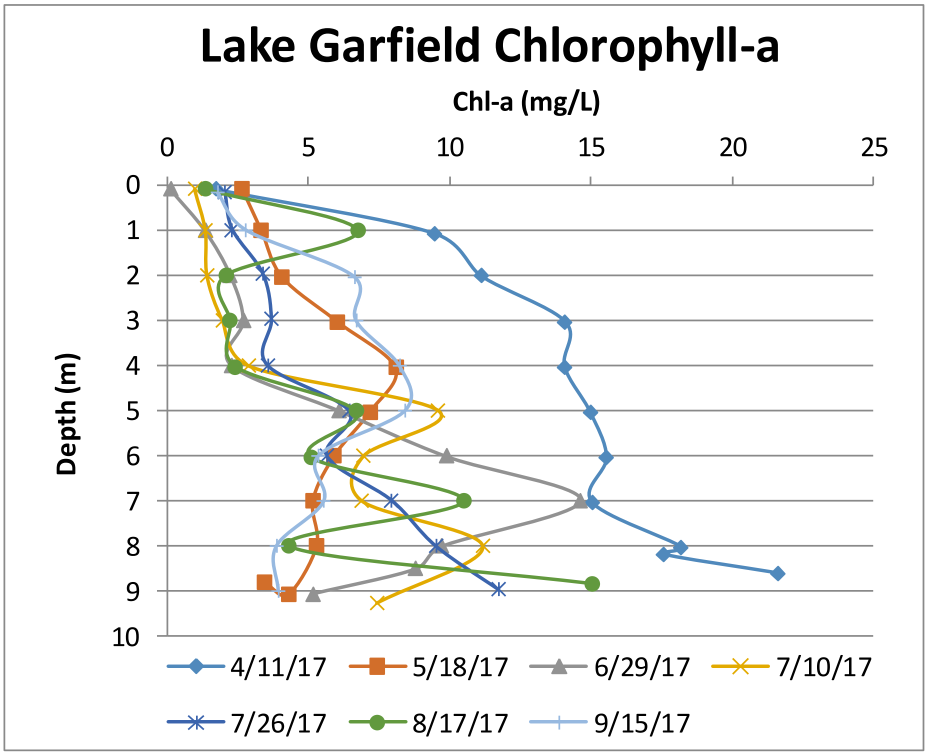


Figure 18. Lake Garfield chlorophyll-a

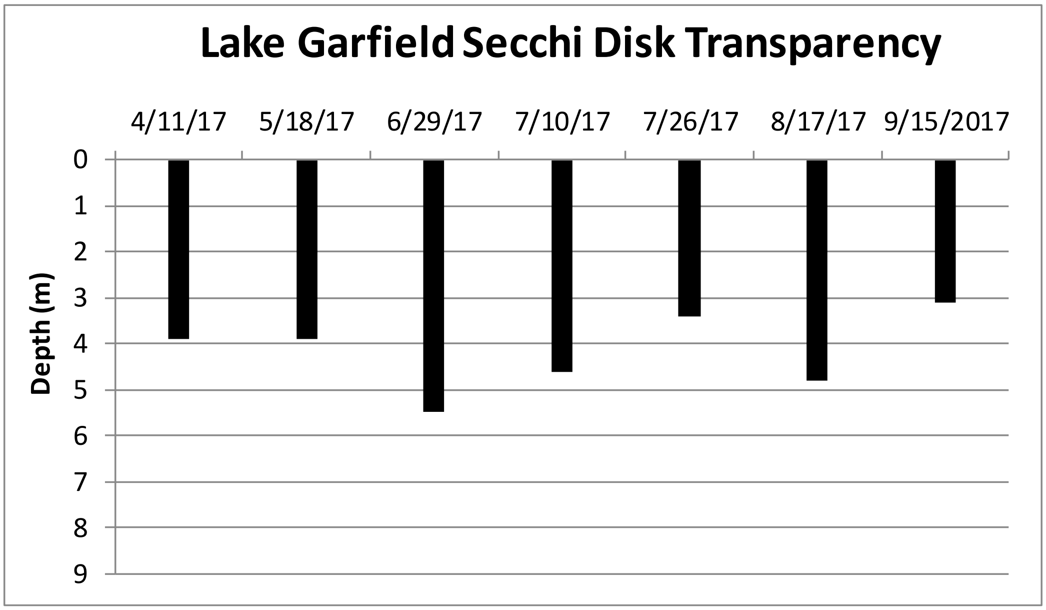


Figure 19. Lake Garfield Secchi disk transparency

meters of depth range from about 3 to 8 µg/L on all dates except the April sampling, which exhibited values of 11 to 15 µg/L. The lake was experiencing elevated densities of golden algae in April and into May. Those cold water algae declined as the water warmed and other types of algae became dominant in June and beyond. However, the ratio of algae biomass to chlorophyll-a is not constant among algal groups, and is highest for cyanobacteria, which were present during summer. The more moderate summer chlorophyll-a values are therefore not necessarily indicative of acceptable conditions. More discussion can be found in the plankton section of this report.

Secchi disk transparency is simply the depth to which the round disk with alternating black and white quadrants can be seen, but it has been related to so many lake features that it is an extremely useful measurement that can be obtained for minimal cost. Values >5 m suggest clear water in southern New England, with values >7 m fairly rare. Values <2 m suggest reduced clarity that most users will notice, and values <1.2 m (4 feet) used to be used in Massachusetts as a legal means to close swimming areas due to low visibility and compromised safety. It remains a non-enforced guideline. Secchi disk transparency in Lake Garfield (Figure 19) was just under 4 m in April and May when golden algae were blooming, and increased in June to the highest observed value of 5.5 m during transition of the algal community. Clarity declined through July with growth of cyanobacteria, including forms that are buoyant and concentrate near the surface, greatly reducing visibility. Secchi disk transparency increased in August and decreased again in September to the lowest observed value of 3.1 m, all seemingly related to shifts in the types and vertical distribution of the algal community. More discussion is provided in the algae section of this report.

In-Lake Phosphorus

In-lake total phosphorus (TP) concentrations <0.01 mg/L (10 µg/L) are indicative of low fertility and will not support algae blooms for any substantial length of time, while concentrations >0.02 mg/L can support blooms. This is a fairly narrow transition range, making P management a top priority for lakes. TP concentrations near the surface of Lake Garfield ranged from a high of 0.018 mg/L in April to a low of <0.0106 mg/L (the detection limit) in July and August, with a slight increase in September to 0.013 mg/L (Table 8, Figure 20). This suggests that the refill of the lake after winter drawdown, completed shortly before the April sampling, results in slightly elevated TP. Yet as algae generated with that P settle out and runoff declines in summer, TP concentrations in the upper water layer declines. The slight increase in September appears to relate to the onset of mixing, with much higher TP concentrations observed in deeper water. Dissolved phosphorus (DP) concentration was below the detection limit in all surface samples after April.

The TP concentrations from samples collected near the bottom of the lake were much higher than surface values from May through September. As the lake stratified, TP built up in the hypolimnion. From the data, it appears that the increase is a mix of particulate and dissolved forms. This suggests that both settling of organic particles from the epilimnion and either release of DP from the sediment or decay of the settled particles contribute to the hypolimnetic TP increase. Values generally increased in deep water throughout the study, with a slight decline in early July, possibly due to storm-induced mixing, and a decline in September that is undoubtedly a result of mixing as the lake destratified.

In addition to the deep hole station, 5 more surface stations were sampled on June 29, 2017, representing a horizontal gradient from the shallow southern end, through the deeper central area, to the shallow northern end near the outlet under Tyringham Road (Figure 6). TP concentrations (Table 9, Figure 21) were similar at the deep hole and the 3 other stations over deep water, at values close to the detection limit of 0.01 mg/L. Values exceeded 0.02 mg/L at stations 1 and 5, the geographic ends of the gradient, where the water was much shallower. All DP values were <0.01 mg/L, suggesting that the higher TP values at the shallow water stations were probably a result of suspended sediment.

Samples collected near the thermocline were intermediate in TP content to the surface and bottom samples, indicating that the bottom waters are not well mixed, but that enough P from deeper water is present at depths where light is sufficient to support algae growth. A common mode of cyanobacteria bloom formation involves growth near the thermocline or at the sediment-water interface near where the upper and lower water layers merge, with formation of gas pockets in cells after sufficient P has been taken into cells and a rise of fully formed filaments or colonies to the surface. As the surface P concentrations are not adequate in Lake Garfield to support extended algae growth near the surface, it is likely that observed cyanobacteria blooms originated in deeper water or at the sediment-water interface in water of moderate depth (4-7 m).

The average TP concentration in the epilimnion of Lake Garfield was 0.012 mg/L, with DP less than the detection limit of 0.01 mg/L. The average hypolimnetic TP concentration during the late June through early September period of stratification was 0.246 mg/L, more than 20 times higher than near the surface. While the watershed is the ultimate source of P to Lake Garfield and is very likely responsible for the higher spring TP surface concentration, the internal cycling of P appears very important in summer algal dynamics.

Table 8. In-lake phosphorus measurements for Lake Garfield

Station	Parameters mg/L	4/11/17	5/18/17	6/29/17	7/10/17	7/26/17	8/17/17	8/22/17	9/15/17
Surface	Total Phosphorus	0.018	0.013	0.012	<0.0106	<0.0106	<0.0106	<0.0106	0.013
	Dissolved Phosphorus	0.015	<0.0106	<0.0106	<0.0106	<0.0106	<0.0106	<0.0106	<0.0106
Thermocline	Total Phosphorus					0.026		0.062	0.021
	Dissolved Phosphorus					<0.0106		<0.0106	0.012
Bottom	Total Phosphorus	0.027	0.053	0.124	0.084	0.200	0.406	0.414	0.109
	Dissolved Phosphorus	<0.0106	0.014	0.124	<0.0106	0.017	0.119	0.106	0.011

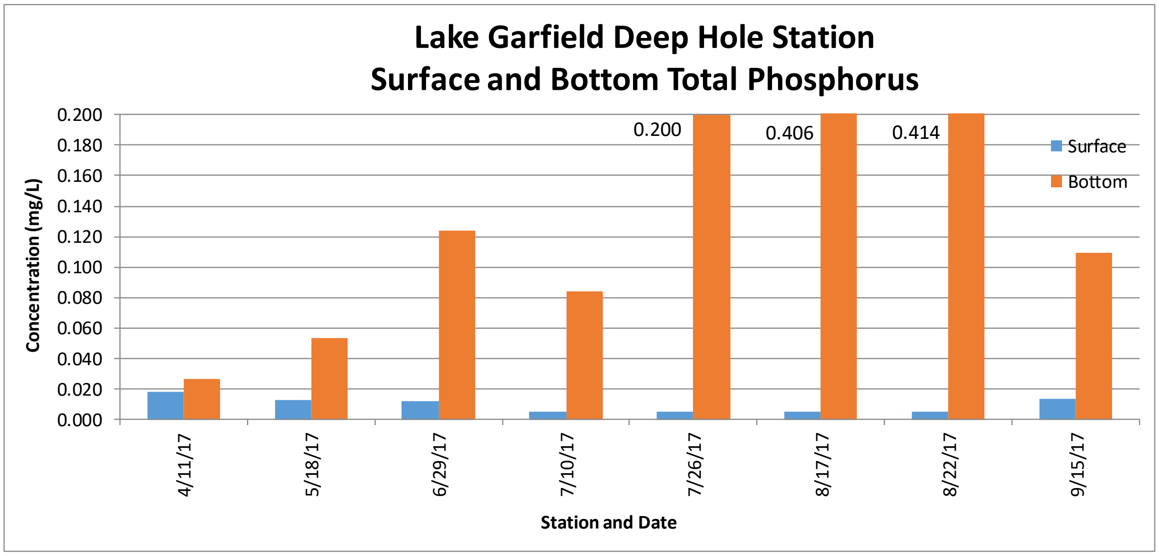


Figure 20. In-lake phosphorus measurements for Lake Garfield

Table 9. Horizontal variation of phosphorus in Lake Garfield on June 29, 2017

In-Lake Stations	Parameters mg/L	Deep Hole	LG-1	LG-2	LG-3	LG-4	LG-5
6/29/17	Total Phosphorus	0.0117	0.0266	<0.0106	0.0117	<0.0106	0.0244
	Dissolved Phosphorus	<0.0106	<0.0106	<0.0106	<0.0106	<0.0106	<0.0106

