Bantam Lake

2024 Water Quality Monitoring Report

Bantam Lake Protective Association
Morris, CT

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I. Executive Summary

The Bantam Lake Protective Association (BLPA) annually supports lake and watershed water quality monitoring as part of its lake management strategy. Brawley Consulting Group, LLC (BCG) completed the lake monitoring again in the 2024 season. However, the lake was treated with a hydrogen peroxide-based algicide in April before the monitoring commenced, and in this report additional analyses were conducted to assess the efficacy of that treatment. In 2024 Bantam Lake exhibited eutrophic water quality, particularly after stratification when the water column was completely mixed in early August. Average summer chlorophyll-*a* concentrations, Secchi disk transparency, and season average total phosphorus were all within eutrophic ranges based on trophic criteria used in Connecticut. Water quality from April through mid-July was good with very low chlorophyll, low epilimnetic total phosphorus concentrations, and high Secchi transparency.

Cyanobacteria cell concentrations increased from very low early-season levels to very high levels at the end of the season. Based on State municipal guidance on cyanobacteria levels at public beaches, cyanobacteria cell concentrations ranged throughout the season from Visual Rank Category 1, which present little to no public threat, to Visual Rank Category 3 conditions which present the highest risk. The cyanobacteria toxin microcystin was measured on a biweekly basis at two of the water quality monitoring sites in July and August. Microcystin concentrations were never measured at an unsafe level. All concentrations were <1 µg/L. The Federal and State limit for acceptable conditions for recreation waters is 8 µg/L.

Season average specific conductance, sodium, chloride, calcium, and alkalinity were all at 7-year lows. Sodium, chloride, calcium, and alkalinity were all highly correlated with specific conductance. Recent decreases in these variables may be related to milder winters resulting in less deicing salt use. Improved stormwater management could also result in this type of water quality change.

Changes in historical water quality were documented based on historical studies from the 1930s, 1970s, and 1990s, and based on a paleolimnological study that inferred changing conditions since the 1850s. Those studies indicated that Bantam Lake shifted from an oligotrophic to eutrophic lake and salt levels increased over time. Trends in recent years – 2018 to 2024 – were examined using *Multiple Linear Regression* and *Analysis of Variance*. For the first time since 2018, results from those analyses indicated that Bantam had experienced a water quality change in recent years which was largely due to <u>decreasing</u> specific conductance levels. Trophic variables such as Secchi disk transparency, chlorophyll-a concentrations and total phosphorus concentration have not exhibited statistically significant changes since 2018.

The 2023, 2024, and the combined 2023 and 2024 season averages for epilimnetic total phosphorus, total nitrogen, and chlorophyll-*a* concentrations, and Secchi disk transparency were determined and compared to nutrient goals and projections in the Bantam Appendix of the State's TMDL document. The combined 2023/2024 averages for total phosphorus, total nitrogen, and chlorophyll-*a* were higher than the TMDL goals and projections, while the Secchi average was below the TMDL projected level.

Cyanobacteria productivity and growth rates over the last five years were closely examined to assess impacts from the hydrogen peroxide-based algicide treatment performed in April of 2024. No statistically significant differences were found in seasonally averaged Secchi disk transparency, chlorophyll-*a* concentration, relative phycocyanin concentration, or cyanobacteria cell concentration from 2020 to 2024. On a monthly basis, conditions in April through July of 2024 appeared better than those of 2020 to 2023. However, August through October conditions in 2024 were some of the poorest in the last five years.

Center Lake site cyanobacteria growth rates since 2020 were examined following modified CyanoMonitoring protocols. The growth rates in 2024, when the peroxide-based algicide was used, appeared less acute than when copper sulfate was used once or twice in the 2020 to 2022 seasons, or when no algicide was used in the 2023 season. Recommendations to advance the goals of the BLPA are provided at the end of the report.

II. Introduction

Bantam Lake is a 966-acre waterbody located in the towns of Litchfield and Morris, Connecticut; and is the largest natural lake in the State. Geologically, it is situated in the Western Uplands of Connecticut (Bell 1985, Canavan & Siver 1995). That geological region has erosion resistant, crystalline bedrock comprised of schists, gneiss, granite gneiss, and granofels (Healy & Kulp 1995). The watershed of Bantam Lake is 20,218 acres resulting in a watershed to lake ratio of approximately 21. In a 1995 survey, land use was characterized as mainly deciduous forest and agriculture lands with smaller areas of medium-density residential land use, wetlands, and coniferous forests (Healy & Kulp 1995). Much of the shoreline is now lined with homes, beaches and several seasonal camps. There is also a large tract of conservation land along the northern shoreline which is owned by the White Memorial Foundation (See Location Map, Fig 1.).

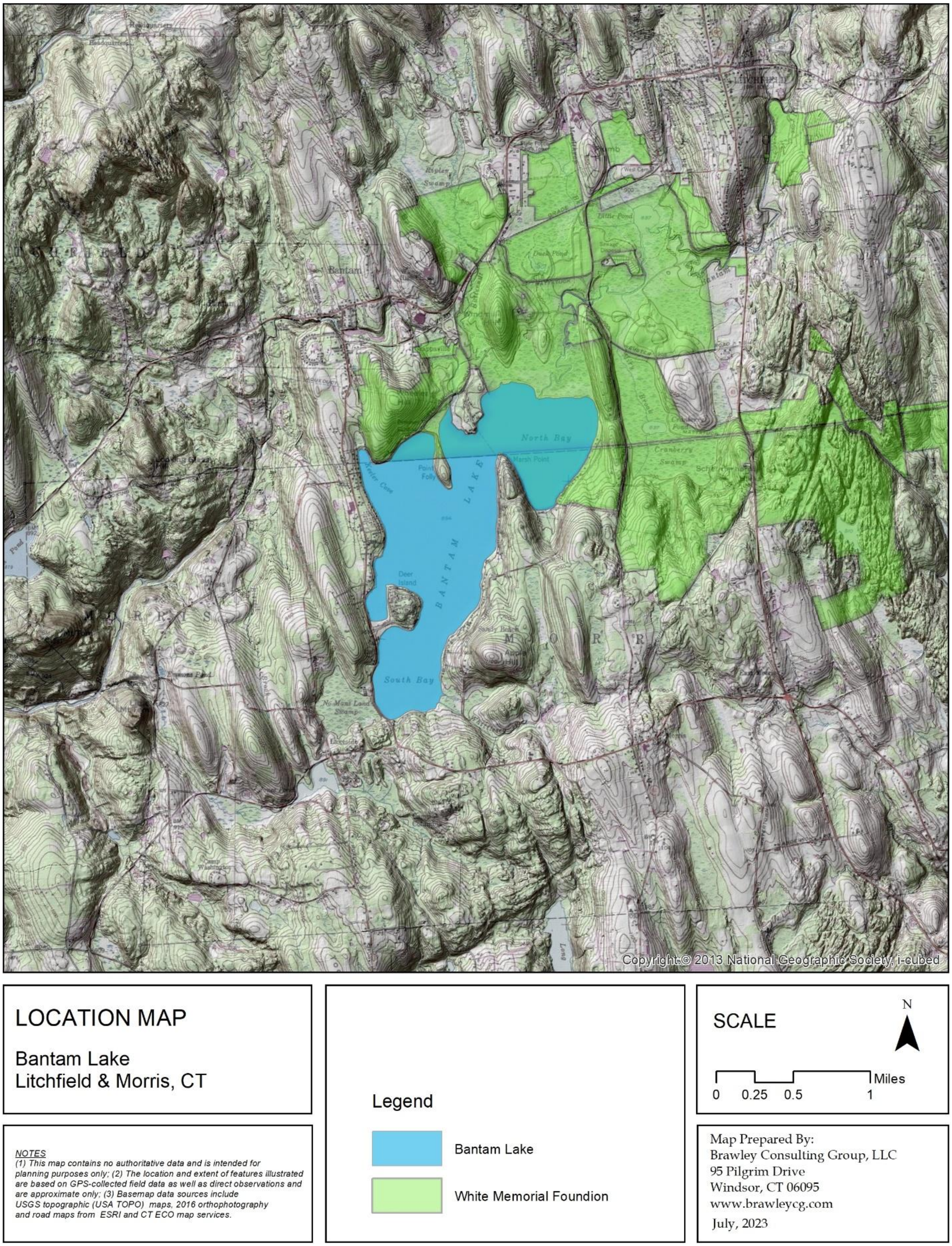
A. Historical Water Quality Studies

Earliest published assessments of Bantam Lake occurred in the late 1930s (Deevey 1940). That study and several others occurring over the next 70 years (Frink & Norvell 1984, Canavan & Siver 1994, 1995, Healy and Kulp 1995) included Bantam Lake as part of statewide surveys of Connecticut Lakes that used standard in-situ measurements and laboratory analyses of water samples to develop historical records of water quality. These studies resulted in important historical water quality baselines for many of Connecticut's lakes. Several of those statewide surveys have been compiled in Canavan and Siver (1994, 1995).

A paleolimnological study of Bantam Lake's water quality used statistically significant inference models and the remains of fossil bearing algae layered chronologically in a sediment core to estimate changes in water quality over time (e.g., trophic level, conductivity levels, and pH, over time; Siver 1993, Siver and Marsicano 1996). The oldest sediments in the Bantam sediment core dated back to *ca* 1857.

Based on the earliest fossil assemblages, Bantam Lake was oligotrophic (low primary/algal productivity) to early mesotrophic from *ca* 1857 through *ca* 1898 (Fig. 2). Subsequently, the lake's trophic status changed and by *ca* 1926, Bantam Lake was mesotrophic (moderate productivity). The lake became more eutrophic between *ca* 1946 and *ca* 1964. The fossil assemblages near the top of the core dated to *ca* 1991 and indicated that the lake was eutrophic (high primary/algal productivity).

Bantam Lake continues to exhibit eutrophic characteristics including high levels of algal productivity that have become one of the primary management concerns of the Bantam Lake Protective Association (BLPA). High concentrations of cyanobacteria and bloom-like conditions are common between the midsummer and fall periods of the recreation season. This has resulted in the inclusion by the State of Connecticut of Bantam Lake in the State's list of impaired waterbodies, which cites algae, chlorophyll, and nutrients as the causes of impairments (CT DEEP 2020).



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July, 2023

Figure 1. Location map for Bantam Lake.

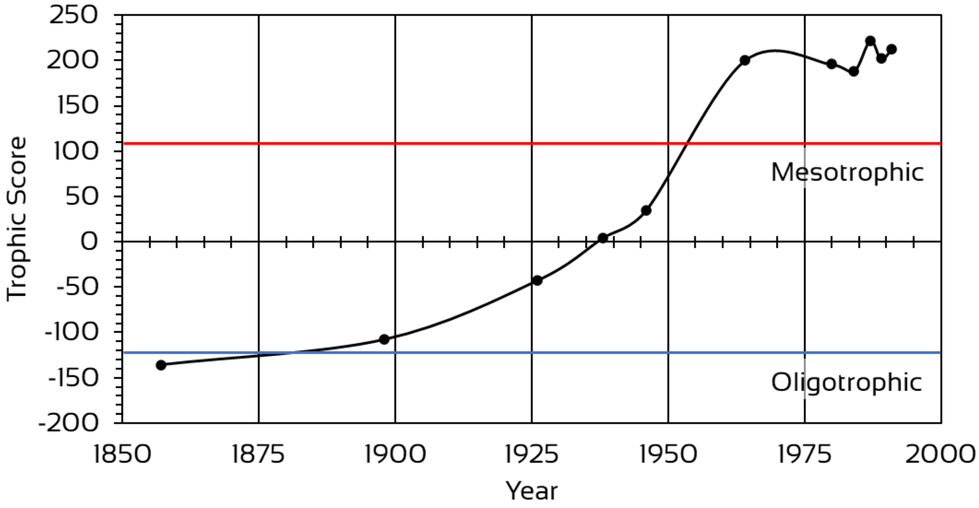

Figure 2. Reconstruction of Bantam Lake trophic status between *ca* 1857 and *ca* 1991. The blue horizontal line represents the division between oligotrophic and mesotrophic lakes; and the red horizontal line represents the division between mesotrophic and eutrophic lakes based on a trophic score (Siver and Marsicano 1996).