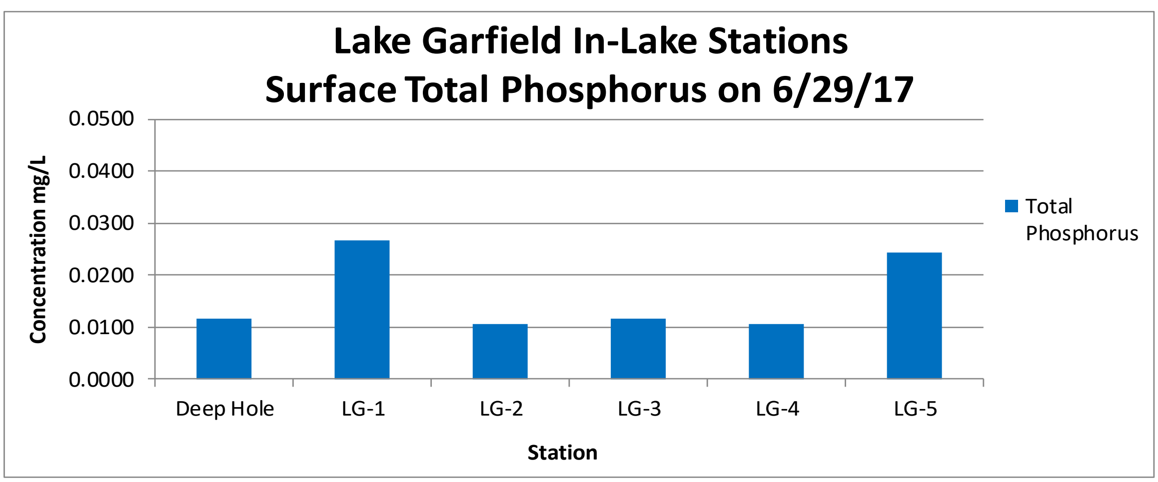


Figure 21. Horizontal variation of phosphorus in Lake Garfield on June 29, 2017

Surface Inflow Phosphorus

The least studied aspect of Lake Garfield water quality has been its inflows. Reconnaissance of the watershed in early 2017 revealed many possible input points, but only a few channels had any significant flow. As winter transitioned to spring, sampling commenced and continued exploration of the watershed revealed more possible stations with flow, but also a few of the identified stations proved to be active only during snow melt. The final set of sampling stations (Table 1, Figure 7) was deemed representative of surface water inputs to Lake Garfield.

Sampling runoff in a meaningful manner is challenging, especially with a limited budget, but this aspect of P loading is one of the most variable modes possible, and requires substantial effort to properly characterize the associated inputs. The intent was to capture at least 3 storm events, with a pre-storm, first flush, and post-storm sample for each defined input point, but as described in the QAPP, it was expected that not all samples could be collected for all targeted storms. Ultimately, we sampled snow melt on 3 dates and 5 storm events for a total of 88 inflow samples, exclusive of duplicates. The results (Table 10, Figure 22) are insightful but not ideal.

TP and DP were assessed for all samples that could be delivered to the lab on the same day as collection, while only TP from acid-preserved samples was measured for samples that were collected when the lab was not open. Values less than the detection limit of 0.0106 mg/L were reported as one half the detection limit in Table 10. Particulate phosphorus (PP) was calculated as the difference between TP and DP, but this is not a trustworthy value where TP and/or DP were lower than the detection limit.

Variability over space and time is substantial, typical of surface water samples representing the range of dry and wet conditions, but there were only a few truly elevated values obtained. TP concentrations in runoff are usually >0.05 mg/L, containing sediment, leaf fragments, and other particulates. Values >0.10 mg/L are of concern, and values >0.30 mg/L are distinctly elevated. For reference, typical urban storm water TP averages between 0.30 and 0.45 mg/L based on national surveys. Of the 88 TP values for Garfield inflows, 20 (22.7%) were greater than 0.05 mg/L, 14 (15.9%) were greater than 0.10 mg/L, and 4 (4.5%) were greater than 0.30 mg/L. With over three quarters of all inflow samples at relatively low levels, the watershed is less of an issue than for many other regional lakes, but the TP load is still substantial. No station with more than 3 samples had all values <0.05 mg/L, so no drainage area would be categorically excluded as a potentially influential source based on water quality.

Considering average concentrations for forms of P under snow melt, pre-storm, first flush and post-storm conditions (Table 11, Figure 23), pre-storm concentrations were routinely low (<0.05 mg/L), snow melt and post-storm values were low to moderate (0.05 to 0.10 mg/L), and first flush levels ranged from low to high, but were mostly moderate. Pre-storm and first flush values are missing for several smaller stations which had very limited flows.

TP load is a function of concentration and flow, and while flow measurements were made when collecting samples, these were approximate and not necessarily representative of the range of flows in each drainage area, as staff was not on site during most of each storm. Only one flow >10 cfs was encountered, a value of 13.1 cfs for station H during a 1.1 inch rain event on May 26th. Stations D, F+G, H, and I produced flows >1 cfs during most storm events. Flow is generally proportional to drainage area, but there is variation in slope and drainage channel pattern that does influence flows. Drainage area E has small channels and no discrete flows were observed during storms. Drainage area K seems similar to D but had lower flow under wet and dry conditions; overall water yield may be the same, but the measured flows were not as high.

Table 10. Tributary phosphorus data

(Note: Values <MDL, which is 0.0106 mg/L, are reported as one half the MDL, or 0.0053 mg/L, for the purpose of averaging and related distributive statistics. As particulate P is not measured directly, values of 0 result when both the total P and dissolved P values are <MDL)

Station/ Drainage	P Form (all in mg/L)	Snow Melt 3/28/17	Snow Melt 4/4/17	Pre-storm/ Snow Melt 4/11/17	1st Flush 4/12/17	1st Flush 4/21/17	Post-storm 4/21/17	Pre-storm 5/18/17	1st Flush 5/26/17	Post-storm 5/26/17	Pre-storm 6/27/17	1st Flush 7/3/17	Post-storm 7/3/17	1st Flush 7/14/17	Post-storm 7/14/17
	Total Phosphorus		0.094												
	Dissolved Phosphorus		0.074												
A	Particulate Phosphorus		0.019												
	Total Phosphorus		0.015												
	Dissolved Phosphorus		0.005												
B	Particulate Phosphorus		0.010												
	Total Phosphorus		0.036	0.018	0.015	0.045	0.023	0.005	0.081	0.020	0.016	0.071			
	Dissolved Phosphorus		0.024	0.011	0.011	0.027	0.014	0.005	0.039	0.014	0.015				
C	Particulate Phosphorus		0.012	0.008	0.004	0.018	0.009	0.000	0.042	0.006	0.001	0.071			
	Total Phosphorus	0.005	0.035	0.005	0.005		0.015	0.005	0.030	0.005	0.005	0.287	0.005		
	Dissolved Phosphorus	0.005	0.005	0.005	0.005		0.005	0.005	0.023	0.005	0.005				
D	Particulate Phosphorus	0.000	0.030	0.000	0.000		0.010	0.000	0.007	0.000	0.000	0.287	0.005		
	Total Phosphorus		0.037											0.223	
	Dissolved Phosphorus		0.005											0.219	
F	Particulate Phosphorus		0.032											0.004	
	Total Phosphorus		0.023											0.279	
	Dissolved Phosphorus		0.005											0.027	
G	Particulate Phosphorus		0.018											0.252	
	Total Phosphorus	0.012		0.024	0.011	0.108	0.005	0.016	0.329		0.012			0.251	0.039
	Dissolved Phosphorus	0.005		0.005	0.005	0.012	0.005	0.005	0.011		0.012			0.123	0.017
F/G	Particulate Phosphorus	0.006		0.019	0.005	0.096	0.000	0.011	0.318		0.000			0.128	0.022
	Total Phosphorus	0.005	0.048	0.005	0.005	0.166	0.017	0.005	0.027	0.016	0.014			0.101	0.223
	Dissolved Phosphorus	0.005	0.005	0.005	0.005	0.005	0.017	0.005	0.020	0.005	0.005			0.016	0.011
H	Particulate Phosphorus	0.000	0.043	0.000	0.000	0.161	0.000	0.000	0.006	0.011	0.009			0.085	0.212
	Total Phosphorus	0.005	0.031	0.013	0.005		0.012	0.005	0.057	0.023	0.011	0.068	0.005		
	Dissolved Phosphorus	0.005	0.005	0.013	0.005		0.005	0.005	0.018	0.012	0.005				
I	Particulate Phosphorus	0.000	0.026	0.000	0.000		0.007	0.000	0.039	0.012	0.005	0.068	0.005		
	Total Phosphorus		0.048		0.016										
	Dissolved Phosphorus		0.039		0.005										
J	Particulate Phosphorus		0.009		0.011										
	Total Phosphorus		0.019	0.005	0.005		0.027	0.005	0.320	0.014		1.098			
	Dissolved Phosphorus		0.005	0.005	0.005		0.005	0.005	0.068	0.014					
K	Particulate Phosphorus		0.014	0.000	0.000		0.022	0.000	0.252	0.000		1.098			
	Total Phosphorus	0.039	0.104	0.012	0.005		0.038	0.005	0.130	0.032	0.020	0.418	0.014		
	Dissolved Phosphorus	0.005	0.016	0.011	0.005		0.030	0.005	0.044	0.016	0.014				
L	Particulate Phosphorus	0.034	0.088	0.001	0.000		0.009	0.000	0.086	0.016	0.006	0.418	0.014		
	Total Phosphorus		0.039												
	Dissolved Phosphorus		0.022												
M1	Particulate Phosphorus		0.017												
	Total Phosphorus			0.005	0.013		0.005	0.040	0.053	0.021					
	Dissolved Phosphorus			0.005	0.005		0.005	0.005	0.012	0.005					
M2	Particulate Phosphorus			0.000	0.008		0.000	0.035	0.041	0.016					

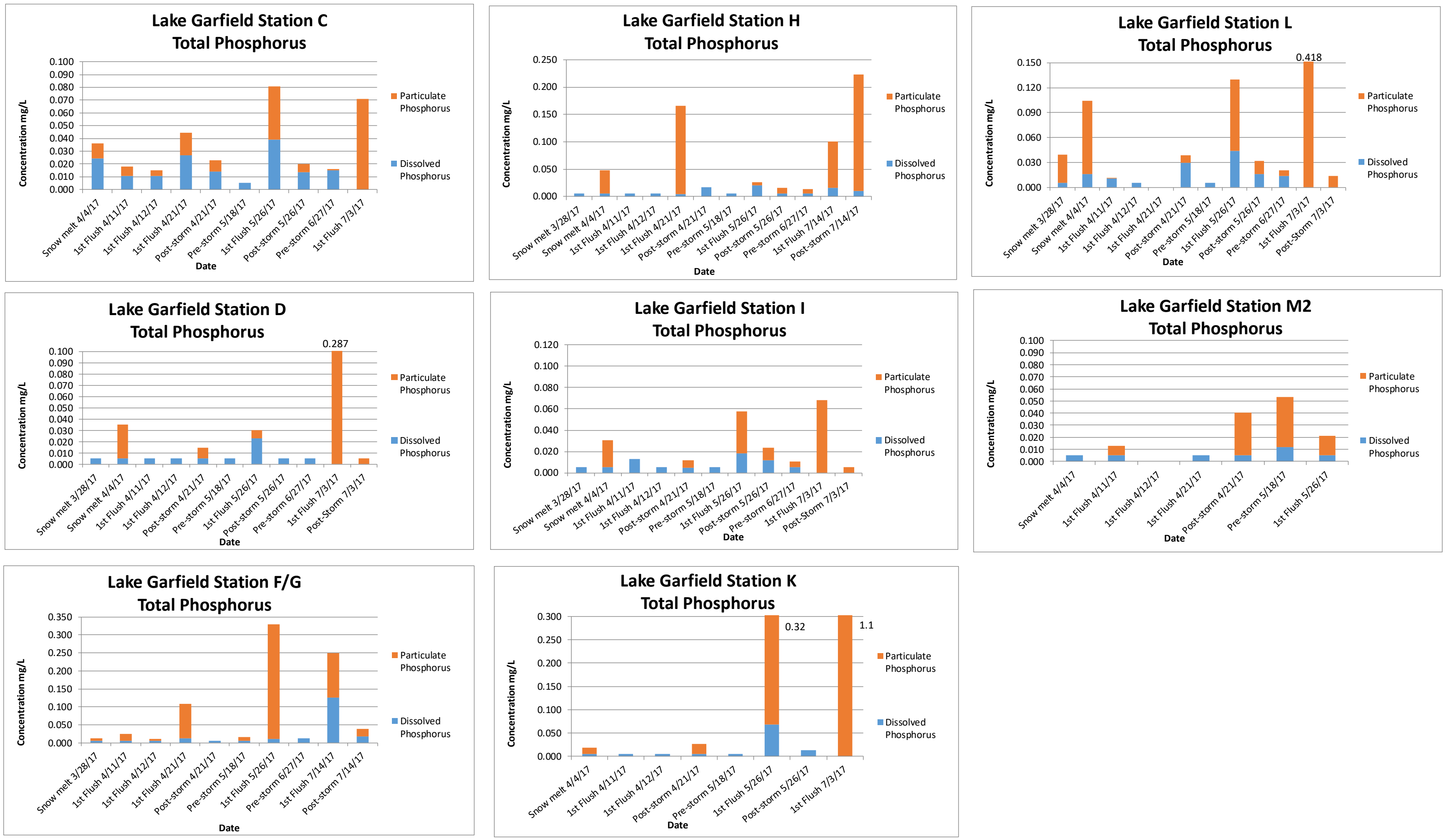
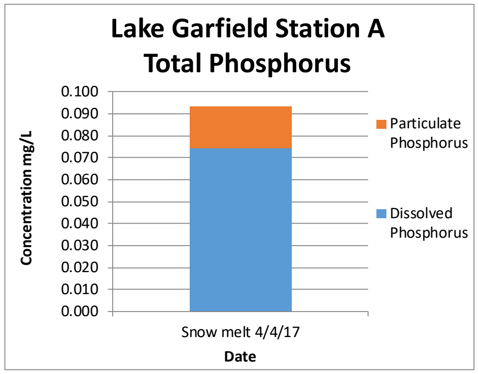
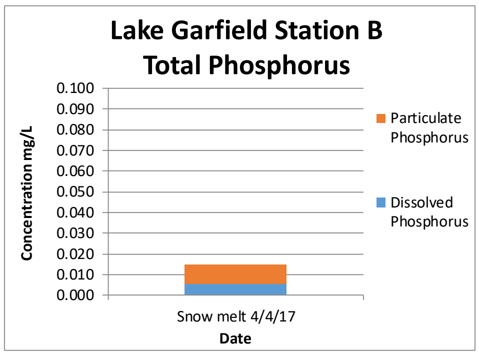
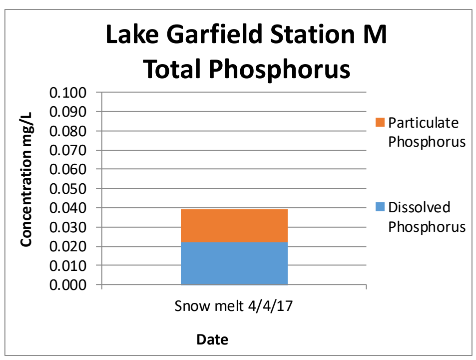


Figure 22. Tributary phosphorus data







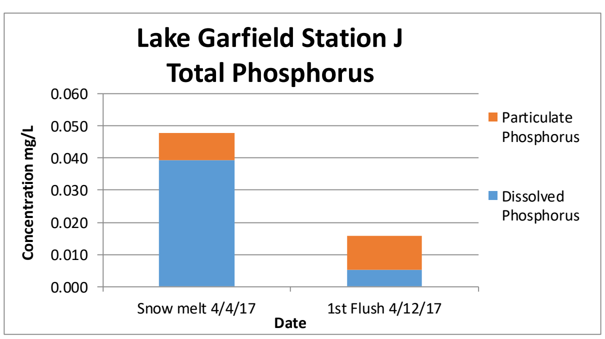


Figure 22 (continued). Tributary phosphorus data

Table 11. Average values for tributary phosphorus data

Station	Phosphorus form (mg/L)	Snow melt	Pre-storm	1st flush	Post storm
A	Total Phosphorus	0.094			0.094
	Dissolved Phosphorus	0.074			0.074
	Particulate Phosphorus	0.019			0.019
B	Total Phosphorus	0.015			0.015
	Dissolved Phosphorus	0.005			0.005
	Particulate Phosphorus	0.010			0.010
C	Total Phosphorus	0.027	0.013	0.053	0.026
	Dissolved Phosphorus	0.018	0.010	0.026	0.017
	Particulate Phosphorus	0.010	0.003	0.034	0.009
D	Total Phosphorus	0.015	0.005	0.107	0.013
	Dissolved Phosphorus	0.005	0.005	0.014	0.005
	Particulate Phosphorus	0.010	0.000	0.098	0.009
F/G	Total Phosphorus	0.037	0.017	0.200	0.023
	Dissolved Phosphorus	0.005	0.007	0.066	0.008
	Particulate Phosphorus	0.032	0.010	0.134	0.016
H	Total Phosphorus	0.023	0.008	0.075	0.062
	Dissolved Phosphorus	0.005	0.005	0.012	0.009
	Particulate Phosphorus	0.018	0.003	0.063	0.053
I	Total Phosphorus	0.018	0.010	0.044	0.015
	Dissolved Phosphorus	0.005	0.008	0.012	0.007
	Particulate Phosphorus	0.013	0.002	0.036	0.010
J	Total Phosphorus	0.020		0.016	0.048
	Dissolved Phosphorus	0.005		0.005	0.039
	Particulate Phosphorus	0.014		0.011	0.009
K	Total Phosphorus	0.016	0.005	0.474	0.020
	Dissolved Phosphorus	0.008	0.005	0.037	0.008
	Particulate Phosphorus	0.009	0.000	0.450	0.012
L	Total Phosphorus	0.048	0.012	0.184	0.045
	Dissolved Phosphorus	0.039	0.010	0.024	0.017
	Particulate Phosphorus	0.009	0.003	0.168	0.032
M1	Total Phosphorus	0.012			0.039
	Dissolved Phosphorus	0.005			0.022
	Particulate Phosphorus	0.007			0.017
M2	Total Phosphorus	0.052	0.023	0.033	0.013
	Dissolved Phosphorus	0.011	0.005	0.009	0.005
	Particulate Phosphorus	0.041	0.018	0.024	0.008

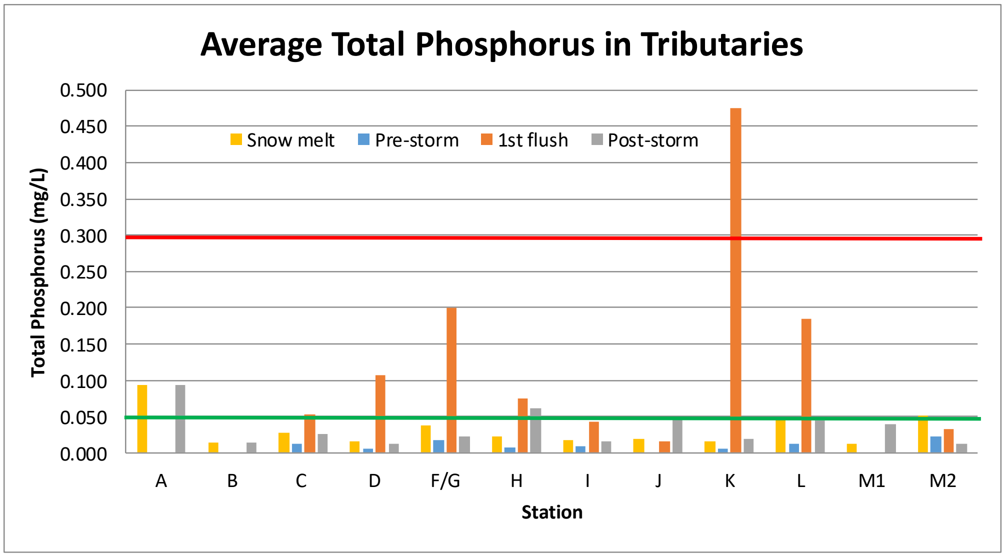


Figure 23. Average total phosphorus for tributary stations

Figure 24. Drainage basins for Lake Garfield

To estimate P loading from surface water flows, some assumptions had to be made. Drainage basins within the overall watershed were delineated based on drainage pattern, land use, and available data, combining two sampling stations in a few cases (Figure 24). The area of each defined drainage basin was determined with Google Earth Pro tools. The year was apportioned among 3 conditions based on average weather pattern: snow melt (45 days, which includes some dry and some wet weather), dry weather (260 days) and wet weather (60 days). The expected flow on the basis of yield as cubic feet per second per square mile of drainage area (cfsm) for each of the defined conditions was set in accordance with flow measurements and typical values (Dunne and Leopold 1978), with snow melt yielding 5 cfsm, wet weather yielding 4 cfsm, and dry weather yielding 0.6 cfsm. The expected average water yield for this area is 1.6 cfsm, based on extensive USGS records and the pioneering approach of Sopper and Lull (1970) and Higgins and Colonell (1971), and prorating each of the weather category yields for the portion of the year over which they occur generates a total yield of 1.54 cfsm, a close match. Applying this approach, annual flows for each defined drainage area are generated for each weather category (Table 12).

Values for TP and DP on different dates were averaged for each category of snow melt, pre-storm, first flush and post-storm samples (Table 11). We then further lumped and averaged values for stations that fell into combined drainage areas and used one half the detection limit where no data were available, which was only for dry weather values for drainage areas A+J and B. Pre-storm values were considered to be dry weather values, and first flush and post-storm concentrations were averaged to get a general wet weather value. Those values could then be multiplied by the corresponding flow values to generate estimates of TP and DP loading from each defined drainage area for each defined weather type on an annual basis, which sum to the total load for each drainage area, which in turn sums to the total from surface water to Lake Garfield (Table 12).

The estimated TP load to Lake Garfield from surface water is 153.2 kg/yr, with 89.5 kg/yr (58.5%) from wet weather, 49.4 kg/yr from snow melt (and any other flows during the snow melt period, 32.2%), and 14.3 kg/yr (9.3%) during dry weather. Top contributors were drainage areas F+G and H at 37 kg/yr (24.1% each), followed by E+M2 and I+M1 at about 20 kg/yr (13.1%) each. Drainage area C+L contributed 17.3 kg/yr (11.3%), and the other 4 drainage areas each contributed no more than 8%, collectively 14.3%. Loading follows drainage area in a general way, but not precisely; the 4 largest contributors are the 4 largest drainage basins, but within those 4 drainages, the order is not by area.

Shifts in P loading order outside of drainage basin area are a function of variation in P contribution per unit area, called the export coefficient. Dividing the total contribution by area, export coefficient estimates for each drainage basin are generated (Table 13). Mixed residential and forested uses tend to export at a range of 0.3 to 0.5 kg/ha/yr, and drainage areas F+G and K fall into that range. Other developed basins (H, A+J, C+L, E+M2) have export coefficients of 0.18 to 0.23 kg/ha/yr, which do not suggest substantial impact from surface water despite higher density of housing. Remaining basins (B, D, I+M1) have export coefficients of 0.07 to 0.14 kg/ha/yr, which are not appreciably different from values expected of totally forested basins. Overall, export coefficients do not suggest excessive impact from development in the watershed, but a priority order for improvement does emerge from this analysis.

As much of the TP load will be in particulate form and most of those particles that make it to the lake will settle soon after entry, much of the TP load will not translate directly into P concentration in the water column. The settled particulates may eventually contribute to internal loading and will certainly support rooted plant growth in shallow areas, but the direct impact on

Table 12. Estimation of phosphorus loading to Lake Garfield from surface water

			Duration			Flow/Unit Area			Flow on Annual Basis			Flow on Annual Basis			
Drainage	**Area**	**Area**	**Snow melt**	**Dry**	**Wet**	**Snow melt**	**Dry**	**Wet**	**Snow melt**	**Dry**	**Wet**	**Snow melt**	**Dry**	**Wet**	**Total**
	(ha)	**(sq.mi)**	**(days)**	**(days)**	**(days)**	**(cfsm)**	**(cfsm)**	**(cfsm)**	**(cf/yr)**	**(cf/yr)**	**(cf/yr)**	**(m3/yr)**	**(m3/yr)**	**(m3/yr)**	**(m3/yr)**
A+J	23.7	0.093	45	260	60	5.0	0.6	3.0	1799719	1247805	1439775	50984	35349	40787	127119
B	7.1	0.028	45	260	60	5.0	0.6	3.0	539156.3	373815	431325	15274	10590	12219	38082
C+L	74.2	0.290	45	260	60	5.0	0.6	3.0	5634563	3906630	4507650	159619	110669	127695	397984
D	26.1	0.102	45	260	60	5.0	0.6	3.0	1981969	1374165	1585575	56146	38928	44917	139992
E + M2	108.6	0.424	45	260	60	5.0	0.6	3.0	8246813	5717790	6597450	233621	161977	186897	582494
F+G	124.4	0.486	45	260	60	5.0	0.6	3.0	9446625	6549660	7557300	267610	185543	214088	667240
H	206.6	0.807	45	260	60	5.0	0.6	3.0	15688688	10877490	12550950	444439	308144	355551	1108134
I + M1	194.3	0.759	45	260	60	5.0	0.6	3.0	14754656	10229895	11803725	417979	289799	334383	1042161
K	26.3	0.103	45	260	60	5.0	0.6	3.0	1997156	1384695	1597725	56577	39226	45261	141064
Total	**791.3**	**3.091**							**60089344**	**41661945**	**48071475**	**1702248**	**1180225**	**1361798**	**4244271**