B. CT DEEP's TMDL Study and Watershed-Based Plan

Recent efforts at Bantam Lake by the Connecticut Department of Energy of Environmental Protection (CT DEEP), in conjunction with the US EPA and the BLPA included a *Total Maximum Daily Loading* (TMDL) *Study* (CT DEEP 2021) and a *Watershed-Based Plan* (CT DEEP 2021). The TMDL study and watershed plan provided important information on: (1) the lake and watershed; (2) sources of the nutrients that impair the lake; and (3) measures to reduce nutrient exports to the lake. Recommendations are also provided on future lake and watershed monitoring efforts.

Recent efforts by the BLPA highlighted target nutrient goals for Bantam Lake established by the CT DEEP in the TMDL study. These goals were established after considering multiple predictive models and a *weight of evidence evaluation* of those models. The target in lake total phosphorus goal was 20 µg/L and the target total nitrogen goal was 400 µg/L. Nutrient loading targets of 1,211.1 kg/year (3.32 kg/day) total phosphorus and 20,326 kg/year (55.7 kg/day) total nitrogen were established to achieve the target nutrient concentration goals. Based on those nutrient loading targets, predicted in lake chlorophyll-*a* concentration was 11.8 µg/L and Secchi disk transparency was 2.1 meters. Total phosphorus, total nitrogen, chlorophyll-a, and Secchi disk transparency are all water quality parameters described and reported on below for 2024. Additionally, historical data for these parameters have been examined below to provide context for target nutrient and chlorophyll concentrations and Secchi disk transparency.

C. Current Water Quality Concerns and Algicide Treatments

Cyanobacteria harmful algal blooms have been a key issue in lake management over the last 20 years because of the risk they present to ecosystem and public health. In addition to depletion of oxygen from the water column following a bloom, many genera of cyanobacteria are potentially toxic. The cyanotoxins are generally grouped into one of several categories: hepatotoxin that cause liver damage, neurotoxins that have been associated with neurological disorders like amyotrophic lateral sclerosis (ALS), and others in groups classified as dermatoxins, cytotoxins, and endotoxins. The State of Connecticut provides a useful summary on cyanobacteria, the toxins they produce, and standards by which municipal health department can assess conditions at public beaches (CT DPH & CT DEEP 2019).

In efforts to manage cyanobacteria concentrations and minimize public health risk at Bantam Lake, copper sulfate treatments were implemented in response to the bloom or imminent bloom conditions. Treatments occurred most seasons up through 2023 when no treatment occurred. In some years two copper sulfate treatments have taken place. The first was typically in July with the other occurring in August. See Table 1 for dates of treatments since 2017 along with the type of algicide used.

Table 1. Algicide treatments at Bantam Lake since 2018. The type of algicide used is identified as either copper sulfate (Copper) or peroxide-based (Peroxide).

Year	1st Treatment	Algicide	2nd Treatment	Algicide
2017	July 24	Copper	No 2nd treatment	
2018	July 26	Copper	No 2nd treatment	
2019	July 30	Copper	No 2nd treatment	
2020	August 6	Copper	August 24	Copper
2021	July 29	Copper	August 24	Copper
2022	July 17	Copper	August 30	Copper
2023	No 1st treatment		No 2nd treatment	
2024	April 10	Peroxide	No 2nd treatment	

In the spring of 2024, a new treatment approach was tested. This approach was not reactive like previous copper sulfate treatments, but rather preventative, and based on ongoing research by the Army Corps of Engineers and others using hydrogen peroxide-based algicides early in the season (e.g. Calomeni et. al. 2023, Kinley-Baird et. al. 2023). Each spring, resting cells of cyanobacteria overwintering in the lake sediments germinate to begin a new cycle. This new approach used a hydrogen peroxide-based algicide to target those germinating cells before cyanobacteria began accelerated cell division and reached problematic levels. Some of the additional benefits of peroxide include eliminating accumulation of copper in the sediments and selectivity, i.e., cyanobacteria were sensitive to the treatment whereas other algal taxa were not. Some researchers have found that other algal taxa are sensitive to the treatment, e.g. diatoms (Allaf et. al. 2023). One of the disadvantages of hydrogen peroxide-based algicides is the high cost.

In this report, we closely examined cyanobacteria indicator data (cell counts, relative phycocyanin, chlorophyll-*a*, and Secchi disk transparency) over the last five years to determine if the new treatment had resulted in any improvements in the conditions.

III. Methods

A. Water Quality Monitoring

Four sampling sites were visited seven times each between April and October of 2024 for water quality monitoring (See Fig. 3). Data and water sample collections occurred on: April 9th, May 18th, June 17th, July 15th, August 23rd, September 23rd, and October 21st. Sites were identified as North Bay (NB), Center Lake (CL), Point Folly (FP), and South Bay (SB). Maximum depths were approximately 6 meters (m), 8m, 6.5m, and 4.5m at the NB, CL, FP, and SB sites, respectively.

During site visits, Secchi transparency was measured with a 26 cm diameter Secchi disk. Vertical profile data for seven water column properties were collected using a Eureka Manta II Multiprobe. Profiled data were measured at 0.5m from the surface and at one-meter intervals down to 0.5m above the bottom and included the following variables: temperature (°C), dissolved oxygen (mg/L), percent oxygen saturation (%), specific conductance (µS/cm), pH (SU), oxidation-reduction potential (mV), and relative cyanobacteria biomass (µg/L).

Water samples were collected at only the NB, CL, and SB sites during visits and analyzed for the variables listed in Table 2. Samples collected in April – July were analyzed by York Analytical Laboratories in Newtown, CT. Those samples were stored in an ice-filled cooler until delivered on the same day to the laboratory. Samples collected in August – October were analyzed by the University of Connecticut Center for Environmental Science and Engineering (UCONN CESE) in Storrs, CT. Those samples were also kept on ice until frozen at BCG facilities before delivery later in week.

Water samples were collected using two different methods and at several depths in the water column. For nutrient and alkalinity analyses, samples were collected with a horizontal Van Dorn water sampler at 1 meter (m) below the surface (epilimnion), at approximately 0.5m above the sediment-water interface (hypolimnion), and at the thermocline, which was determined using vertical temperature profile data collected at each site on each sample date. For base cations of sodium (Na+), potassium (K+), calcium (Ca2+), magnesium (Mg2+), and the anion chloride (Cl-), samples were collected on three occasions during the season from 1m below the surface. For chlorophyll-*a*, a weighted tube sampler was used to collect and integrate water from the top three meters of the water column at the NB and CL sites.

B. Cyanobacteria Monitoring

Pelagic algae samples were collected on 15 different dates in 2024, including the dates water quality data and samples were collected (Table 2). Samples for algae analyses were collected at the NB, CL, and SB sites by integrating the top three meters of the water column with a 3-meter-long sampling tube. Samples were treated with Lugol's solution and kept on ice. Samples were treated with hydrostatic pressure to collapse gas vesicles that might be creating positive buoyancy in cyanobacteria cells (Lawton et al. 1999).

When necessary, measured volumes of the preserved whole water samples were concentrated into smaller measured volumes with centrifugation and a vacuum pump / filtration flask system. This step was omitted when cyanobacteria concentrations were high based on a visual assessment at the lake or based on low Secchi disk transparency. A known portion of those concentrates or whole water samples were pipetted into a counting chamber and genus-level algal cell enumerations were performed by counting algae cells in a subset of fields within the counting chamber slide using an inverted Nikon Diaphot research microscope. Those counts were then corrected to be reflective of the whole sample.

Table 2. Summary of data collections for Bantam Lake in 2024. NB = North Bay Site, CL = Center Lake Site, FP = Folly Point Site, and SB = South Bay Site. Profiles include parameters measured throughout the water column (e.g. temperature, dissolved oxygen, etc.). TP = total phosphorus; TN = total nitrogen; NH_3 = ammonia; base cations = sodium, potassium, calcium, and magnesium.

2024 Dates	Profiles and Secchi	Algae	TP, TN, NH_3 & Alkalinity	Base Cations and Chloride	Chlorophyll
9-Apr-24	NB, CL, FP, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB
22-Apr-24	NB, CL, FP, SB	NB, CL, SB			
30-Apr-24	NB, CL, SB	NB, CL, SB			
18-May-24	NB, CL, FP, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB
1-Jun-24	NB, CL, FP, SB	NB, CL, SB			
17-Jun-24	NB, CL, FP, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB
6-Jul-24	NB, CL, FP, SB	NB, CL, SB			
15-Jul-24	NB, CL, FP, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB
1-Aug-24	NB, CL, FP, SB	NB, CL, SB			
7-Aug-24	NB, CL, SB	NB, CL, SB			
23-Aug-24	NB, CL, FP, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB
8-Sep-24	NB, CL, FP, SB	NB, CL, SB			
23-Sep-24	NB, CL, FP, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB
6-Oct-24	NB, CL, FP, SB	NB, CL, SB			
21-Oct-24	NB, CL, FP, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB	NB, CL, SB

Additionally, a 10µm plankton net was used to collect a concentrated algal sample from within the top 3m of the CL water column. Those samples were examined to create a list of all alga genera. All the important genera were photographed in the laboratory using a Wolfe DigiviTM CVM Microscope with Motic Images Plus 3.0 software. During each algae sample collection visit, Secchi transparency and vertical profile data were collected at NB, CL, FP, and SB sites.

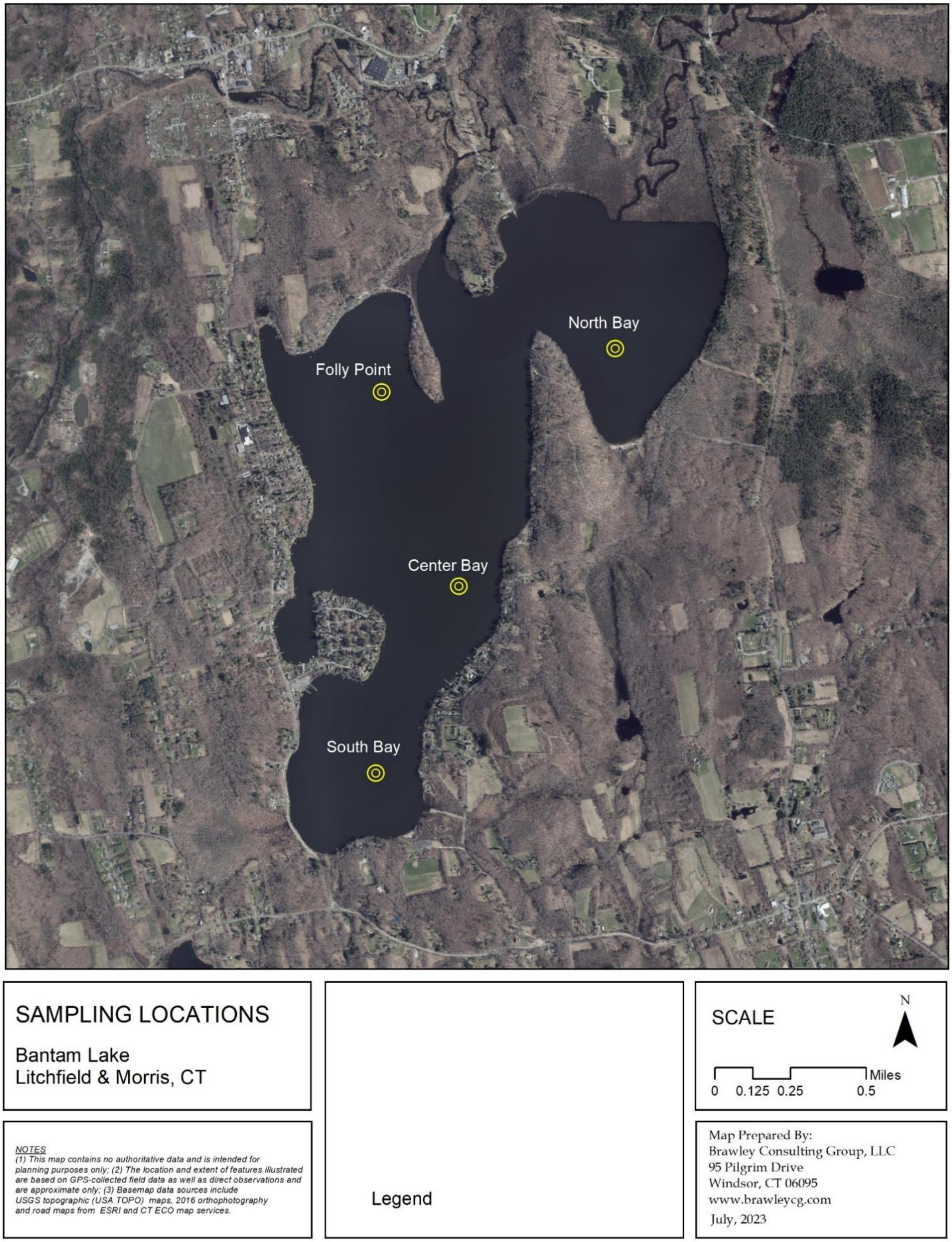

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Figure 3. Sampling Locations Map for Bantam Lake.