	Total P Concentration			Total Phosphorus Load				Diss. P Concentration			Diss. Phosphorus Load			
Drainage	**Snow melt**	**Dry**	**Wet**	**Snow melt**	**Dry**	**Wet**	**Total**	**Snow melt**	**Dry**	**Wet**	**Snow melt**	**Dry**	**Wet**	**Total**
	(mg/L)	**(mg/L)**	**(mg/L)**	**(kg/yr)**	**(kg/yr)**	**(kg/yr)**	**(kg/yr)**	**(mg/L)**	**(mg/L)**	**(mg/L)**	**(kg/yr)**	**(kg/yr)**	**(kg/yr)**	**(kg/yr)**
A+J	0.057	0.005	0.052	2.9	0.2	2.1	5.2	0.040	0.005	0.039	2.0	0.2	1.6	3.8
B	0.015	0.005	0.015	0.2	0.1	0.2	0.5	0.005	0.005	0.005	0.1	0.1	0.1	0.2
C+L	0.038	0.013	0.077	6.0	1.4	9.9	17.3	0.030	0.010	0.021	4.7	1.1	2.7	8.5
D	0.015	0.005	0.060	0.8	0.2	2.7	3.8	0.005	0.005	0.010	0.3	0.2	0.4	0.9
E + M2	0.052	0.023	0.023	12.1	3.7	4.3	20.2	0.011	0.005	0.007	2.6	0.8	1.3	4.7
F+G	0.037	0.017	0.112	9.9	3.2	23.9	37.0	0.005	0.007	0.037	1.4	1.4	7.9	10.7
H	0.023	0.008	0.068	10.2	2.5	24.3	37.0	0.005	0.005	0.011	2.4	1.6	3.7	7.7
I + M1	0.015	0.010	0.033	6.3	2.8	10.9	20.0	0.005	0.005	0.014	2.1	1.4	4.6	8.1
K	0.016	0.005	0.247	0.9	0.2	11.2	12.3	0.008	0.005	0.023	0.4	0.2	1.0	1.7
Total				**49.4**	**14.3**	**89.5**	**153.2**				**16.0**	**7.0**	**23.3**	**46.3**

Table 13. Phosphorus export coefficients for Lake Garfield drainage basins

Drainage	Area	TP	TP	DP	DP
	(ha)	(kg/yr)	(kg/ha/yr)	(kg/yr)	(kg/ha/yr)
A+J	23.7	5.2	0.22	3.8	0.16
B	7.1	0.5	0.07	0.2	0.03
C+L	74.2	17.3	0.23	8.5	0.11
D	26.1	3.8	0.14	0.9	0.03
E + M2	108.6	20.2	0.19	4.7	0.04
F+G	124.4	37.0	0.30	10.7	0.09
H	206.6	37.0	0.18	7.7	0.04
I + M1	194.3	20.0	0.10	8.1	0.04
K	26.3	12.3	0.47	1.7	0.06
Total	**791.3**	**153.2**		**46.3**	

in-lake P concentrations may be better reflected by the load of DP. Repeating the analysis above but substituting DP concentrations for TP, the lower bound of P loading is estimated (Table 12). The total DP load is 46.3 kg/yr, 30% of the TP load, reflecting the dominance of particulate P in the TP load. This is typical of both forested and urban basins in this region.

The DP load includes 23.3 kg/yr (50.3%) from wet weather inputs, 16 kg/yr (34.6%) from snow melt, and 7 kg/yr (15.1%) from dry weather flow. This skews the loading slightly away from wet weather inputs, but not greatly, as runoff inputs are still dominant, followed by snow melt and then dry weather inputs. The order of input magnitude among basins changes slightly for DP loading as well, with F+G remaining as the top contributor (23.1%), but followed by C+L, I+M1, and H (16.6-18.4%), then E+M2 and A+J (8.2-10.2%). Remaining basins (B, D, K) contribute a collective 6% of the entire DP load.

Calculating DP export coefficients for the set of drainage basins (Table 13), A+J and C+L have the highest values and are two of the most developed basins. H has perhaps the densest development in one section of that drainage basin, but much of the basin is forested and "waters down" the export coefficient for that area. F+G has the next highest DP export coefficient. E+M2 has a low export coefficient despite substantial development, but the footprints for developed areas tend to be small and limited sampling in this area reduces the reliability of that estimate. None of the export coefficients suggest extreme impact, but these estimates can be used in conjunction with the TP estimates to establish a priority order for improvements. The actual load of P to Lake Garfield that contributes directly to water column concentrations is likely to be in between the TP and DP estimates. The TP load that settles as particulates could still contribute via internal loading at some future date and will support rooted plant growths, so the entire load is potentially important, but for the purpose of this project, much of the particulate load will not directly affect in-lake P concentration.

Sediment Phosphorus

Amounts and forms of P in surficial sediment can affect transfer into the overlying waters. Total P, ranging from 1210 to 1511 mg/kg (Table 14), is considered moderate, but the fractionation among forms is very important to availability. Of particular concern is P bound to iron (Fe-P), which is easily released when oxygen approaches depletion at the sediment-water interface. An additional concern is biogenic P, the organic form most likely to be released by decay processes.

The concentration of forms of P in the surficial sediment must be adjusted according to the percent solids and specific gravity of the solids (usually about 1.1 for organic muck) to calculate the mass of P that can become available, but in general we consider concentrations of Fe-P >50 mg/kg to have some potential for impact and values >200 mg/kg to have definite impact potential. Concentrations of Fe-P in 5 Lake Garfield samples plus a duplicate sample ranged from 132 to 175 mg/kg (Table 14), a fairly tight range and moderate in magnitude. Biogenic P concentrations are harder to evaluate in terms of potential contribution, but are substantial at a range of 307 to 494 mg/kg, also a fairly tight range for this feature. Biogenic P represents almost 70% of the total organic P content on average, a large fraction.

Other P fractions include calcium-bound P (Ca-P), present at 240 to 288 mg/kg, lower than expected for a Berkshire lake, but still larger than the Fe-P fraction. Aluminum-bound P (Al-P) is slightly more abundant at 295 to 425 mg/kg. Ca-P and Al-P represent minimally available P forms in sediment. Loosely bound P, the most available fraction, is negligible.

The sediments are highly organic, as indicated by total organic carbon content in excess of 10%. Solids content is also typically low, with a range of 8.1 to 11.8% and an average of 10%. The layer of sediment that interacts with the overlying water in lakes is generally accepted as being 4-10 cm thick. Multiplying a sediment volume of 0.04 to 0.10 m^3 by the solids content times a specific gravity of 1.1, and then multiplying by the Fe-P concentration, the mass of P that could be released from iron compounds in Lake Garfield ranges from 0.6 to 2.1 g/m^2 for the area exposed to anoxia.

Only a fraction of the Fe-P mass is likely to be released in any one period of stratification, usually about 10% of the total, so we would expect that 60 to 210 mg of P could be released from each square meter of sediment exposed to anoxia. With 290,000 m^2 exposed to low oxygen over the summer, 17.4 to 60.9 kg might be released into the 481,000 m^3 hypolimnion, enough to raise the P concentration by 0.036 to 0.127 mg/L.

Applying the same procedure to the biogenic P fraction, 1.36 to 4.44 g of biogenic P are found in the upper 4 to 10 cm of sediment exposed to anoxia, and it is unlikely that >5% of this will get released in a summer season. This suggests 39.4 to 127.6 kg of P being released into 481,000 m^3, enough to raise the P concentration by 0.08 to 0.27 mg/L.

The actual increase in deep water P content is approximately 0.38 mg/L, very close to the sum of expected Fe-P and biogenic P release with a 10 cm interactive sediment depth (0.40 mg/L). The increase of 0.38 mg/L may not extend throughout the hypolimnion, however; measures of TP near at a depth of 6 m were lower, suggesting a vertical gradient through the hypolimnion, with an increase of only about 0.05 mg/L near the top of the lower water layer. This suggests an average increase of perhaps 0.22 mg/L in the hypolimnion over the summer, solidly between expectations for the 4 to 10 cm interactive sediment depth (0.12 to 0.40 mg/L).

Table 14. Sediment data for Lake Garfield from July 2017 samples

Sample ID	Solids	Water	Total Organic Carbon	Total P	Loosely Bound P (NH4Cl extr)	Fe Bound P (Dithionate extr)	Al Bound P (NaOH extr)	Ca Bound P (HCl extr)	Biogenic P	Organic P
	%	%	%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
SED A	8.4	91.6	13.5	1344	1	149	369	227	421	599
SED B	8.1	91.9	12.1	1511	1	169	363	272	494	707
SED C	10.1	89.9	10.7	1227	1	148	370	255	307	454
SED D	11.8	88.2	10.9	1229	1	159	392	240	318	437
SED E	11.4	88.6	11.2	1210	1	132	295	288	345	495
SED F (C dupl)	10.3	89.7	11.6	1406	1	175	425	270	357	535
Average	10.0	90.0	11.7	1321	1	155	369	259	374	538

An increase of 0.22 mg/L in the 481,000 m^3 hypolimnion with an area of 290,000 m^2 and an accumulation time of 90 days suggests a sediment release rate of 4 $mg/m^2/d$, a very typical value for anoxic sediment.

An increase of 0.22 mg/L in the hypolimnion suggests an internal load of 106 kg/yr, although much of that load will never reach upper waters where it could be readily converted into algae. Effectively, only a small portion of this increase will get mixed into the lake, unless there is a major windstorm (i.e., a hurricane) that mixes the lake. Yet some algae, especially cyanobacteria, can grow at very low light levels in the transition zone between epilimnion and hypolimnion, and the P available there will allow them to grow and store excess P. At the point where light limits growth after considerable P storage, many forms of cyanobacteria can form gas pockets within cells and float upward, potentially causing a bloom. The surface P measurements from this study demonstrate surface P values near the detection limit all summer, and thermal stratification keeps most of the internally generated P in the hypolimnion, but buoyant cyanobacteria were observed from late July into September.

Ground Water Seepage

Seepage measures were made at 15 stations (Figure 9), with replicate measures (back to back periods of time) for 5 of those stations (Table 15). Seepage <5 L/m2/d is considered very low, while values >40 L/m2/d are possible in very porous soils. The Berkshire region is known for less permeable soils, and low values would be expected. A seepage survey of Lake Garfield in October of 1994 (Fugro 1994) that included 10 sets of 3 seepage meters placed around the lake found seepage at levels ranging from 0 to 2.7 $L/m^2/d$ and averaged only a little over 1 $L/m^2/d$. The survey conducted in summer 2017 yielded seepage values ranging from 0 to 8 L/m2/d and averaging just over 4 $L/m^2/d$. All of these values are low and consistent with expectations for this lake.

Letting each seepage meter represent an area extending half way to the next seepage meter in each direction along shore and out to a water depth of 20 feet (beyond which muck is expected to eliminate ground water inflow), the inflow of ground water to Lake Garfield can be estimated. Total ground water input to Lake Garfield, based on the summer 2017 measurements, is slightly more than 656,000 m^3. Direct precipitation landing on Lake Garfield provides about 1.1 million m^3, while surface water inflows are estimated to provide over 4.2 million m^3, so ground water represents a relatively minor input of water.

Littoral Interstitial Porewater (LIP) samples represent the ground water entering the lake at the rate suggested by the seepage meters. P concentrations are evaluated much as for storm water; concentrations <0.05 mg/L are not of substantial concern, while values >0.10 mg/L are considered to be elevated. Data from the 1994 study yielded P concentrations ranging from <0.01 to 0.08 mg/L, low to moderate values, with an average of 0.030 mg/L. At the low rate of seepage, the load of P was not considered significant. For the 2017 study, P concentrations ranged from <0.01 to 0.09 mg/L (Table 16), with an average of 0.023 mg/L, also suggesting no strong influence on P loading to Lake Garfield.

Using the seepage and LIP sample results to get loading for each defined shoreline segment (Table 16), the range of P input is 0.0 to 3.5 kg/yr, with a lakewide total of 13.2 kg/yr. Relative to lake volume, this is a small load. The expressed concern was that on-site waste water disposal systems were contaminating the lake, and with regard to P, that is not the case.

Forms of nitrogen were also examined as part of the seepage study, and two locations did yield high values for total N concentration (stations 6 and 10 on Figure 9, concentrations in Table 16). Other locations produced moderate (0.3 to 1.0 mg/L) or even low (<0.3 mg/L) values for N. The main source of N is likely to be waste water, and N is not adsorbed to soil particles as is P, so waste water disposal may provide substantial N to Lake Garfield. Using the same approach to estimate N loading from ground water as applied for P, the total N load from ground water is 802 kg/yr, more than 60 times the load of P. This study does not address N loading from other sources, so the ground water N load cannot be put in perspective, but there is much less concern over N loading to Lake Garfield than there is over P loading. N will influence the types of algae present, but P controls the quantity of algae.

Table 15. Seepage measurements from Lake Garfield from August 2017

Lake Seepage					
Station	Water Depth (ft)	Distance From Shore (ft)	Total Time In Lake (hr)	Net Gain Volume (mL)	Seepage (L/sq.m/day)
1	1.5	2	4.2	0	0.0
2	3.0	8	3.4	285	8.0
3	2.5	10	3.2	202	6.0
4	3.0	20	2.9	70	2.3
5	3.0	9	2.7	80	2.9
6	2.0	22	2.7	60	2.1
7a	1.0	5	3.0	175	5.6
7b	1.0	5	3.0	120	3.9
8a	2.0	10	3.0	155	4.9
8b	2.0	10	3.0	135	4.3
9a	1.0	10	3.2	45	1.4
9b	1.0	10	3.2	80	2.4
10a	2.0	15	3.3	150	4.4
10b	2.0	15	3.3	150	4.4
11a	1.5	8	3.0	200	6.4
11b	1.5	8	3.0	185	5.9
12	2.0	7	4.0	40	1.0
13	2.0	4	4.1	308	7.2
14	2.5	5	3.6	160	4.2
15a	1.5	10	3.9	203	5.1
15b	1.5	10	4.4	190	4.1

Table 16. LIP sample testing results and calculation of nutrient loading from ground water

Station or Zone	TKN mg/L	N mg/L	TDN mg/L	TDP mg/L	Area m2	Seepage quantity L/m2/day	Annual seepage m3/yr	TDP mg/day	TDP kg/yr	TDN mg/day	TDN kg/yr
1						0.0	0	0	0.0	0	0.0
2	0.303	0.025	0.328	0.012	25130	8.1	24801	2367	0.9	66352	24.2
3	0.050	0.025	0.075	0.005	11325	6.0	9223	360	0.1	5096	1.9
4	0.259	0.025	0.284	0.031	10845	2.3	46436	778	0.3	7176	2.6
5	0.131	0.025	0.156	0.029	44021	2.9	29407	3651	1.3	19846	7.2
6	1.640	0.025	1.665	0.034	37825	2.1	53822	2739	1.0	134146	49.0
7	0.050	0.528	0.578	0.005	26145	4.8	13632	658	0.2	71807	26.2
8	0.106	0.025	0.131	0.015	9055	4.6	13660	620	0.2	5450	2.0
9	0.050	0.025	0.075	0.005	29061	1.9	109652	293	0.1	4152	1.5
10	0.200	6.640	6.840	0.005	48787	4.4	87779	1145	0.4	1478293	539.6
11	0.050	0.599	0.649	0.005	37577	6.2	11753	1227	0.4	150226	54.8
12	0.414	0.025	0.439	0.013	34939	1.0	72333	429	0.2	14725	5.4
13	0.050	0.542	0.592	0.048	27334	7.3	72333	9473	3.5	117319	42.8
14	0.050	0.581	0.631	0.091	14777	4.2	22653	5673	2.1	39161	14.3
15	0.322	0.025	0.347	0.028	52944	4.6	88873	6720	2.5	84490	30.8
TOTAL					**409764**		**656358**		**13.2**		**802.4**

Plankton

Phytoplankton form the base of the aquatic food web, and are essential to supporting a desirable fishery, but with excess nutrients, algae can grow faster than they are consumed and accumulate biomass, causing blooms. Some phytoplankton are less edible than others, notably many cyanobacteria, and there are various strategies employed by different algal groups to gain advantage. Diatoms and golden algae metabolize oils, which is most efficient at colder temperatures, making them the more likely group to dominate in early spring after the ice goes off the lake. Cyanobacteria store food mostly as sugars, metabolized most efficiently at higher temperatures, making them more of a threat to bloom in summer. The most common pattern of algal succession in lakes involves dominance by diatoms and golden algae in spring, giving way to green algae, which yield to cyanobacteria later in summer. But this pattern can be disrupted by weather and nutrient ratios, which much like choice of food storage, affect the fitness of different algae groups.

The phytoplankton of Lake Garfield has been known to contain all the major algae groups from sporadic samples and observations over at least two decades, but a more complete assessment has been lacking. The phytoplankton of 2017 (Table 17, Figure 25) was dominated by the golden alga (Chyrsophyta) *Dinobryon* in April and May at elevated biomass. Algae biomass at <1000 µg/L is generally considered low, while values >3000 µg/L are elevated and values >10,000 µg/L are extreme. There were few green algae (Chlorophyta), a group that prefers intermediate temperatures and high available N concentrations at any time in 2017. Diatoms (Bacillariophyta) were a minor component of the phytoplankton in all samples. Dinoflagellates (Pyrrhophyta) were present but not dominant in June through August. The phytoplankton community transitioned to dominance by cyanobacteria (blue-green algae, or Cyanophyta) in June, with a decrease in biomass from spring levels, seemingly consistent with reduced surface P concentrations.

A variety of coccoid and generally non-toxic forms of cyanobacteria were present in July, August and September, but there were also two filamentous genera known for dominating when available N is low and potentially forming toxic blooms, *Aphanizomenon* and *Dolichospermum*. No toxicity testing was performed, but the biomass of cyanobacteria was low to moderate in all but one sample in 2017, so the threat was very limited. The one sample of concern was collected from the surface early in the morning on July 26th and contained an elevated biomass of *Planktothrix*, a filamentous cyanobacterium usually associated with toxicity and known for forming dense growths near the thermocline. *Planktothrix* is buoyant and can come to the surface after mixing events or with extreme calm, the latter being the case on July 26th. All phytoplankton samples were collected in the upper meter of the lake, but the sample dominated by *Planktothrix* was literally a surface scum, purplish in color (typical of *Planktothrix*), with clumps of this alga floating on the surface. The biomass in that scum was high, but the wind picked up at 10 AM and the scum dispersed.

The water was obviously discolored in April and May, with a brownish hue, consistent with elevated golden algae and diatom biomass. The water was clear to greenish the rest of the sampling period, with cyanobacteria particles visible in the upper water column on most days but noted at high density only on July 26th, when conditions were unusually calm on Lake Garfield and a surface scum formed. These conditions are consistent with past observations and anecdotal information going back several years, usually in association with rooted plant surveys.

The zooplankton community of Lake Garfield (Table 18, Figures 26 and 27) included a typical variety of forms, including moderately abundant copepods and cladocerans in all samples. Protozoans and rotifers were present in many samples but not common. The August sample

Table 17. Phytoplankton composition and relative abundance as biomass

	PHYTOPLANKTON BIOMASS (UG/L)							
	Garfield	Garfield	Garfield	Garfield	Garfield	Garfield	Garfield	Garfield
	DH	DH	DH	DH	DH	DH Surf	DH	DH
TAXON	**04/11/17**	**05/18/17**	**06/29/17**	**07/10/17**	**07/26/17**	**07/26/17**	**08/11/17**	**09/15/17**
BACILLARIOPHYTA								
Centric Diatoms								
Aulacoseira	7.5	39.4	18.0	9.1	4.2	20.8	0.0	9.8
Cyclotella	35.0	170.8	2.0	1.5	1.4	3.5	0.0	0.0
Araphid Pennate Diatoms								
Asterionella	225.0	23.4	16.0	6.0	0.0	0.0	0.0	0.0
Fragilaria/related taxa	15.0	219.0	36.0	45.3	33.6	415.2	0.0	29.3
Tabellaria	0.0	29.2	32.0	241.6	616.0	498.2	26.6	260.8
Monoraphid Pennate Diatoms								
Achnanthidium/related taxa	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0
Biraphid Pennate Diatoms								
Cymbella/related taxa	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pinnularia	125.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CHLOROPHYTA								
Flagellated Chlorophytes								
Eudorina	0.0	17.5	48.0	0.0	0.0	0.0	26.6	0.0
Coccoid/Colonial Chlorophytes								
Elakatothrix	0.0	1.5	2.0	3.0	0.0	0.0	0.0	0.0
Oocystis	0.0	0.0	0.0	0.0	0.0	55.4	0.0	0.0
Schroederia	0.0	0.0	100.0	0.0	0.0	0.0	41.5	0.0
Sphaerocystis	0.0	0.0	0.0	0.0	0.0	0.0	19.9	0.0
Filamentous Chlorophytes								
Desmids								
Cosmarium	0.0	5.8	0.0	0.0	0.0	0.0	0.0	0.0
CHRYSOPHYTA								
Flagellated Classic Chrysophytes								
Dinobryon	6037.5	4664.7	780.0	317.1	42.0	0.0	99.6	0.0
Mallomonas	6.3	0.0	0.0	0.0	7.0	0.0	0.0	8.2
CRYPTOPHYTA								
Cryptomonas	0.0	0.0	16.0	0.0	0.0	0.0	0.0	0.0
CYANOPHYTA								
Unicellular and Colonial Forms								
Aphanocapsa	0.0	0.0	0.0	0.0	8.4	0.0	6.6	0.0
Chroococcus	0.0	0.0	0.0	36.2	0.0	0.0	0.0	0.0
Gomphosphaeria	0.0	0.0	0.0	7.2	105.0	62.3	19.9	19.6
Snowella	0.0	0.0	0.0	27.2	16.8	20.8	3.3	9.8
Woronichinia	0.0	1.5	4.0	0.0	0.0	86.5	0.0	65.2
Filamentous Nitrogen Fixers								
Aphanizomenon	0.0	0.0	234.0	176.7	163.8	269.9	172.6	762.8
Dolichospermum	0.0	0.0	0.0	302.0	168.0	692.0	199.2	65.2
Filamentous Non-Nitrogen Fixers								
Limnoraphis	0.0	14.6	40.0	30.2	56.0	69.2	33.2	0.0
Planktothrix	0.0	0.0	0.0	0.0	0.0	3017.1	0.0	0.0
EUGLENOPHYTA								
Trachelomonas	50.0	0.0	40.0	0.0	0.0	34.6	16.6	16.3
PYRRHOPHYTA								
Ceratium	0.0	63.5	348.0	262.7	121.8	301.0	144.4	0.0
Peridinium	0.0	0.0	0.0	31.7	0.0	0.0	34.9	0.0
DENSITY (CELLS/ML) SUMMARY								
BACILLARIOPHYTA	**420.0**	**481.8**	**106.0**	**303.5**	**655.2**	**937.7**	**66.4**	**352.1**
Centric Diatoms	42.5	210.2	20.0	10.6	5.6	24.2	39.8	48.9
Araphid Pennate Diatoms	240.0	271.6	84.0	292.9	649.6	913.4	26.6	303.2
Monoraphid Pennate Diatoms	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0
Biraphid Pennate Diatoms	137.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CHLOROPHYTA	**0.0**	**24.8**	**150.0**	**3.0**	**0.0**	**55.4**	**88.0**	**0.0**
Flagellated Chlorophytes	0.0	17.5	48.0	0.0	0.0	0.0	26.6	0.0
Coccoid/Colonial Chlorophytes	0.0	1.5	102.0	3.0	0.0	55.4	61.4	0.0
Filamentous Chlorophytes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Desmids	0.0	5.8	0.0	0.0	0.0	0.0	0.0	0.0
CHRYSOPHYTA	**6043.8**	**4664.7**	**780.0**	**317.1**	**49.0**	**0.0**	**99.6**	**8.2**
Flagellated Classic Chrysophytes	6043.8	4664.7	780.0	317.1	49.0	0.0	99.6	8.2
Non-Motile Classic Chrysophytes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Haptophytes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tribophytes/Eustigmatophytes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Raphidophytes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CRYPTOPHYTA	**0.0**	**0.0**	**16.0**	**0.0**	**0.0**	**0.0**	**0.0**	**0.0**
CYANOPHYTA	**0.0**	**16.1**	**278.0**	**579.5**	**518.0**	**4217.7**	**434.9**	**922.6**
Unicellular and Colonial Forms	0.0	1.5	4.0	70.7	130.2	169.5	29.9	94.5
Filamentous Nitrogen Fixers	0.0	0.0	234.0	478.7	331.8	961.9	371.8	828.0
Filamentous Non-Nitrogen Fixers	0.0	14.6	40.0	30.2	56.0	3086.3	33.2	0.0
EUGLENOPHYTA	**50.0**	**0.0**	**40.0**	**0.0**	**0.0**	**34.6**	**16.6**	**16.3**
PYRRHOPHYTA	**0.0**	**63.5**	**348.0**	**294.5**	**121.8**	**301.0**	**179.3**	**0.0**
TOTAL	**6513.8**	**5250.9**	**1718.0**	**1497.6**	**1344.0**	**5546.4**	**884.8**	**1299.1**
BIOMASS DIVERSITY	**0.16**	**0.23**	**0.75**	**0.88**	**0.77**	**0.70**	**0.97**	**0.61**
BIOMASS EVENNESS	**0.17**	**0.22**	**0.62**	**0.75**	**0.69**	**0.61**	**0.83**	**0.56**