IV. Temperature and Oxygen Profiles

Much of the data collected from the water column, e.g., temperature and dissolved oxygen, has been displayed as *isopleth* plots in the following sections of this report. Measurements of water quality variables are displayed as shades of color throughout the water column at each depth and for all dates when data was collected. Values were then interpolated between depths and dates. Variables of the same value (and color) were connected between dates regardless of depth to create a theoretical representation of change throughout the water column over time. All numeric field data and laboratory data have been compiled in Appendix A.

The water temperature and oxygen profile data and isopleth plots provided a view into the thermal and oxygen dynamics of the lake and impacts that seasonal stratification and mixing, resulting from temperature/density differences between depths, have on those dynamics. In shallow New England lakes, or shallow sites in deep lakes, stratification can occur. When a lake is thermally stratified, a middle transitional layer (known as the metalimnion) often separates the upper warmer mixed layer (epilimnion) from lower colder waters below (hypolimnion). Within the boundaries of the metalimnion is the thermocline, which is the stratum where the temperature/density change and resistance to mixing are the greatest. Stratification may be short in duration at shallow sites because wind energy can mix the water column. In deeper lakes or sites, stratification is not as easily broken down by wind energy.

An oxygen concentration of 5mg/L is generally considered the threshold that delineates favorable conditions for most organisms requiring oxygen in freshwater systems. As concentrations decrease below that threshold, conditions start to become stressful for many aquatic organisms. Minimum oxygen requirements for fisheries in Connecticut’s lakes and ponds range from 4 to 5 mg/L for cold-water fish (e.g., trout), 2 mg/L for cool-water fish (e.g., walleye), and 1 to 2 mg/L for warm-water fish (e.g., bass and panfish; Jacobs and O’Donnell 2002).

The loss or absence of oxygen at the bottom of the water column modifies the chemical environment compared to conditions when oxygen is present. These “anoxic” conditions and resulting “reduced” environment result in the dissolution of certain compounds (e.g., iron phosphate) in the sediments that can then diffuse in the interstitial waters between sediment particles and eventually into the waters above the sediments.

The four Bantam Lake sites vary in depth from approximately 8 meters at CL to 4 meters at SB. Water column temperatures and patterns of mixing and stratification were related to site depths and location. Water temperatures near the surface of all sites were just above 10°C in early April and were slightly lower in late April (Fig. 4). Differences between temperatures at top and bottom of the water column in April varied by site: <1 – 5 °C at CL and FP; <1 – 3 °C at SB; and from 2 – 4 °C at NB.

Surface water temperatures gradually warmed to the season highs of 26 to 28°C by mid-July and through early August. Temperatures near the bottom warmed as well but not at the same rate as those near the surface. By mid-July the greatest differences between surface and bottom temperatures were observed. At CL differences were approximately 10 – 11 °C, while 6 – 9 °C at NB. At FP, the greatest temperature gradients were observed from mid-June to early July when surface temperatures were 22 to 26 °C but bottom temperatures were 7 – 7.5 °C lower. At SB, differences between surface and bottom temperatures never exceeded 3 °C.

At the shallow SB site, a thermocline was only observed in late April and resistance to mixing was not strong. The SB water column was mixed on the other 14 sampling dates of the 2024 season.

The deep CL site was stratified from late April through early August (Fig. 4). A thermocline was observed between 5 and 7 meters of depth from early June to early August. Resistance to mixing at CL was strong (RTRM >80) on all June and July dates and in early August.

The NB and FP sites, both with intermediate depths, had similar patterns of stratification and mixing. Both were stratified from late April through early August (Fig. 4). Resistance to mixing did not become strong until early June at NB. By mid-June, the thermocline was 1 meter or less above the bottom with strong resistance to mixing at both sites. The thermocline persisted till early August but resistance to mixing weakened. The water columns were mixed at both sites after early August.

Anoxic conditions were first observed in mid-May at the very bottom of the CL and FP water columns. By early June, NB was also anoxic at the bottom. The NB and CL water column exhibited anoxic oxygen concentrations (<1 mg/L) up from the bottom to 5 meters of depth in mid-July and early August. Also, in mid-July through early August anoxic waters were observed extending above the thermocline at the NB, CL, and FP sites. Anoxic conditions persisted in the water column at CL through early September (Fig. 5). At the NB and FP sites, anoxic conditions persisted only through early August. Anoxic conditions were never observed in the SB water column.

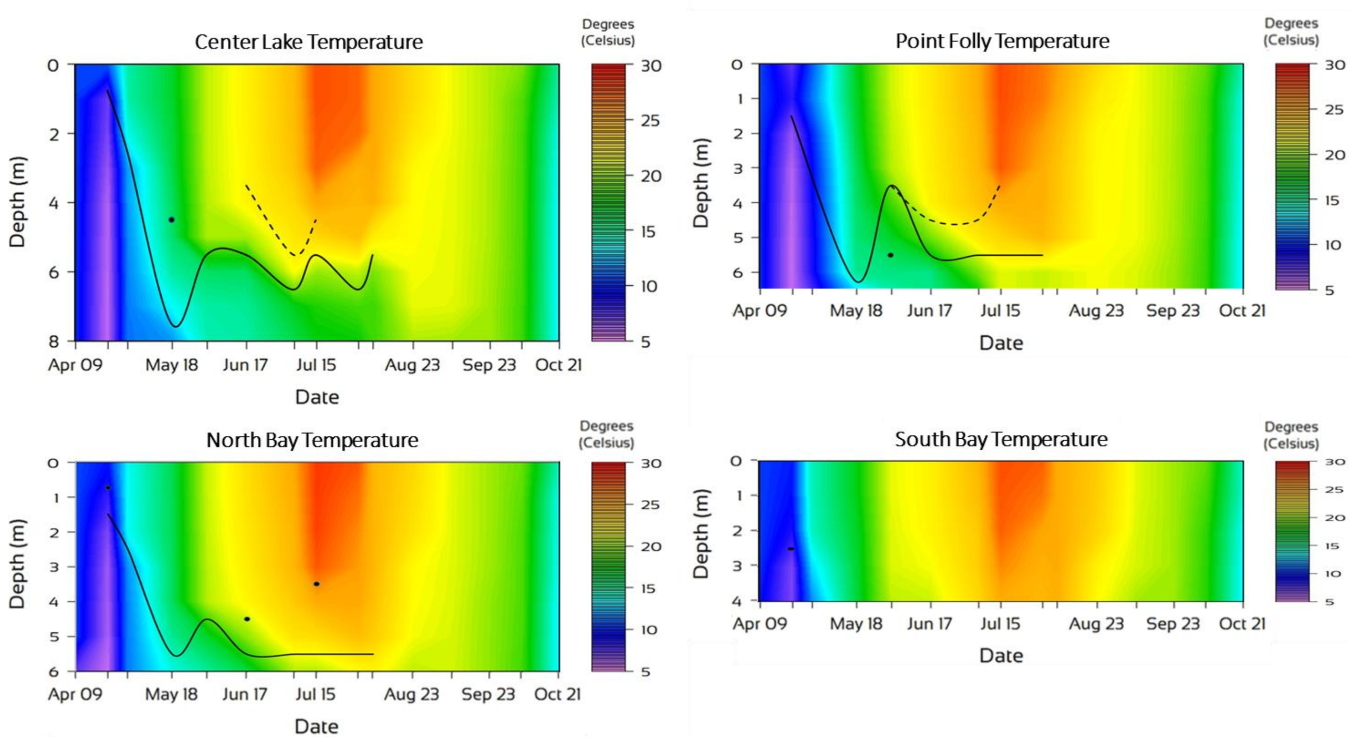


Figure 4. Temperature isopleth charts for the Center Lake, North Bay, Folly Point, and South Bay monitoring sites. The solid black lines and solid black circles represent the position of the thermocline. The black dashed lines and solid black triangles represent the upper and/or lower metalimnetic boundaries. Lines were used when a thermocline or metalimnetic boundary was observed in consecutive sampling events.

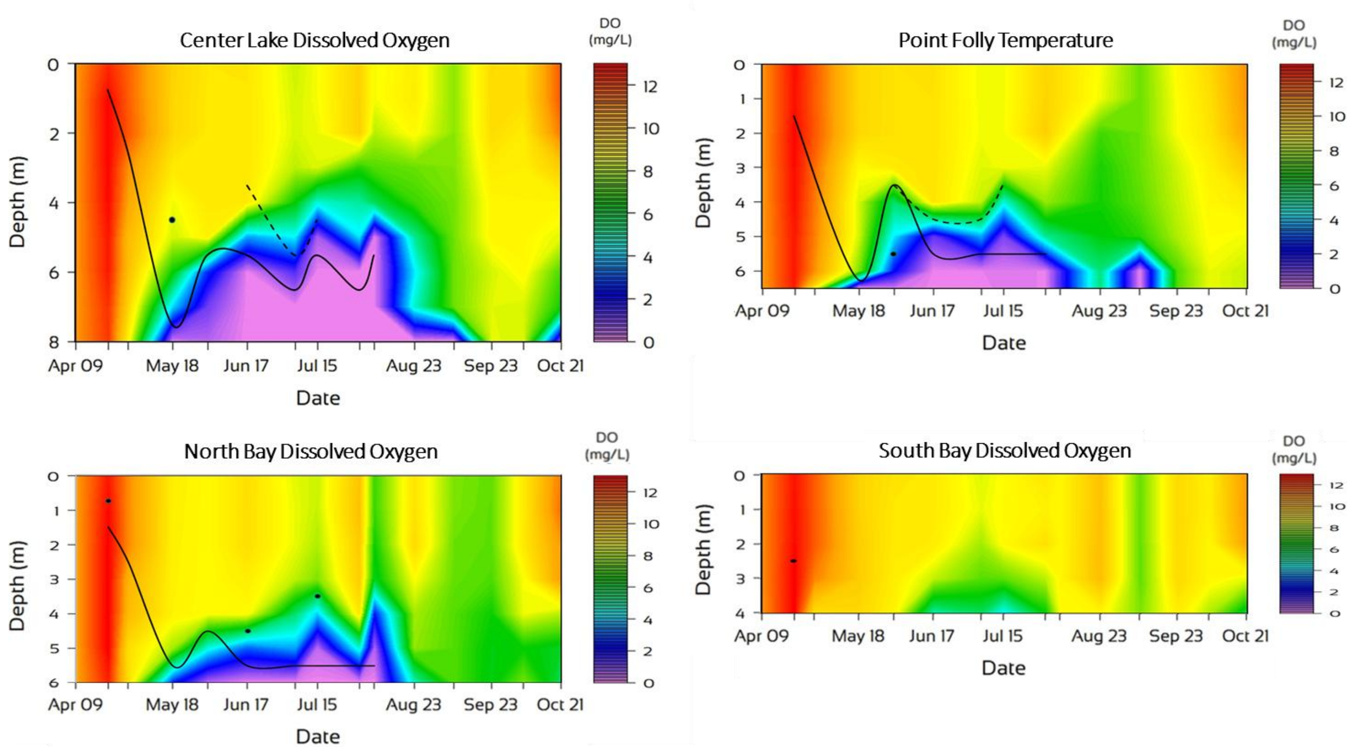


Figure 5. Dissolved oxygen isopleth charts for the Center Lake, North Bay, Folly Point, and South Bay monitoring sites. The solid black lines and solid black circles represent the position of the thermocline. The black dashed lines and solid black triangles represent the upper and/or lower metalimnetic boundaries. Lines were used when a thermocline or metalimnetic boundary was observed in consecutive sampling events.