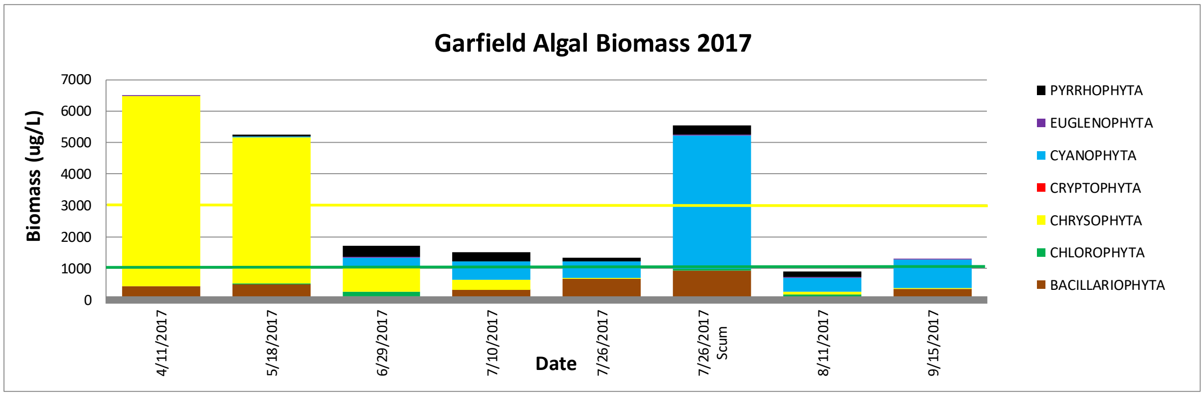


Figure 25. Phytoplankton biomass in Lake Garfield in 2017

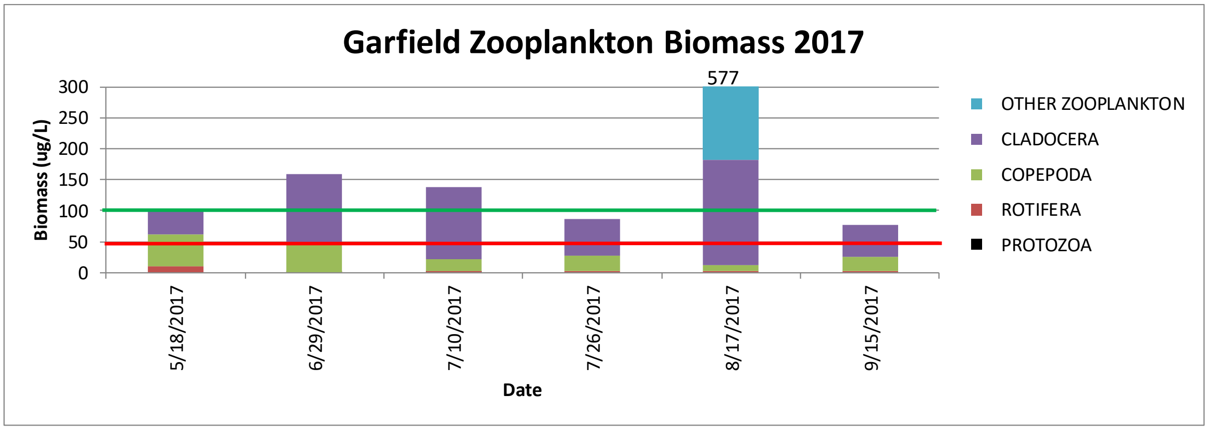


Figure 26. Zooplankton biomass in Lake Garfield in 2017

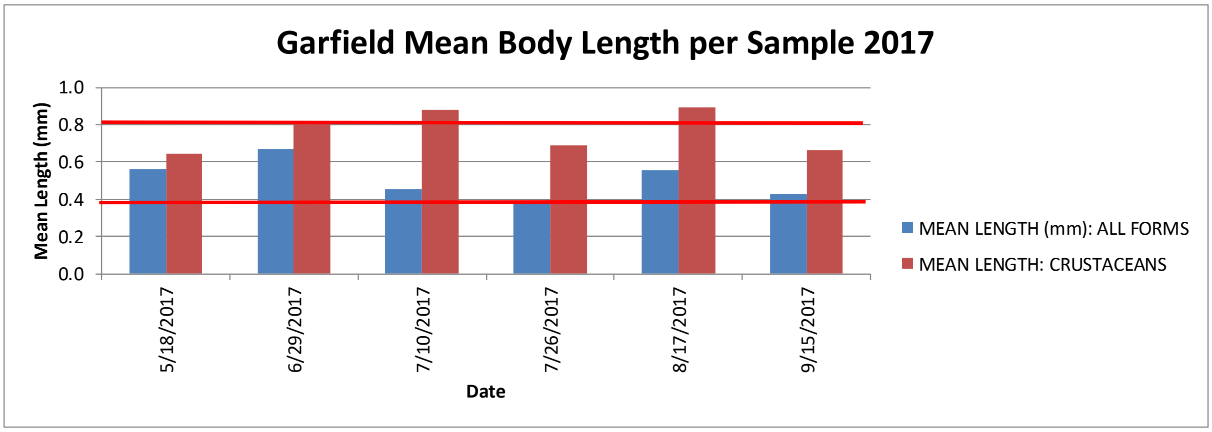


Figure 27. Zooplankton mean length in Lake Garfield in 2017

Table 18. Zooplankton composition and relative abundance as biomass

	ZOOPLANKTON BIOMASS (UG/L)					
	Garfield	Garfield	Garfield	Garfield	Garfield	Garfield
TAXON	5/18/2017	6/29/2017	7/10/2017	7/26/2017	8/17/2017	9/15/2017
PROTOZOA						
Ciliophora	0.0	0.2	0.3	0.0	0.0	0.0
ROTIFERA						
Asplanchna	10.3	0.0	0.0	0.0	0.0	0.0
Conochilus	0.0	0.0	1.7	2.1	2.4	1.9
Kellicottia	0.1	0.0	0.0	0.0	0.0	0.0
Keratella	0.0	0.0	0.0	0.1	0.0	0.0
Polyarthra	0.0	0.1	0.0	0.0	0.0	0.0
Trichocerca	0.0	0.0	0.0	0.0	0.0	0.9
COPEPODA						
Copepoda-Cyclopoida						
Cyclops	21.2	5.8	0.0	3.9	3.9	0.0
Mesocyclops	11.9	17.1	1.0	7.9	2.0	10.9
Copepoda-Calanoida						
Diaptomus	5.3	14.7	10.3	1.9	2.3	3.4
Other Copepoda-Nauplii	12.6	10.5	8.4	10.5	2.1	8.4
CLADOCERA						
Daphnia ambigua	39.2	109.8	117.0	60.7	169.2	52.2
OTHER ZOOPLANKTON						
Chaoboridae	0.0	0.0	0.0	0.0	395.0	0.0
SUMMARY STATISTICS						
BIOMASS						
PROTOZOA	0.0	0.2	0.3	0.0	0.0	0.0
ROTIFERA	10.3	0.1	1.7	2.3	2.4	2.9
COPEPODA	50.9	48.0	19.6	24.1	10.2	22.6
CLADOCERA	39.2	109.8	117.0	60.7	169.2	52.2
OTHER ZOOPLANKTON	0.0	0.0	0.0	0.0	395.0	0.0
TOTAL ZOOPLANKTON	100.4	158.0	138.6	87.1	576.8	77.7
MEAN LENGTH (mm): ALL FORMS	0.56	0.67	0.45	0.39	0.55	0.43
MEAN LENGTH: CRUSTACEANS	0.64	0.82	0.88	0.69	0.89	0.66

contained a large biomass of Chaoborus, the phantom midge, which is a large zooplankter that eats other zooplankton and usually lives near the boundary of the oxygenated zone in deeper water where darkness and limited oxygen lower predation by fish. Whether the sample net was deeper than usual or this population had migrated upward is not known, but this is not an unusual finding. It did raise zooplankton biomass markedly on that date, however. Overall zooplankton biomass was at least moderate (>50 µg/L) in all but the April sample (which is early for a zooplankton community to develop from overwintering resting stages) and was desirably elevated (>100 µg/L) in half the samples.

Cladocerans represent the zooplankton group that exercises the most control over algae, being relatively non-selective filter feeders. Among them, *Daphnia* is most desirable, as it is among the larger cladocerans and filtering capacity is related to body length. The biomass of *Daphnia* varied but was at least moderate all summer, indicating substantial filtering capacity that should improve water clarity. *Daphnia* are a favored food resource for small fish, and the continued presence of *Daphnia* through the summer suggests limited predation by small fish, probably as a consequence of predation on those small fish by larger fish. No fishery survey was conducted at Lake Garfield as part of this program, but a UMASS project resulted in surveys in 2015 and 2016. Large fish were abundant, especially bass, and this is reflected in the size distribution of the zooplankton (Figure 27), with mean length in the ideal zone or even above it (possibly suggesting too many large fish relative to the forage base) in 2017.

Phosphorus Loading Model

The loading model is a spreadsheet where key values are added and many calculations are made. It is not a black box approach, and it is not based on the extensive quantities of data necessary to more sophisticated, mechanistic models. Yet the Lake Loading Response Model (LLRM) has proven very useful in numerous lake studies and TMDL development efforts. It addresses inputs from precipitation, surface water, ground water (with septic system influence), internal loading, any discharges, and direct wildlife inputs. It has multiple points where reality checks using actual data can be applied. It predicts the resulting in-lake concentrations of N and P and provides expected values for chlorophyll-a, water clarity, and bloom frequency. It is a steady state model, predicting longer term average conditions, not daily to seasonal values. For systems where eutrophication is a primary concern, it is a very useful model.

Precipitation was not directly assessed in this study, but average P concentrations in precipitation range from about 0.01 to 0.04 mg/L in southern New England, based on many lake studies, with an average near 0.017 mg/L. Average precipitation for the Lake Garfield area is 1.16 m/yr (46 inches). Multiplication for the area of the lake suggests an annual input of 19.2 kg. Use of an export coefficient of 0.2 kg/ha/yr for direct precipitation inputs to the area of Lake Garfield yields an input of 19.5 kg/yr, a close match.

Internal loading has been estimated by likely release from sediment and changes in water column concentrations over the summer, and is calculated at up to 200 kg/yr. This is only a fifth of the load projected in an earlier study (AES 1990), and represents the upper limit on annual internal loading in the 2017 study. With an average hypolimnetic concentration increase over the summer of 0.22 mg/L, the internal load would be estimated at 106 kg/yr, which equates to a release rate of 4 $mg/m^2/d$ over a 90 day period of anoxia in water >7 m deep. Such a release rate is within the normally encountered range for anoxic situations and is quite believable. However, much of that released P remains in the hypolimnion and does not mix into the upper waters. Fall breakdown of stratification will bring oxygen to the deep waters, and most of that P will precipitate and settle out quickly. However, some algae, including the problem cyanobacteria observed in Lake Garfield, can grow near the top of the hypolimnion and at the sediment-water interface around its edge, so some of this P will move upward with rising algae. It is estimated that about 25% of the total internal load is mixed within the lake, becoming part of the effective load that figures into prediction of epilimnetic P concentration. This would be about 26.5 kg/yr. Use of an export coefficient of 0.9 kg/ha/yr over the 29 ha affected by anoxia yields an internal load of 26.1 kg, a reasonable match.

Direct wildlife inputs were not assessed in this study, but Lake Garfield does not have large resident populations of waterfowl, beaver, or other water dependent wildlife. Steep, rocky shores around most of the lake limit such populations. Assuming the equivalent of 30 individuals of some form of wildlife (mostly waterfowl) at a common literature input rate of 0.2 kg P per individual per year, an input of 6 kg/yr is projected.

On-site waste water inputs (e.g., septic systems) can be estimated in the model by adding the number of housing units in zones based on distance from the lake, factoring in occupation rates and portion of the year the residence is in use, multiplying by the amount of P generated per person per year, and adjusting for attenuation in the soil on the way to the lake. There are 74 residences within 100 feet of the lake and another 68 within 300 feet, most occupied for less than half the year, with an average occupancy of 2.5 people/residence. The calculation in the model results in a predicted P input from waste water of 9.7 kg/yr. The seepage measurement exercise conducted as part of this study resulted in an estimate of ground water P load of 13.2 kg/yr, which includes waste water and background P in the ground water. The match is close and the results are believable.

The greatest effort in this study and in LLRM is devoted to assessing the influence of the watershed on surface water quality and related P loading. Land use in the watershed was divided into 3 categories: forested/wetland, residential, and agricultural (Table 19). There is just a small amount of agricultural land in the watershed (<1% of total area), all one farmstead off Rt 23 that is split between drainage areas B and C+L (Figure 24), and that farm has a substantial buffer between it and any tributary or the lake itself. Residential uses are found in all drainage areas of Lake Garfield and total to 28% of land use in the watershed. Residential areas were considered to be of low density in all drainages except H, which had moderate density housing in just part of the drainage area. However, few homes have large lawn areas or extensive drainage systems, so P inputs are expected to be on the low end of the possible range. The remaining 71% of the watershed is forested or wetland.

Table 19. Land use in Lake Garfield drainage basins

Drainage Designation	Agricultural (ha)	Residential (ha)	Forest (ha)	Total (ha)
A+J	0.0	5.6	18.1	23.7
B	3.1	1.0	3.0	7.1
C+L	4.2	20.3	49.7	74.2
D	0.0	1.9	24.2	26.1
E + M2	0.0	42.9	65.7	108.6
F+G	0.0	16.1	108.3	124.4
H	0.0	28.4	178.2	206.6
I + M1	0.0	101.5	92.8	194.3
K	0.0	4.5	21.8	26.3
Total	**7.3**	**222.2**	**561.8**	**791.3**

Multiplying land use areas within defined drainage areas by export coefficients selected to best represent expected P output provides estimates of P load from surface water under wet and dry conditions. Water yield is also calculated based apportionment of precipitation into runoff and baseflow. Reduction in loading from attenuation on the way to the lake can be factored in, and this attenuation will vary greatly among drainage areas based on land use and topography. Resulting outputs from each drainage area include water volume, P concentration and P load values that can be compared to alternative calculations (see the Surface Inflow Phosphorus section).

Comparison of the drainage basin by drainage basin loads from direct calculation and the model (Table 20) suggests that the model load predictions are between the TP and DP loads from direct calculation. As many of the water samples were collected some distance before the stream entered the lake, loss of particulate P would be expected and the TP load from direct calculation will be an overestimate. Additionally, as discussed previously, much of the particulate load (70% of the total on average) will settle out quickly even if it reaches the lake. Therefore, the model loads, based on empirically derived export coefficients, should be in between the directly calculated TP and DP loads, and they are in all but one case (for drainage area B, for which we have only one sample due to lack of access).

Table 20. Comparison of directly calculated and modeled phosphorus loads

	Direct Calculation		LLRM
	TP	DP	P
Drainage	(kg/yr)	(kg/yr)	(kg/yr)
A+J	5.2	3.8	2.8
B	0.5	0.2	1.8
C+L	17.3	8.5	10.3
D	3.8	0.9	2.1
E + M2	20.2	4.7	6.4
F+G	37.0	10.7	11.8
H	37.0	7.7	22.0
I + M1	20.0	8.1	12.4
K	12.3	1.7	2.3
Total	**153.2**	**46.3**	**71.8**

The P load to Lake Garfield as derived through LLRM is 133.2 kg/yr (Table 21). The corresponding water load is slightly more than 5.4 million m^3/yr, suggesting an average input concentration of 0.025 mg/L. This is a generally acceptable P concentration for inputs, but is slightly high for preferred in-lake P concentration. As settling particulate P is largely factored out of this input concentration already, the final concentration in the lake can be expected to be a function of internal processes, including flushing, water depth, algal uptake, and further settling, all of which are embodied in a series of empirical equations used to convert the P load into an average, in-lake, epilimnetic P concentration. Those equations predict an average P concentration of 0.012 mg/L. If we set epilimnetic P concentrations from the 2017 monitoring program that were less than the detection limit to a value of 0.01 mg/L, the average concentration from those measurements is 0.012 mg/L, matching the prediction. The model appears to appropriately represent the lake and its watershed.

Table 21. Water and phosphorus loading to Lake Garfield from LLRM

Source	Water	Phosphorus
	(m3/yr)	(kg/yr)
Direct Load to Lake		
Atmospheric	1132160	19.5
Internal	0	26.1
Wildlife	0	6.0
Waste water	15975	9.7
Watershed Load to Lake	4261324	71.8
Total Load to Lake	**5409459**	**133.2**

The breakdown of loads in Table 21 suggests that the watershed provides the greatest amount of P at 54% of the total, and dominates the water input at 78.8% of the total. Internal loading is the second largest itemized contributor of P at almost 20%, but supplies no water. Direct precipitation is the third largest source of P at just under 15%, and supplies 21% of the incoming water. Waste water and wildlife contribute 7.3 and 4.5% of the P load, respectively. Note that ground water inputs are not calculated separately in the model, which estimates waste water from on-site disposal systems and lumps other ground water inputs into the watershed load. Waste water is estimated to contribute <1% of the water load, while the total ground water load from seepage measurements represents about 12% of the total water load. The ground water contribution to P load beyond that provided by waste water is estimated at about 3 kg/yr, a very small value. So surface water inputs are the dominant component of both the water and P loads, while internal loading is a major contributor to the P load.

Using LLRM to determine what the most appropriate P loading target is complicated by differing perceptions among user groups. In general, there is serious concern among many over the abundance of Eurasian watermilfoil, but considerably less concern has been expressed over lake water quality. Leaders in the community are worried about the possible impact of waste water disposal, shown now in 2 studies to be a minor contributor of P, and storm water runoff from developed areas, found here to be the dominant P source, but not at an extreme loading level. Water clarity is generally regarded as acceptable, and averaged close to 4 m in this study. However, knowledgeable parties in town do not want cyanobacteria blooms in Lake Garfield, and one was observed in 2017 that had a high enough cell count to warrant posting of the lake with non-contact warnings. Yet while cyanobacteria were the most abundant algae in the lake during summer, cell counts and associated biomass were generally low to moderate.

If the general desires of lake users can be translated into numeric values, it would be fair to say that water clarity of 4 m and a minimal probability of cyanobacteria blooms would represent those desires. That equates to a TP concentration of 0.010 mg/L, very close to the current value for Lake Garfield. The lake is listed for excessive P concentration and low oxygen, but the basis for the excessive P designation is not clear and the low DO is only below a depth of 7 m, not at all unusual among Massachusetts lakes. It would indeed by appropriate to raise the deep water DO level, if only for the sake of aquatic life, but that would also be expected to reduce the internal load of P and may reduce the growth of algae in general and cyanobacteria in particular.

LLRM predicts a set of conditions under the current P inputs that can be compared to predicted conditions under alternative conditions (Table 22). The existing conditions are similar to but not exactly the same as measured values, mostly because there are only a few dates of actual measurement of chlorophyll-a and Secchi disk transparency. This does highlight the existence of some uncertainty, however, and model results should be interpreted in general terms.

Table 22. Model predictions from LLRM under defined conditions

SUMMARY TABLE FOR SCENARIO TESTING	Existing Conditions		Background Conditions (no human influence)	90% Reduction of Internal Load	25% Reduction of Watershed Load	Internal and Watershed Load Reduction
	Calibrated Model Value	Actual Data	Model Value	Model Value	Model Value	Model Value
Phosphorus (mg/L)	0.012	0.012	0.007	0.010	0.011	0.008
Mean Chlorophyll (ug/L)	4.0	6.7	2.0	3.0	3.0	2.0
Peak Chlorophyll (ug/L)	14.3	21.6	7.5	11.0	11.8	8.5
Mean Secchi (m)	3.3	4.2	5.0	3.9	3.8	4.5
Peak Secchi (m)	4.8	5.5	5.6	5.1	5.0	5.4
Probability of Chl >10 ug/L	1.8%	<3%	0.0%	0.4%	0.6%	0.1%

If all human influence is removed, which would mean no residential development or roads, no waste water disposal, and a return to very limited internal loading, the predicted background P concentration for Lake Garfield would be 0.007 mg/L. Average chlorophyll-a would decline to 2 µg/L from the current predicted average value of 4 µg/L and clarity would rise from the currently predicted 3.3 m to 5.0 m. The probability of a bloom would be negligible, compared to the current prediction of 1.8%. Current conditions are generally acceptable, but the expected background condition is indeed better. A return to background conditions is not a realistic expectation, but the comparison between current and background conditions does bracket the range of possible improvement. Reaching an average P concentration of 0.010 mg/L represents an approximate mid-point between current and background conditions.

Using LLRM to evaluate possible changes through management actions, a 90% reduction in internal loading would result in a predicted epilimnetic TP concentration of 0.010 mg/L, a Secchi disk transparency value of 3.9 m, and the probability of chlorophyll-a >10 ug/L (defined here subjectively as a bloom) of 0.4%. This would seem to meet the perceived goals of lake users. Reduced internal loading can be expected to reduce oxygen demand and increase deep water oxygen, but it is likely that some degree of anoxia would persist, and cyanobacteria might still grow in deeper water and rise to form an occasional bloom.

Reducing the P load from surface water will be a matter of managing runoff; dry weather inputs are minimal, but snowmelt and storm water loads are substantial. Getting major reductions in P loading from a watershed can be challenging, especially with limited human impacts at this time; an extensive USEPA database (accessed on 2/22/18 at https://www.epa.gov/water-research/geoplatform-stormwater-bmp-performance-database-0) suggests that >50% reduction is very hard to achieve, even where human impacts are clearly identifiable. Yet the Lake Garfield watershed has almost no runoff controls, so it seems reasonable to expect that a 25% reduction could be achieved. Such a reduction at Lake Garfield is predicted to reduce the average in-lake P concentration to 0.011 mg/L, resulting in a predicted average clarity value of 3.8 m and bloom frequency of 0.6%. While not quite as much a reduction as that achievable by controlling the internal load, the results are close to what is perceived as necessary to meet goals.

Decrease in internal loading and reduction of storm water loading are independent and could both be undertaken. The combined effort is predicted to result in a P concentration of 0.008 mg/L, clarity of 4.5 m, and a bloom probability of 0.1%. This would provide some margin of safety for achieving water quality goals, although it may not eliminate anoxia or completely prevent blooms, depending on the methods chosen.

Diagnostic Conclusions

With regard to phosphorus loading and resultant water quality, Lake Garfield is not in an undesirable condition overall, but has a few shortcomings. The average P concentration is close to 0.012 mg/L, average Secchi disk transparency is predicted at 3.3 m but measured at slightly more than 4 m, and the probability of chlorophyll-a >10 µg/L is about 2%. However, cyanobacteria are the most abundant phytoplankton during summer and surface scums can form during calm conditions. The most abundant cyanobacteria are possible toxin producers. Oxygen is low in only about 11% of the lake volume during stratification, in water >7 m deep, but this anoxia extends over about 30% of the lake bottom and encourages internal P loading.

The effective load of P to Lake Garfield is about 133 kg/yr on average, with about 54% from surface water, over 50% of which is from runoff during wet weather. Snowmelt represents about 37% of the surface water P load, and is largely retained by the lake as it refills in late winter and early spring. The internal load represents about 20% of the total P load, and is focused during summer when such loading has the most impact on phytoplankton growth and lowers the N:P ratio, favoring cyanobacteria. Remaining P sources are small and more difficult to control. Ground water with P from on-site waste water disposal was not found to be a substantial contributor of P to Lake Garfield. Direct precipitation is not believed to be a large source, but was not explicitly assessed. Wildlife inputs also appear to be minor.

Lake Garfield is on the impaired waters list established by the MA DEP for excessive P and low oxygen. The average surface P concentration does not appear to be excessive from this investigation, but the accumulation of P in the hypolimnion during summer is high and likely promotes cyanobacteria growth and dominance. The P concentrations on August 25, 2003 when MassDEP last sampled Lake Garfield were 0.011 mg/L at the surface and 0.66 mg/L near the bottom at 9 m. Other monitoring of the lake between 1985 and 2000 reported surface P concentrations ranging from 0.010 to 0.040 mg/L, but quality control and detection limit issues restrict reliance on those data. No deep water P concentrations are known to us prior to the MassDEP 2003 survey or since, until this 604b project. High hypolimnetic concentrations (>0.40 mg/L) are fostered by release of P from sediment exposed to anoxia, so the low oxygen is a major factor in summer P loading.

The low oxygen in deeper water is apparent from both this investigation and that of MassDEP in 2003. Oxygen is near depletion below a depth of 7 m. Older monitoring efforts, mainly by volunteers, used a DO meter with a cable length of only 6 m and reported no oxygen impairment, but it is likely that about 30% of the lake bottom has experienced low oxygen during stratification for many years.

The lake is not listed for impairment by algae in general or cyanobacteria in particular, and blooms may not be frequent enough to warrant such listing, but cyanobacteria are a threat in Lake Garfield and chlorophyll-a concentrations are sometimes elevated. The chlorophyll-a concentrations for integrated samples collected by MassDEP on August 25, 3002 were 13.2 and 14.4 µg/L, which are elevated values and likely included cyanobacteria. Yet the integrated value for August 17, 2017 was <5 µg/L and the highest 2017 value of 15 µg/L was for April when golden algae dominated. There are not enough data to conclude that Lake Garfield is impaired by algae, but there are indications that it may be, and the dominance of cyanobacteria in summer is undesirable.

Reduction of P loading could improve water quality in Lake Garfield. Modeling suggests that the background P concentration in the absence of human influence would be about 0.007 mg/L, with water clarity of 5 m and a negligible probability of algae blooms. Reaching historic background conditions is not a realistic goal, but achieving an average P concentration of <0.010 mg/L is possible and is a worthwhile goal for management. Reduction of runoff and internal loads would be the logical targets of management efforts.