V. Trophic Characteristics

Several water quality parameters measured were used to assess the trophic status of Bantam Lake. A lake's trophic status is based on the level of primary productivity it can and does support and is determined with variables that limit or reflect algal productivity, including phosphorus and nitrogen concentrations, Secchi disk transparency, and chlorophyll-*a* concentrations (See Table 3). Lakes supporting low levels of algal productivity are typically clearer and are referred to as oligotrophic lakes; lakes supporting high levels of productivity are more turbid and are termed eutrophic or highly eutrophic. It is generally those eutrophic or highly eutrophic lakes that experience regular and intense algal blooms. Lakes with characteristics between oligotrophic and eutrophic conditions can lie within several categories of mesotrophic conditions. Mesotrophic and even oligotrophic lakes can experience algal blooms but those are generally much less intense and infrequent.

Based on the sampling data and classification criteria in Table 3, the trophic status of Bantam Lake in 2024 was eutrophic**.**

Table 3. Trophic classification criteria used by the Connecticut Experimental Agricultural Station (Frink and Norvell, 1984) and the CT DEEP (1991) to assess the trophic status of Connecticut lakes. The categories range from oligotrophic or least productive to highly eutrophic or most productive.

Trophic Category	Total Phosphorus (µg / L)	Total Nitrogen (µg / L)	Summer Chlorophyll-*a* (µg / L)	Summer Secchi Disk Transparency (m)
Oligotrophic	0 - 10	0 - 200	0 - 2	>6
Early Mesotrophic	10 - 15	200 - 300	2 - 5	4 - 6
Mesotrophic	15 - 25	300 - 500	5 - 10	3 - 4
Late Mesotrophic	25 - 30	500 - 600	10 - 15	2 - 3
Eutrophic	30 - 50	600 - 1000	15 - 30	1 - 2
Highly Eutrophic	> 50	> 1000	> 30	0 - 1

A. Secchi Disk Transparency

Secchi disk transparency is a measure of how much light is transmitted through the water column. Light transmission is influenced by several variables including the quantity of inorganic and organic particulate material in the water column that absorbs or reflects light. In the open water environment, Secchi disk transparency is inversely related to algal productivity, i.e., the more algae in the water, the less Secchi transparency will be; the less algae in the water, the greater Secchi transparency will be.

Light in lakes is important for several reasons, particularly for its role in open water photosynthesis and algal productivity. As light diminishes with depth, so does photosynthetic potential. Since photosynthesis decreases with depth, there is a depth where oxygen produced from algal photosynthesis is equal to the oxygen consumed via

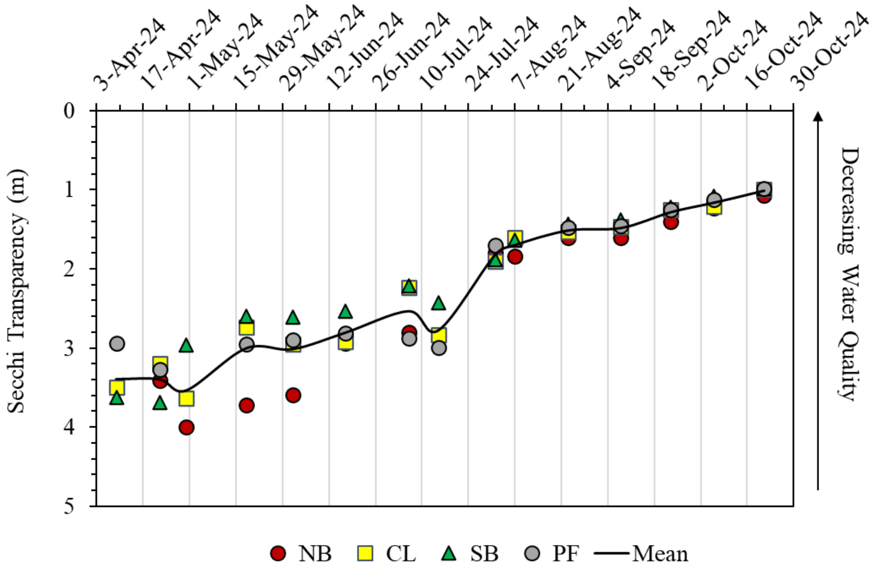


Figure 6. Secchi disk transparency measurements from North Bay (NB), Center Lake (CL), South Bay (SB) and Point Folly (FP) sites on Bantam Lake in 2024. The solid black line represents the lake average.

algal cellular respiration. That is referred to as the Compensation Point and is estimated by multiplying the Secchi disk transparency by 2 (see below).

Secchi disk transparency was measured at Bantam Lake a total of 58 times in 2024. Data was collected 15 times at NB, CL, SB and 13 times at FP. The lake average for the season was 2.28 meters. Season site averages for the four sites ranged from 2.15 meters from SB to 2.49 meters recorded at NB but were not significantly different ($p>0.05$). The lake summer month (July through September) average was 1.88 meters and characteristic of eutrophic conditions. Site summer averages ranged from a low of 1.74 meters from the SB site to 1.99 meters from the NB site. Summer site averages were not significantly different ($p>0.05$).

Secchi disk transparency gradually decreased from early April through mid-July. A more precipitous loss of transparency occurred between mid-July and early August (Fig. 6). Afterwards, transparency declined gradually through the end of the season. The NB site exhibited better transparency than other sites from late April through late May. Before and after that period, NB Secchi transparency was like that of the other sites especially in August through late October.

B. Chlorophyll-a Concentrations

Chlorophyll-*a* is a photosynthetic pigment common to all freshwater algae and cyanobacteria and a useful surrogate measurement for algal biovolume in the water. Concentrations reported here were reflective of the algal productivity in the top three meters of the water column where the composites water samples for these analyses were collected.

Concentrations were analyzed in 21 samples (7x each at the NB, CL, and SB sites) and ranged from a season low of 1.9 µg/L in early April at NB to a high of 63.8 µg/L in late October at CL (Fig. 7). The season average was 16.7 µg/L and the summer month (July – September) average was 18.5 µg/L. Both were characteristic of eutrophic conditions (Table 3).

Concentrations were lowest in early April when the lake average was 2.8 µg/L. The average only increased to 7.4 µg/L by mid-July. However, by late August and late September the lake averages were 21.8 and 26.4 µg/L, respectively. The highest lake average of 47.1 µg/L was from the late October samples.

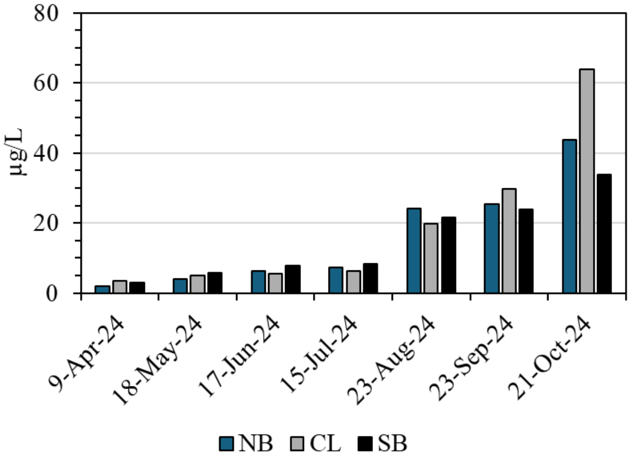


Figure 7. Chlorophyll-*a* concentrations measured at the North Bay (NB), Center Lake (CL), and South Bay (SB) sites in Bantam Lake in the 2024 season.

Site season and summer month averages were not significantly different (p>0.05).

C. Total Phosphorus Concentration

Algae and cyanobacteria require a variety of micro- and macronutrients to grow. In most freshwater systems, phosphorus is the nutrient that limits algae growth (i.e., the limiting nutrient) since it is usually the least available relative to algal metabolic requirements. Therefore, total phosphorus (the sum of particulate and dissolved forms of phosphorus) also serves as a measure of productivity in most lake assessments.

Epilimnetic total phosphorus concentrations ranged from a low of 12 µg/L at the SB site to 47 µg/L from the CL site and averaged 27 µg/L for the season. That average was representative of late mesotrophic conditions (Table 3). The epilimnetic lake averages in August through October were all >30 µg/L and characteristic of eutrophic conditions. Seasonal site averages at NB, CL, and SB were 27, 30, and 21 µg/L, respectively, and were not significantly different (p>0.05). On average, epilimnetic concentrations were lower in samples collected in May, June, and July with lake averages of 17 to 20 µg/L. The early April average was higher at 29 µg/L as were lake averages from August, September, and October that were 31, 38 and 34 µg/L, respectively (Fig. 8).

Metalimnetic total phosphorus concentrations mirrored those measured in the epilimnion. Season metalimnetic averages for the NB, CL, and SB sites were 27, 25, and 22 µg/L, respectively, and not statistically different from the corresponding epilimnetic averages. Like the epilimnetic concentrations, metalimnetic concentrations were on average higher in April, lowest in May and June, and highest in September and October when the lake averages were 33 and 36 µg/L, respectively.

Hypolimnetic concentrations exhibited much greater temporal and site variability. The SB site had the least variability, averaged 30 µg/L which was the lowest of the three site averages, and was not significantly different from epilimnetic and metalimnetic averages at that site ($p>0.05$). The season high at SB was 59 µg/L which was measured in the early April sample and the low was 12 µg/L which was measured the following month. Hypolimnetic concentrations generally increased after mid-May up to 40 µg/L from by late September and only decreased to 34 µg/L by late October.

Like the SB average, the average hypolimnetic total phosphorus concentration at NB was 39 µg/L, and not significantly different from the site epilimnetic or metalimnetic averages ($p>0.05$). The season low at NB of 21 µg/L in early April steadily increased to a season high of 64 µg/L by mid-July. Starting in late August, that pattern was repeated with initial concentrations of 23 µg/L increasing through the end of the season to 50 µg/L (Fig. 8).

The CL site exhibited the greatest hypolimnetic total phosphorus variability with a season low of 22 µg/L measured in early April which increased to a season high of 122 µg/L by mid-July. Concentrations decreased to 30 µg/L by late September, then increased to 50 by late October. The CL season average of 61 µg/L was significantly higher than corresponding epilimnetic and hypolimnetic averages, and significantly different than hypolimnetic averages at the other two sites ($p<0.05$).

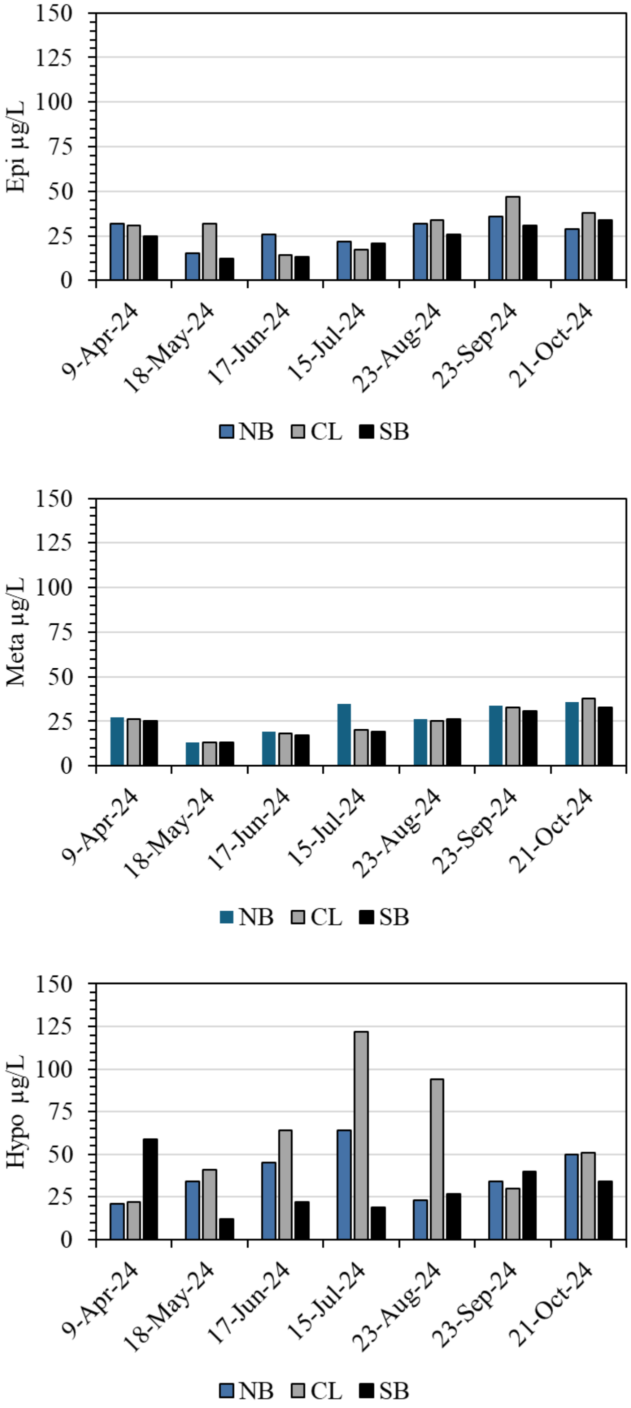


Figure 8. Epilimnetic (Epi; top panel), metalimnetic (Meta; middle panel) and hypolimnetic (Hypo; bottom panel) total phosphorus concentrations at the North Bay (NB), Center Lake (CL), and South Bay (SB) sites in Bantam Lake in the 2024 season.

D. Total Nitrogen and Ammonia

Nitrogen is typically the second most limiting nutrient for algae growth in freshwater systems and also useful for assessing trophic conditions in lakes. It can be present in several forms in lake water. Ammonia – a

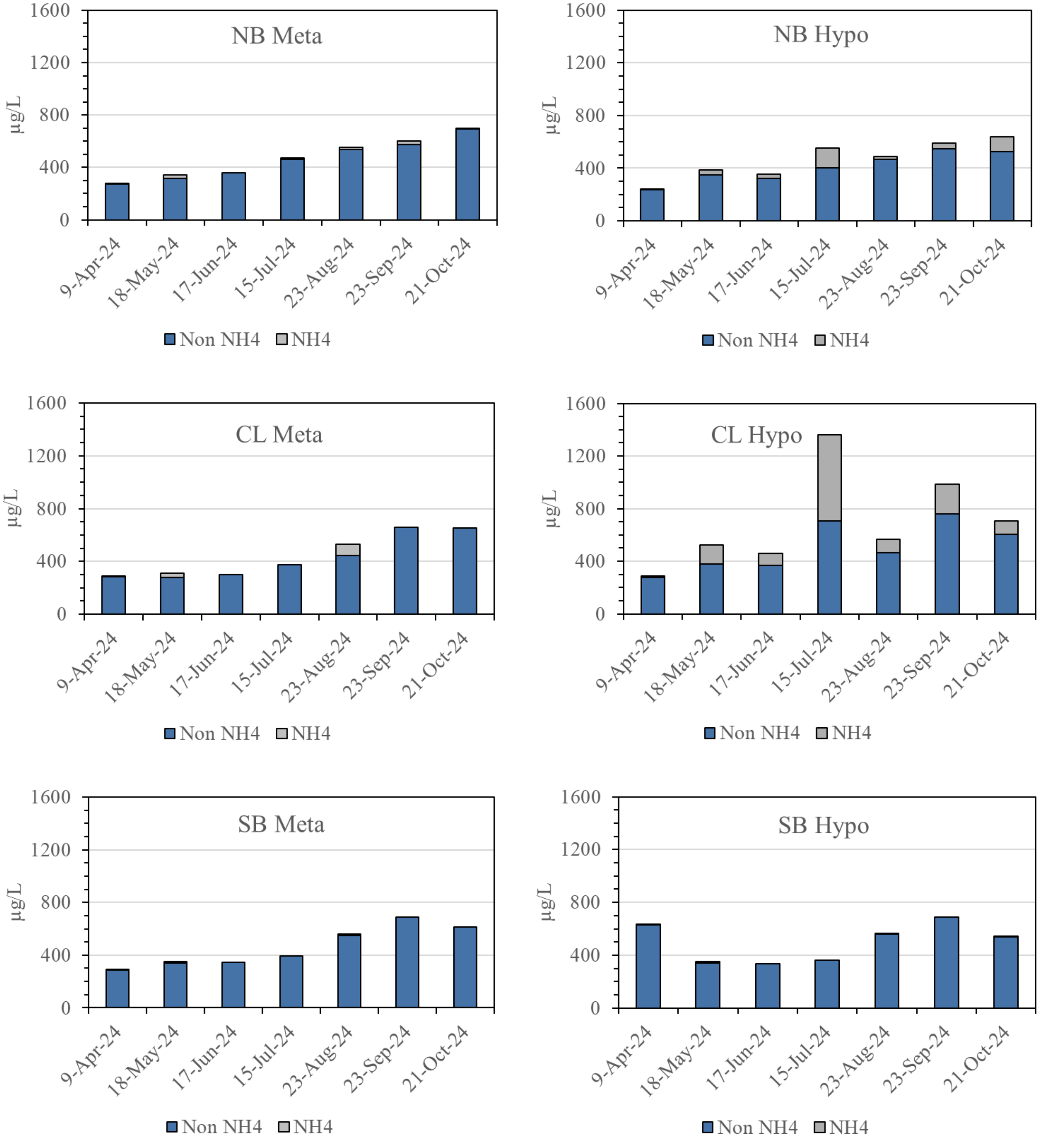


Figure 9. Total nitrogen displayed as ammonia (NH4) and nitrogen not measured as ammonia (non NH4) in the metalimnion (Meta) and hypolimnion (Hypo) at North Bay (NB), Center Lake (CL), and South Bay (SB) sites of Bantam Lake in 2024.

reduced form of nitrogen – is important because it can affect the productivity, diversity, and dynamics of algal and plant communities. Ammonia can be indicative of internal nutrient loading since bacteria will utilize other forms of nitrogen oxide compounds (e.g., nitrite and nitrate) in lieu of oxygen for cellular respiration under anoxic conditions, resulting in ammonia enrichment of the hypolimnion. Once available nitrogen oxides are used up, oxidized iron, including iron phosphate, will be utilized for cellular respiration, which results in the enrichment of phosphorus in the hypolimnion.

Total nitrogen accounts for all forms of nitrogen in the water including reduced forms like ammonia, organic proteins, nitrate and nitrite. Nitrate and nitrite are rarely detected above minimum detection limits due to those being quickly used up by algae and plants. For this program, total nitrogen and ammonia were measured. Total nitrogen was graphically represented in Figure 7 as that measured as ammonia and that which is not ammonia for samples collected in the metalimnion and hypolimnion.

Season epilimnetic total nitrogen averages for NB, CL, and SB were 456, 491, and 445 µg/L and not significantly different ($p>0.05$). The lake average was 464 µg/L and indicative of mesotrophic conditions as were all site averages. The fractions of total nitrogen in the epilimnion comprised of ammonia were small, and on average between 2 and 8 µg/L among the three sites. Ammonia was not detected in nearly half of all samples collected in the epilimnion.

Metalimnetic total nitrogen levels were similar to corresponding epilimnetic levels. Season averages at NB, CL, and SB were 471, 446, 464 µg/L and the lake average was 460 µg/L. Epilimnetic and metalimnetic season averages from all three sites were not significantly different ($p>0.05$). The portion of total nitrogen in the metalimnion that was ammonia was also small, and averaged 12, 18, 2 µg/L at NB, CL, and SB, respectively. Several notable ammonia concentrations in the metalimnion included 34 and 87 µg/L at CL in mid-May and late August, respectively (Fig. 9).

Hypolimnetic total nitrogen levels were more variable than epilimnetic or metalimnetic levels, particularly at the CL site where the site average was 699 µg/L. That average was significantly higher than the NB and SB hypolimnetic averages of 465 and 490 µg/L, respectively ($p<0.05$ for NB; $p<0.005$ for SB). The hypolimnetic average at CL was also significantly greater than the epilimnetic and metalimnetic averages at the same site. The Hypolimnetic averages at NB and SB were not significantly different than corresponding epilimnetic and hypolimnetic averages ($p>0.05$).

At NB, hypolimnetic total nitrogen generally followed a seasonal pattern similar to that observed in the epilimnion and metalimnion. Lowest concentration was measured in the early April sample and the highest in the late October sample. Ammonia was measured in all hypolimnetic samples total nitrogen but was minor with only notable concentrations of 153 and 114 µg/L measured in mid-July and late October, respectively. The average for season was 58 µg/L.

The SB hypolimnetic total nitrogen concentrations decreased from 632 µg/L in early April to between 337 and 364 µg/L from mid-May to mid-July. Concentration increased to a season high of 688 µg/L by late September and remained high at the end of the season at 541 µg/L. The percentage of hypolimnetic total nitrogen that was ammonia was very minor. The greatest amount of ammonia measured in the SB hypolimnion was 10 µg/L measured in mid-May. The SB hypolimnetic ammonia average was 3.5 µg/L.

At CL, hypolimnetic ammonia comprised more of the monthly total nitrogen, especially in mid-July, when nearly 50% of the total concentration of 1361 µg/L was ammonia. The mid-July total nitrogen and ammonia concentrations were season highs. The season lows of 287 µg/L total and 8 µg/L measured as ammonia occurred in early April. There was an oscillating high / low pattern of ammonia and that nitrogen which was not ammonia from mid-July through the end of the season (Fig. 9). In other words, highs in mid-July and late September were followed by lower measurements in the following months. The hypolimnetic ammonia average at CL was 191 µg/L which was significantly greater than the SB average ($p<0.05$), but not significantly different from the NB average.

E. Redfield Ratios

Limnologists frequently use the Redfield ratio of 16 (16:1 nitrogen to phosphorus) to determine whether nitrogen or phosphorus is the most limiting nutrient in a freshwater system (Redfield 1958). The ratio is molar-based and when converted to mass, 7.2 µg/L is the threshold. Values lower than the threshold are indicative of nitrogen limitation while ratios above 7.2 µg/L indicate phosphorus limitations. Nitrogen limitation can favor cyanobacteria productivity due to the ability of some cyanobacteria to harvest elemental nitrogen dissolved into the water from the atmosphere, aka nitrogen fixation. Other algae taxa do not possess this ability.

Redfield ratios were determined for all dates and strata. Epilimnetic ratios ranged from 10 to 27 and averaged 18.5, while metalimnetic ratios ranged from 10 to 27 and averaged 19.0. Both were indicative of phosphorus limitation. Hypolimnetic ratios ranged from 6 to 2 and averaged 15.1 which was mostly indicative of phosphorus limitation. Mid-June and late August hypolimnetic ratios at CL were ≥7. Similarly at NB, hypolimnetic ratios of 8 and 9 were determined for mid-June and mid-July.