Reducing the load from runoff and snowmelt by 25% is possible, and would reduce the predicted average P concentration from 0.012 to 0.011 mg/L with a predicted increase in clarity from 3.3 m to 3.8 m and a predicted decrease in bloom probability from 1.8% to 0.6%. Reducing internal P loading by 90% is also possible, and would lower the P concentration to 0.010 mg/L, with an increase in clarity to 3.9 m and a decrease in bloom probability to 0.4%. Reducing both watershed inputs by 25% and internal load by 90% would result in a predicted P concentration of 0.008 mg/L with an increase in clarity to 4.5 m and a decrease in bloom probability to 0.1%. Addressing internal loading is likely to provide greater benefits than watershed management in terms of cyanobacteria control, but both watershed and internal load control are worthwhile for overall water quality enhancement and protection in Lake Garfield.

Management Options

Projects completed under section 604b are expected to lead to sound management of subject study lakes, although complete management planning is typically outside the project scope. Yet this project has provided clear direction for future management, and specific recommendations can be made. The two P sources that warrant management attention are the internal load and runoff from the watershed relating to both storms and snowmelt. Available options for addressing each are well known and applicability can be evaluated based on the data generated in this study.

Management of Internal Loading

Management of internal loading requires control of the interchange between sediment and overlying water, especially where oxygen is low. There are three main options for achieving such control: dredging, P inactivation, and oxygenation. Each can be effective, but cost and regulatory/community acceptability are also important in choosing an approach.

Dredging is true restoration, removing problem sediment and setting the lake back in geologic time. Lake Garfield is largely a natural lake, although the water level was raised almost 2 m by damming and the water level changed drastically over the course of each year in the 1800s and early 1900s as a function of water release for use in downstream mills. As such, it has accumulated sediment since the last period of glaciation, and that sediment is largely organic and nutrient-rich. As a result, it creates oxygen demand that causes low oxygen in deeper waters and can release substantial amounts of P that fuel algae growth. Dredging would be the best technical approach to improving water quality in Lake Garfield, but this approach suffers from great expense and regulatory constraints.

It is very unusual for a lake to be dredged when lost depth does not need to be recovered, and Lake Garfield is still deep over the area that would be targeted for sediment removal. The cost of dredging is a minimum of $50,000 per acre-foot of sediment removed, with values up to 3 times that cost possible if there are technical difficulties (e.g., uphill pumping of sediment slurry, disposal area limitations) or sediment contamination (especially by hydrocarbons and metals). The exact depth of sediment that would need to be removed is unknown, and a proper feasibility study would cost on the order of $80,000, but removal of just one foot of sediment over the 71.5 acres of area influenced by anoxia would cost a minimum of $3.6 million after completion of engineering and permitting. This is an unlikely course of action at Lake Garfield.

P inactivation involves adding chemicals that bind the currently available P and prevent its release from sediment, even with future exposure to anoxia. Much of the target P is bound to iron, and under anoxia the iron and P can dissociate and dissolve in the overlying water. Additional P is bound in easily decayed organic matter (biogenic P) that can be harder to inactivate. Inactivation can be accomplished with the addition of calcium, aluminum, or lanthanum in a lake such as Lake Garfield. Calcium treatments have not been overly successful, as calcium tends to stay in the sediment only with very high pH; Lake Garfield has a naturally elevated pH (up to 8.2), but the pH target for calcium inactivation is much higher (about 10). Lanthanum is a newer inactivator, applied with a clay solution that is not yet approved for use in Massachusetts. This leaves aluminum compounds as the logical P inactivators, and aluminum has been used very successfully in Massachusetts lakes.

The dose of aluminum necessary to inactivate the Fe-P measured in the Lake Garfield sediment, estimated stoichiometrically from the mass of P in the upper 10 cm of sediment, is 34 g/m^2.

Inactivation of biogenic P is more experimental and potentially less reliable, but attempting to inactivate the Fe-P and biogenic P in the target area of Lake Garfield would require an aluminum dose of 87 g/m^2. The cost of Fe-P inactivation would be about \$145,000, while the cost of inactivation both Fe-P and biogenic P would be about \$373,000. The duration of benefit from such a treatment is estimated at 20 years.

Oxygenation involves adding enough oxygen to counter the existing demand, thereby avoiding anoxia and keeping P sequestered in the sediment. This approach could also oxygenate the bottom waters to a degree that would better support aquatic life such as fish and invertebrates during summer when there is currently inadequate oxygen in that area. Oxygenation can be accomplished by destratifying the lake, using air bubbled from the bottom or by pumping water upward or downward.

Upward pumping carries the risk of bringing poor quality water to the surface if the system is undersized or shuts down and is restarted later; both have been problems with this approach. Downward pumping is usually not attempted where the water is <9 m deep, as sediment can be resuspended by the water flowing downward if it cannot be released below the thermocline but far enough above the sediment-water interface. That could be an issue at Lake Garfield. Use of compressed air released near the bottom is the oldest method and the design parameters are well understood, but with a thin hypolimnion, the lateral distribution of air will require an extensive network of tubes and diffusers. Each circulatory oxygenation method has drawbacks with regard to Lake Garfield, and it is not clear that destratifying a naturally stratified lake will be well received in the regulatory system.

The alternative approach is to add oxygen to the deeper waters without destratifying the lake. Older methods include pulling the water into a chamber in the lake and bubbling air or pure oxygen through it to raise the oxygen content. Use of air can be effective but is inefficient, and power costs have limited recent application of that approach. Use of pure oxygen is more efficient, but submerged chambers have proven to be maintenance problems. A newer approach involves releasing fine bubbles of pure oxygen near the bottom with the intent of having them completely dissolved before they reach the thermocline and cause destratification. This has worked well where the hypolimnion is at least 5 m thick, but that is not the case in Lake Garfield. This leaves the newest approach, sidestream supersaturation, as the most viable technique. Water is pulled out of the target zone of the lake, oxygenated to well above normal saturation levels in a pressurized chamber on shore, and then put back into the target zone. This approach is gaining popularity in lakes like Lake Garfield.

Sidestream supersaturation would need to supply oxygen at a minimum rate of 0.8 $g/m^2/d$, but adding oxygen raises the rate of oxygen consumption, usually by about 50% when pure oxygen is used, so a supply rate of 1.2 $g/m^2/d$ is advisable. Over the 290,000 m^2 target area, this equates to an input of 348 kg of oxygen each day of operation. The system may not need to operate all summer, but planning for 90 days of operation is appropriate. The capital cost of such a system based empirical data is either \$4100/acre or \$1100/kg of oxygen delivered, suggesting a cost range of \$322,000 to \$383,000. Operational cost is either \$200/acre or \$15/kg, suggesting a range of \$5200 to \$14,300 per year.

Oxygenation would both reduce internal loading and greatly enhance deep water habitat. It will improve other aspects of water quality as well, and would be the preferred approach if affordable. Note that the capital cost of P inactivation and comparable oxygenation projects tends to be similar, but the oxygenation system requires ongoing operational expense. The need for oxygen is likely to be reduced over time, but is not likely to ever be eliminated. For reasons of overall water quality improvement and having ratepayers to support ongoing operation, drinking water suppliers tend to adopt oxygenation approaches. Town and associations managing recreational lakes more often apply P inactivation to avoid ongoing costs, but the benefits of oxygenation can be greater if the cost can be afforded.

Management of Surface Water Loading

Management of watershed inputs would be expected to focus on capturing snowmelt and storm runoff wherever possible, allowing either natural processes of settling and infiltration or augmented treatment such as P inactivation to lower the available P content of the water before it enters the lake. There are no less than a dozen input systems that could be addressed, but the largest flows are associated with two drainage areas (H and I+M1), with the addition of two more (F+G and E+M2) accounting for a cumulative 80% of the total surface water input. The total estimated P load from those 4 basins represents 75% of the total surface water P load, but two additional basins (C+L and K) also provide significant loads (19% of the total together). Only three drainage areas seem to warrant little attention (A+J, B and D), mainly because they represent so little of the total P load (6% together). Even these might be considered where opportunities present themselves for inexpensive P capture.

The simplest and most effective approach to reducing P availability is probably to dose inputs with aluminum much like would be done to inactivate P in surficial sediment, but on a more regular basis in response to snow melt and spring storms. However, this is not as economical as a simple, one-time, sediment inactivation treatment, as there are many input points to cover, each requiring a dosing station. Additionally, there are questions about the long-term effectiveness of aluminum on the organic particles that comprise a lot of the surface water loading. This approach could be highly applicable to a few of the larger drainage areas (e.g., H and F+G), but it may be more appropriate for most basins to adopt particulate P trapping methods.

Particulate trapping could be greatly enhanced by increased detention on the various tributaries and drainage ditches, but the watershed is not overly developed and space is available. There are questions of ownership, regulatory restrictions, and cost that must be addressed and are beyond the scope of this investigation. Much of the land that might make useful detention basins is probably private property. There are opportunities for easy detention in major drainage basins H, F+G, and I+M1, but these involve temporary flooding of existing wetlands and might run into permitting resistance under the Wetlands Protection Act. The cost of detention will vary substantially depending on site-specific conditions, but the total cost can be estimated based on an USEPA database (USEPA 2015) for such systems. The typical range of costs for detention systems is \$371 to \$668 per kg P removed. The target reduction in P loading from surface water is 25%, translating into an annual reduction of 38.3 kg of P. The USEPA estimates are spread over a presumed 20 year lifespan, so the total cost would be estimated at \$284,000 to \$512,000.

A superior trapping system would involve infiltration of the first flush of storm water, but with poorly drained soils in much of the watershed, such systems would have to be heavily engineered. The USEPA database (USEPA 2015) indicates a cost of \$7121 to \$7443 per kg P removed, so the cost would be over ten times that of detention. There may be opportunities for more economical implementation of detention or infiltration, but these will have to be sought out on a case by case basis and vetted through the permitting system.

Property owners could certainly help by implementing low impact development (LID) techniques on their properties, and this would not have to involve any public cost, but getting widespread public support represents a challenge. Use of rain gardens, in which potential runoff is trapped on the properties where it is generated, can be very effective. It may not be easy to infiltrate that runoff with current watershed soils, so systems might be more expensive than usually suggested, but just holding the water for a day and allowing settling to remove particulate P would be beneficial. Use of rain barrels, in which roof runoff is collected for use in watering the landscape when needed, will reduce runoff and related P loading. Use of swales that direct runoff into vegetated areas where it can be spread out is preferable to storm water pipes that deliver the water directly to a tributary or the lake itself. Most LID approaches carry limited cost, but getting property owners to adopt these approaches requires education, commitment, and often incentives.

It is unlikely that a 25% reduction in P loading from surface water could be achieved through LID, as developed land represents only 28% of the watershed, but that land is likely to contribute disproportionately more P than forested land. If every developed lot was subject to LID management and P loads were 50% higher on those lots than on forested watershed land, the reduction would be on the order of 21%. If half the developed property was addressed, which is a more realistic upper limit, a 10-11 % reduction might be expected. The goal would not be achieved, but P loading could be reduced at minimal public cost, leaving less detention or infiltration to be implemented by public projects to meet the goal. Promotion of LID techniques on developed properties would be an appropriate campaign for the Friends of Lake Garfield to adopt, supported by the Town of Monterey.

Preliminary Recommendations

The Town of Monterey should seek funding to improve detention and possible infiltration of storm water runoff from as many surface water input points as possible and consider installation of an oxygenation system to counter low oxygen in water deeper than 7 m (23 feet) in Lake Garfield.

Storm water quality management systems may vary substantially by location, and some site specific survey work will be needed. For many potential target parcels, relatively simple low impact development techniques may be adequate. For a few of the larger tributaries, creation of a detention facility, with or without a filter berm, may be advisable. The greatest benefit would be achieved by addressing drainage areas H and F+G, but all drainage areas warrant consideration and some of the smaller ones, like A+J, C+L, and K have higher export coefficients (larger contribution per unit area). A relatively simple engineering exercise involving field assessment and conceptual planning could lead to valid cost estimates with limited effort, but it would not be surprising to spend >$250,000 on storm water management in this watershed.

For oxygenation, the use of sidestream supersaturation would appear most suited to the Lake Garfield situation. The target layer is fairly large in area but not in thickness. Maintaining a stratified bottom layer would be preferable ecologically, and sidestream supersaturation would involve withdrawing deep water, oxygenating it, and putting it back with limited vertical disturbance. Based on the oxygen demand of 0.8 $g/m^2/d$ over an area of 29 ha, a daily oxygen input of 232 kg would be recommended, but further assessment is warranted prior to design. Based on systems installed elsewhere, the capital cost would be about $300,000 and the annual operating cost would be about $25,000.

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