VI. Algae Community Dynamics

Algae have been used in ecological assessments of lakes for over 100 years (Stevenson 2014). The compositions, concentrations, and biomasses of assemblages of algae in the water column (i.e., phytoplankton) can be diagnostic of environmental conditions in a lake. For example, a lake dominated by Cyanophyta (aka cyanobacteria or blue-green algae) with high cell concentrations and biomass often have high nutrients concentrations and poor water clarity. Algae communities that are more diverse and include species from the Bacillariophyta (aka diatoms), Chrysophyta (aka golden algae), and Chlorophyta (aka green algae), with lower cell concentrations and biomass, are reflective of lower nutrient conditions and good clarity.

High concentrations of cyanobacteria can form harmful algal blooms, which can present public health risks due to toxins that some cyanobacteria can produce. The State provides municipalities guidance on inland water beach closings and reopening due to cyanobacteria (CT DPH & CT DEEP 2024). A ranking system identifies three categories of risk that are based on visual assessments but also have corresponding cyanobacteria cell concentrations. The least amount of risk is associated with Visual Rank Category 1 that has corresponding cyanobacteria cell concentrations of 0 – 20,000 cells/mL. The greatest risk is assigned to Visual Rank Category 3 with corresponding cell concentrations are >100,000 cells/mL. In between Visual Rank Category 1 and 3 is Visual Rank Category 2 with its corresponding cyanobacteria cell concentrations of 20,000 – 100,000 cells/mL.

A. Important Taxa and Richness

Sixty genera of algae or cyanobacteria were identified in samples collected during the 2024 season. The taxonomic group with the greatest richness (greatest number of genera) were the Chlorophyta (aka green algae) with 28 identified genera. Cyanophyta (aka cyanobacteria or blue-green algae) had the next greatest richness with 11 genera identified. Bacillariophyta (aka diatoms), Chrysophyta (aka golden algae), and Pyrrhophyta (aka dinoflagellates) were represented by 8, 5 and 4 genera, respectively. Two other taxonomic groups were represented by 3 genera or less (see Appendix B).

B. Cell Concentrations and Relative Abundances

Cyanobacteria dominated the planktonic algae community for much of the season. The only time cyanobacteria did not comprise a minimum of 55% of all cells counted at the three sites was between late April and early June. During that period, Chrysophyta, with much of those bring the colonial *Uroglenopsis spp*., constituted a large portion of the cells counted (Fig. 10). By mid-June at CL and by mid-July at NB, the relative abundance of cyanobacteria was <90%. Relative abundance is the percentage of cells from a group (e.g., cyanobacteria) in relationship to the total. At SB, the relative abundance of cyanobacteria was >90% on 7 of the 10 times cell counts were performed starting in mid-June.

Cell concentrations, including cyanobacteria, were generally very low at all sites from early April through early July. During that time, cyanobacteria cell concentrations were often <5,000 cells/mL. The one exception occurred in mid-June when cyanobacteria cell concentrations were between 34,000 and 39,000 cells/mL (Fig. 10). Cell concentrations increased by an order of magnitude between mid-July and early August at the NB and CL sites resulting in cyanobacteria concentrations of approximately 130,000 and 119,000 cells/mL, respectively. An increase to 79,000 cells/mL was observed at SB by early August which was not as high as that observed at NB and CL.

Cyanobacteria cell concentrations were lower (between 43,000 and 89,000 cells/mL at NB and CL; between 38,000 and 64,000 cells/mL at SB) from August 7th through the early part of September before exponentially increasing through late September and early October. Season high cyanobacteria cell concentrations at NB and CL occurred in early October and were approximately 200,000 and 400,000 cells/mL, respectively. The season high at SB was approximately 177,000 cells/mL and occurred in late October. Late October cyanobacteria cell concentrations were like early October levels at NB. Concentrations did decrease at CL by late October but were still high at approximately 144,000 cells/mL.

The filamentous cyanobacteria, *Aphanizomenon spp*., was observed throughout the season. In early April, it comprised >70% of all cells counted at the three sites. Abundance decreased from mid- to late April when the colonial golden algae *Uroglenopsis spp*. was the most abundant genus. *Aphanizomenon spp*. then comprised approximately 30 to 50% of all cells counted in mid-May through early June. By mid-June and through late September, a second filamentous cyanobacteria, *Dolichospermum spp*., became co-dominant. Other cyanobacteria genera counted, but at lower concentrations, during that time included *Planktothrix spp*., *Pseudanabaena spp*., *Woronichinia spp*., and *Microcystis spp*. By October, *Aphanizomenon spp*. became the clear dominant genus again, comprising a minimum of 80% of all cells counted at the three sites.

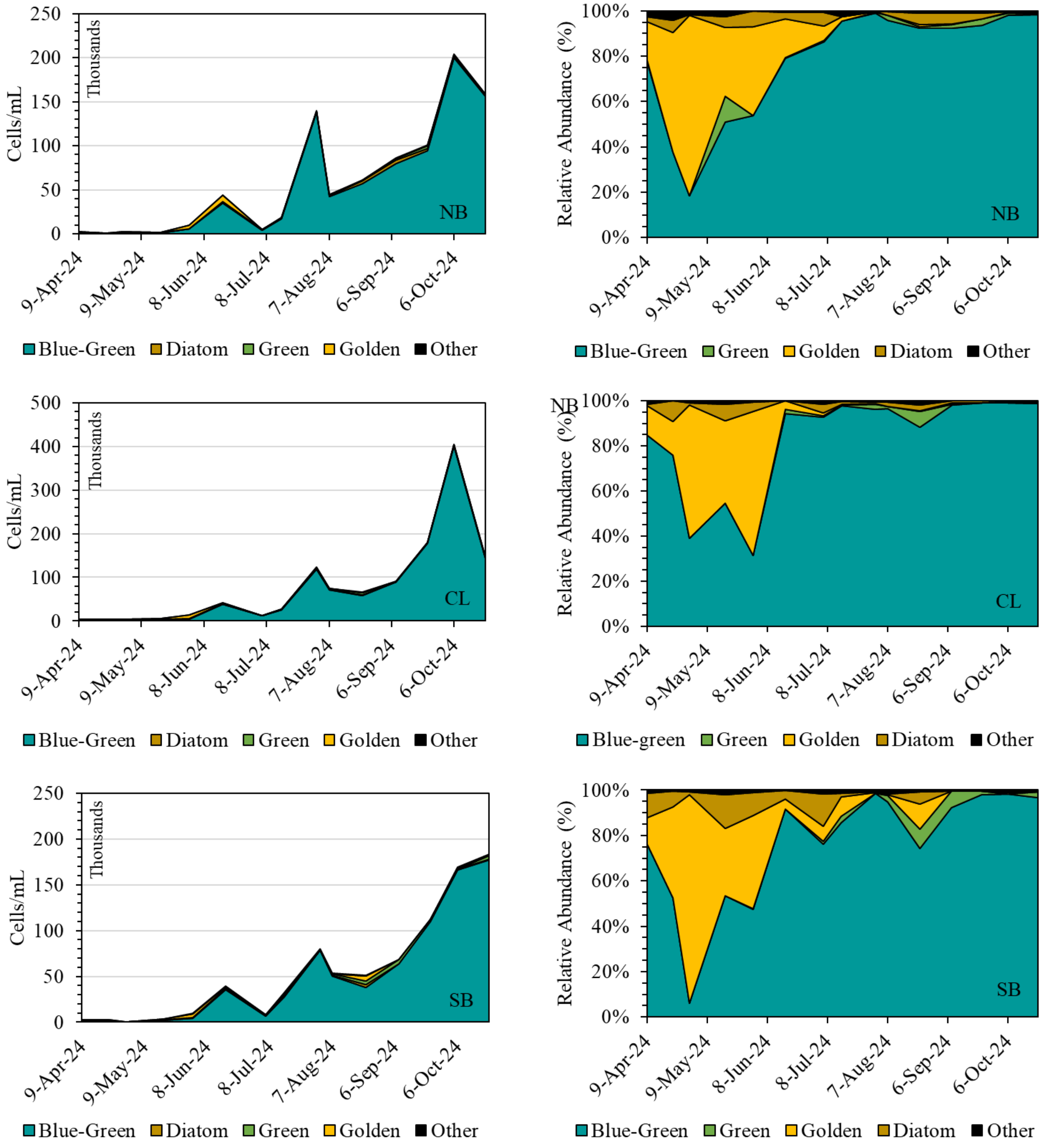


Figure 10. Cell concentrations (left) and relative abundances (right) of major algae taxa counted at the North Bay (NB), Center Lake (CL), and South Bay sites of Bantam Lake in 2024.

C. Cyanotoxin Levels

The cyanobacteria discussed above are all potential toxin producers (CT DPH &CT DEEP 2024, USEPA 2024). There are several different types of cyanotoxins, and different toxic compounds within those groups. Microcystin is a hepatotoxin that primarily affects the liver, and the toxin prescribed as that measured to determine if a beach can be reopened after closing due to a harmful cyanobacteria bloom. The level of 8 µg/L is the threshold for recreational waters under which is considered safe. The BLPA implements a microcystin monitoring program where samples are collected biweekly over a ten-week period and tested for microcystin. Results of analyses of samples collected at Bantam Lake never exceeded 0.3 µg/L (Table 4).

Table 4. Microcystin concentrations in samples collected from North Bay and Center Lake in Bantam Lake between July 7 and August 25, 2024.

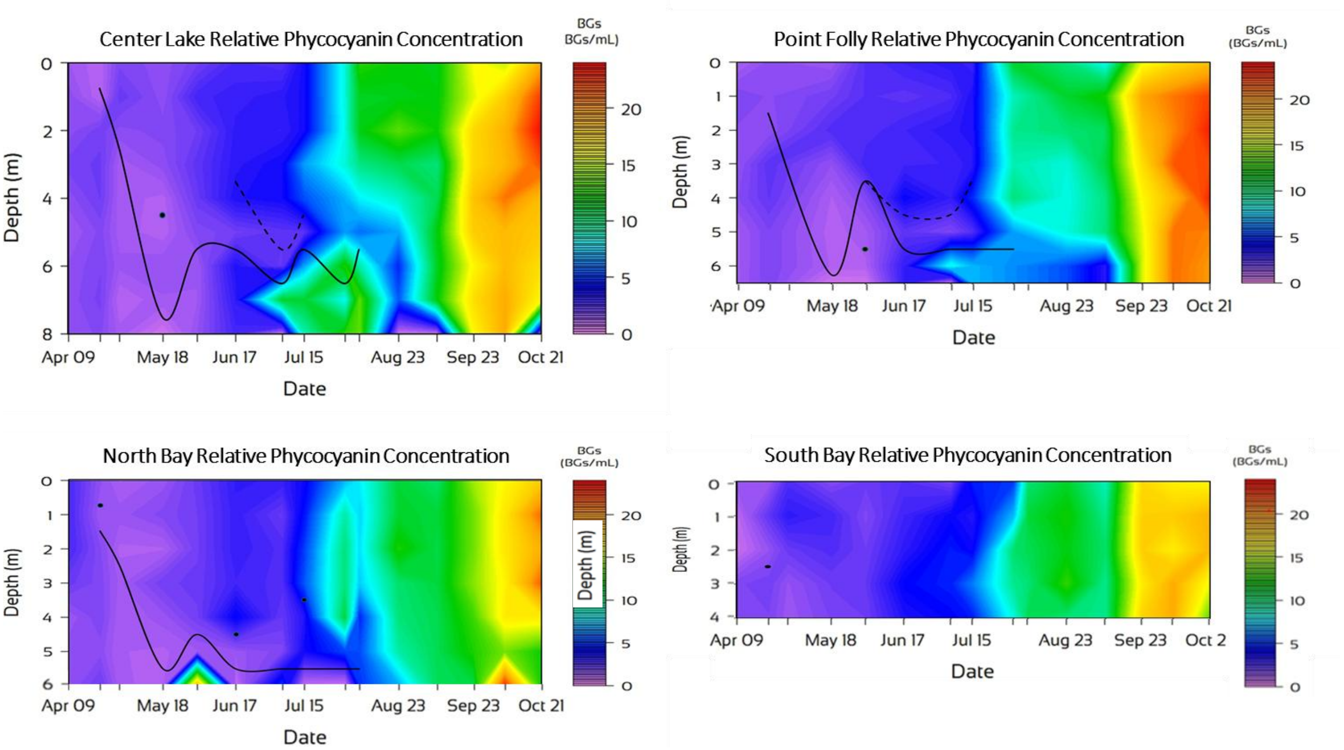
Week of	**North Bay**	**Center Lake**
	µg/L	
July 7	0.036	0.068
July 14	0.127	0.037
July 28	0.024	0.061
August 4	0.108	0.043
August 25	0.211	0.145

D. Temporal and Spatial Distribution

Cell concentrations and chlorophyll-*a* concentrations are useful in understanding algae and cyanobacteria in the top three meters of the water column where those samples for those analyses are collected. Instrumentation used by BCG allows for measuring a relative biomass of the cyanobacteria throughout the water column by measuring relative phycocyanin concentrations.

Phycocyanin is an auxiliary photosynthetic pigment unique to cyanobacteria and relative concentrations were measured with a fluorimeter incorporated into the sensor array of the Eureka Manta II multiprobe. Fluorimeters work on the principle that a particular substance fluoresces at a specific wavelength when light of another wavelength is directed on that substance. The fluorimeter in our instrumentation emits a wavelength that interacts with phycocyanin. This sensor is not calibrated with known concentrations of phycocyanin, so measurements are not quantitative; instead, the measurements are relative to other measurements in the water column and to measurements on other dates. Relative phycocyanin concentrations measured at the four sites were used to construct isopleth charts to display cyanobacteria biomass across time and depths.

Relative phycocyanin isopleths revealed low cyanobacteria concentrations throughout the water column through mid-July at all sites (Fig 11). Afterwards, the concentrations increased and differentiation in the water column became conspicuous with higher concentrations observed above 5 meters of depth except at the SB. There at SB, higher concentrations were observed in the lower depths of the water column. Highest concentrations were observed at all sites starting in late September and remained high throughout the remainder of the sampling season.


Center Lake Relative Phycocyanin Concentration

Point Folly Relative Phycocyanin Concentration

North Bay Relative Phycocyanin Concentration

South Bay Relative Phycocyanin Concentration

Figure 11. Isopleth plots of relative phycocyanin concentration (BGs) at the Center Lake, North Bay, Folly Point, and South Bay sites of Bantam Lake in 2024. The dashed black lines and black circles represent the position of the upper boundary of the metalimnion; the solid black lines and black circles represent the position of the thermocline.

VII. Water Chemistry

Much like biological variables like Secchi transparency, chlorophyll-*a*, and phosphorus concentrations are used to assess current trophic conditions and detect changes from past conditions, so too do chemical characteristics of the lake water provide methods of detecting changes over time. Below, specific conductance, oxidation reduction potential, pH and alkalinity, and base cation and anion concentrations measured in 2024 were described. Later in the report, those will be compared to past levels.

A. Specific Conductance

Conductivity is a surrogate measurement for the sum of the ionized minerals, metals, and salts in the water and a measure of water's ability to transmit an electrical current. Field data collections included measures of both conductivity and specific conductance and were measured in microsiemens per cm (µS/cm). Specific conductance is the same as conductivity but standardized to a set water temperature of 25°C. Specific conductance was reported below since temperature influences conductivity and – in the field – temperature varies with depth and date.

Specific conductance is an important metric in limnological studies due to its ability to detect pollutants and/or nutrient loadings. Specific conductance can also have an influence on organisms that inhabit a lake or pond; particularly, algae. The composition of plant and algal communities has been shown to be related, in part, to conductivity levels in lakes (e.g., June-Wells et.al. 2013, Siver 1993, McMaster & Schindler 2005). As was done with temperature and oxygen profile data, specific conductance and oxidation-reduction potential data have been displayed as an isopleth chart.

Specific conductance levels were lowest in April when they exhibited the least variability throughout the water column (Fig. 12). At that time at the NB, CL, and FP sites, levels were between 138 and 149 µS/cm with slight increases with depth. Levels were even lower at SB and ranged from 134 to 141 µS/cm. Specific conductance increased as the season progressed and changes with depth became more pronounced through early August. Levels above the thermocline increased from the low 150s to the low 160s µS/cm. Below the thermocline in early August, levels reached maximums of 271 µS/cm at NB and 247 µS/cm at CL. The FP site exhibited increases under the thermocline but those did not reach the levels measured at the NB and CL sites. Following the mixing of the water column by late August, specific conductance levels became more homogenous throughout the water column. The CL site did exhibit another elevated level of 279 µs/cm at the very bottom of the water column in early September.

The SB site exhibited the same kinds of seasonal increase observed above the thermocline of the other three sites. However, differences in the water column were minor at the SB site.

B. Oxidation-Reduction Potential

The oxidation-reduction potential (aka redox potential or ORP) in lakes refers to the oxidative or reductive state in a particular stratum of the water column; it can provide insight as to whether phosphorus in sediment compounds (e.g. iron phosphate) are changing from a particulate inactive state to a soluble state that can readily diffuses into water between sediment particles and then into overlying waters. Those can then become available to algae and cyanobacteria if mixed or diffused into layers where enough light can be transmitted to support algae growth.

When OPR is ≥200 milli-volts (mV) phosphate remains bound to available iron; at ORP values of <200 mV, iron is reduced and the phosphate that was chemically bound to the iron becomes soluble (Søndergaard 2009). In some cases, a sudden mixing of phosphate-laden bottom waters to the upper reaches of the water column during a storm or wind event can trigger an algae bloom. ORP data collected at the four sites are presented as isopleth plots (Fig. 9).

ORP levels above the thermocline were mostly ≥200 mV for much of the season (Fig. 13). ORP levels below the thermocline at NB, CL, and FP were often <200 mV (Fig. 9). ORP below the thermocline became progressively more reduced, reaching levels of <-100 mV at bottom of the NB water column, and expanding up from the bottom to 5 meters of depth by early August at CL. During the mid-summer sampling events, ORP of <200 mV was measured at and slightly above the thermocline at the CL site.

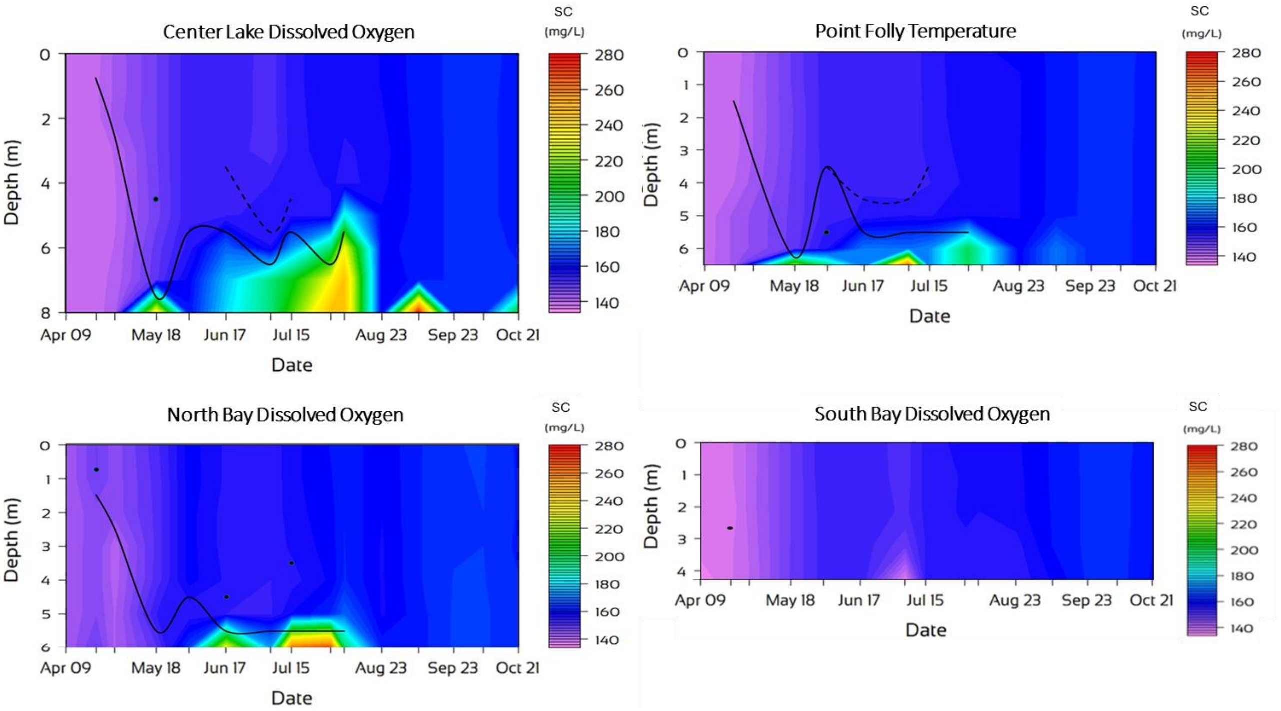


Figure 12. Isopleth plots of water column specific conductance at the Center Lake, North Bay, Folly Point, and South Bay sites of Bantam Lake in 2024. The dashed black lines and black circles represent the position of the upper boundary of the metalimnion; the solid black lines and black circles represent the position of the thermocline.

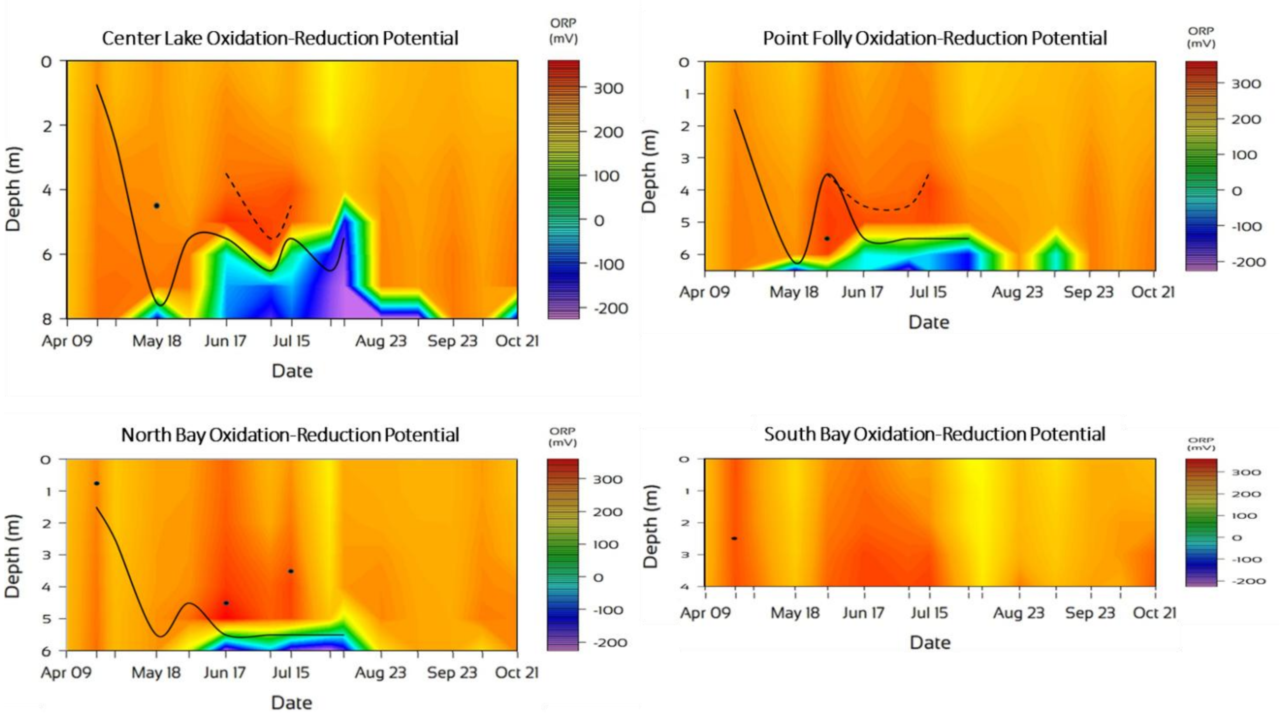


Figure 13. Isopleth plots of water column oxidation-reduction potential at the Center Lake, North Bay, Folly Point, and South Bay sites of Bantam Lake in 2024. The dashed black lines and black circles represent the position of the upper boundary of the metalimnion; the solid black lines and black circles represent the position of the thermocline.

The ORP levels at the SB site were generally always >200 mV. Exceptions occurred in early August when levels were below the 200-mV threshold. On August 1st, levels of 167 to 189 were measured in top two meters of the water column but were between 215 and 234 mV from 3 meters to the bottom. On August 7th, the entire water column was between 168 and 188 mV.

C. pH and Alkalinity

The normal pH of surface waters of lakes in the Northeast can range from approximately 6 to 9 SU (standard units). Very low or very high pH levels will not support diverse fauna and flora in freshwater ecosystems. Algal community composition is influenced by pH. For example, the pH of the water will influence algae community characteristics by determining the type of dissolved carbon in the water column. At pH levels greater than 8.3, bicarbonate is the dominant form of carbon available to the pelagic algal community; the blue-green algae have adaptive advantages over other algae in those conditions in that they can efficiently utilize that form of carbon. Other algal groups are dependent upon carbon dioxide, which is more readily available in water below a pH of 8.3.

Epilimnetic pH levels were similar among the three sites, were between 7.45 to 8.41 standard units (SU), and averaged 7.85 SU for the season. April through June levels of 7.5 to 7.7 SU increased and were between 7.9 and 8.4 SU in August through September. The only exception was the 7.45 SU measured at NB in September.

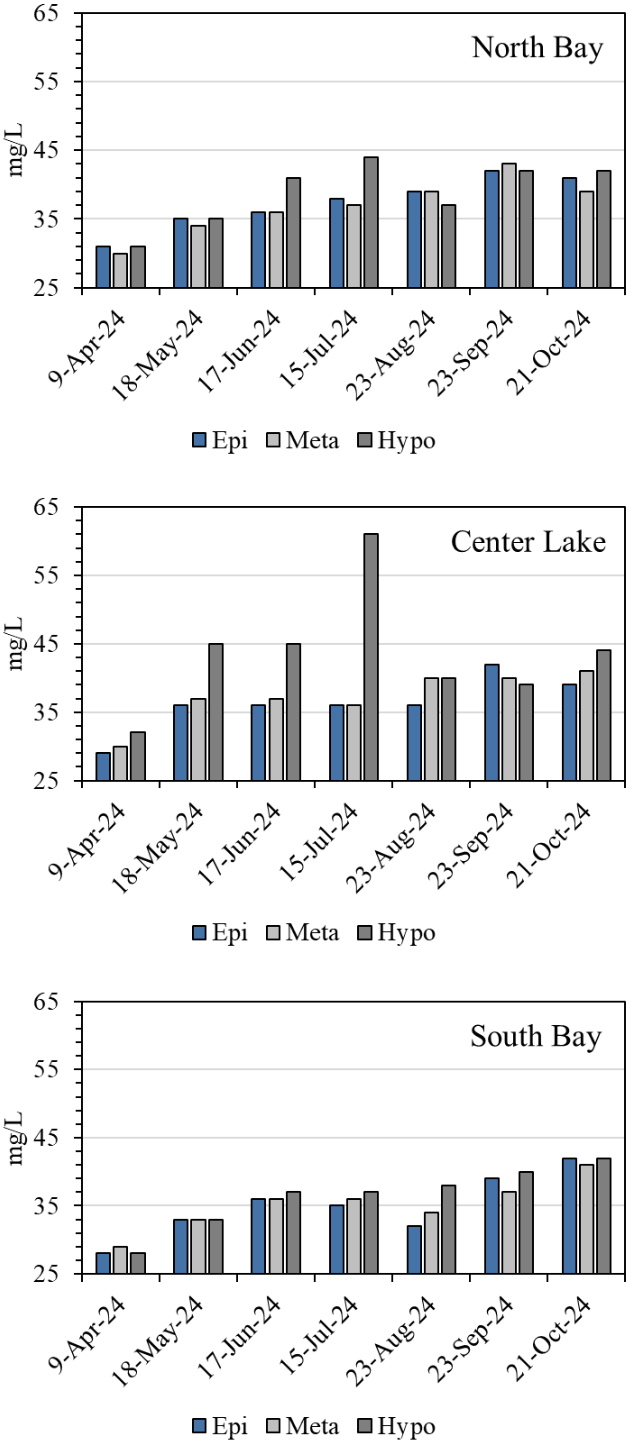


Figure 14. Epilimnetic (Epi), metalimnetic (Meta), and hypolimnetic (Hypo) alkalinity at the North Bay, Center Lake, and South Bay sites on Bantam Lake in 2024.

The lake hypolimnetic pH season average of 7.35 SU was significantly lower ($p<0.0001$) than the epilimnetic average. Hypolimnetic levels were all measured between 6.87 and 8.06 SU. Highest hypolimnetic levels occurred after the lake mixed in early August. Higher epilimnetic and lower hypolimnetic pH levels in deep lakes are common and due to lower carbon dioxide levels in the epilimnion resulting from photosynthesis.

The average pH at the thermocline or mid-depths if the water column was mixed, was 7.47 SU, which was significantly lower than epilimnetic average ($p<0.005$), but not significantly different from the hypolimnetic average ($p>0.05$). Like with epilimnetic and hypolimnetic measurements, the highest levels generally occurred in the latter half of the season.

The season average pH in the metalimnetic layer of 7.5 SU was not significantly different from the averages for the epilimnion and hypolimnion ($p>0.05$). In May, the metalimnetic pH was very similar to the epilimnetic pH, while more like hypolimnetic levels for the remainder of the season.

Alkalinity is a measure of calcium carbonate and provides lake water and its ability to neutralize acid (i.e., buffering capacity). Alkalinity of surface waters is largely influenced by local geology and other watershed characteristics. Alkalinity at the bottom of the water column can also be generated internally from the biologically mediated reduction of iron, manganese, and sulfate via cellular respiration in the anoxic lake sediments, and denitrification of nitrate to elemental nitrogen (Wetzel 2001). For purposes of assessing alkalinity and comparing it between strata and sites, the unit of measure reported by the laboratory, i.e., mg/L, was used.

Epilimnetic and metalimnetic alkalinities generally trend upward with time at NB, CL, and SB (Fig. 15). The lowest concentrations were in early April and were between 29 and 31 mg/L among the three sites. Season highs were observed in late September and late August and ranged from 39 to 43 mg/L. Epilimnetic averages at NB, CL, and SB were 36.8, 35.8, and 33.8 mg/L, respectively. Metalimnetic averages were 36.5, 36.7, and 34.2, respectively. There were no significant differences between epilimnetic and metalimnetic averages at each site and between sites ($p>0.05$).

In contrast, hypolimnetic alkalinity varied from site to site. The least variability was observed at the SB site. There, hypolimnetic alkalinity mirrored that measured in the epilimnion and metalimnion each month and gradually increased over time (Fig. 14). The SB hypolimnetic season average was 35.5 mg/L and not statistically different from the epilimnetic and metalimnetic averages at that or any other site.

The CL hypolimnion alkalinity exhibited the greatest variability. A season low of 32 mg/L was measured in early April. By mid-May the through mid-June, levels had increased to 45 mg/L. A season high of 61 mg/L was measured in mid-July. Hypolimnetic alkalinity decreased precipitously to 40 mg/L by late August. Concentrations increased again between lake September and late October from 39 to 44 mg/L. The CL season average was 43.7 mg/L which was significantly greater than that epilimnetic and metalimnetic averages at SB ($p<0.05$).

Hypolimnetic concentrations at NB were not as variable as those at CL, but more variable than those at SB. Concentrations steadily increased from 31 to 44 mg/L between early April and mid-July, decreased to 37 mg/L by late August, then remained at 42 mg/L for the remainder of the season. The NB hypolimnetic season average was 38.3 mg/L.

D. Cations and Chloride

Base cation and anion concentrations are useful in understanding natural influences (e.g., dissolved salts from bedrock geology) as well as anthropogenic influences in the watershed (e.g., road salts). In most lakes, the dominant base cations in lake waters are calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}) and potassium (K^{+}). Dominant anions include chloride (Cl^{-}), sulfate (SO_4^{2-}), and the alkalinity anions, i.e., carbonate (CO_3^{2-}), and bicarbonate (HCO_3^{-}). Those cations and anions are what collectively create much of the conductivity in lake water. The ratios and other characteristics of those ions can be diagnostic when compared to other lakes, and when compared to levels in the same lake over time.

The UCONN CESE lab who conducted the analyses reported results on a mass basis (mg/L). BCG converted those into their electrochemical equivalents or milliequivalents (meq/L) by dividing the measured mass of an ion by its equivalent weight.[1] This provided meaningful accounting for ionic or electrical charge (positive or negative). Accounting for electric charge is preferable when comparing ion levels to other electrochemical characteristics of lake water, e.g., specific conductance. Ion levels were reported below in both mass and milliequivalents below in Table 5 and Table 6.

Table 5. Base cations (Na = sodium, K = potassium, Ca = calcium, Mg = magnesium) and anion (Cl = chloride, Alk = alkalinity anions of carbonate and bicarbonate) measured at North Bay (NB), Center Lake (CL), and South Bay (SB) at Bantam Lake on three dates in 2024 reported in mg/L.

Site	Date	Na	K	Ca	Mg	Cl	Alk
		mg/L					
NB	9-Apr-24	12.7	1.1	8.3	3.7	20.8	31
	15-Jul-24	12.9	1.3	10.4	4.6	20.3	38
	21-Oct-24	12.8	1.4	10.8	4.7	22.1	41
CL	9-Apr-24	12.1	1.2	8.1	3.6	18.2	29
	15-Jul-24	13.0	1.2	10.2	4.4	20.2	36
	21-Oct-24	12.5	1.4	10.3	4.5	21.7	39
SB	9-Apr-24	12.0	1.1	7.9	3.5	18.5	28
	15-Jul-24	12.6	1.3	9.7	4.3	20.0	35
	21-Oct-24	12.6	1.4	10.7	4.6	22.0	42
	Lake Avg.	12.6	1.3	9.6	4.2	20.4	35.4

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[1] See https://en.wikipedia.org/wiki/Equivalent_weight

Table 6. Base cations (Na^+ = sodium, K^+ = potassium, Ca^{2+} = calcium, Mg^{2+} = magnesium) and anion (Cl^- = chloride, Alk = alkalinity anions of carbonate and bicarbonate) measured at North Bay (NB), Center Lake (CL), and South Bay (SB) at Bantam Lake on three dates in 2024 reported in meq/L.

Site	Date	Na^+	K^+	Ca^{2+}	Mg^{2+}	Cl^-	Alk
		meq/L					
NB	9-Apr-24	0.55	0.03	0.41	0.31	0.60	0.62
	15-Jul-24	0.56	0.03	0.52	0.37	0.59	0.76
	21-Oct-24	0.56	0.04	0.54	0.38	0.64	0.82
CL	9-Apr-24	0.53	0.03	0.40	0.30	0.53	0.58
	15-Jul-24	0.57	0.03	0.51	0.36	0.59	0.72
	21-Oct-24	0.54	0.03	0.52	0.37	0.63	0.78
SB	9-Apr-24	0.52	0.03	0.40	0.28	0.54	0.56
	15-Jul-24	0.55	0.03	0.48	0.35	0.58	0.70
	21-Oct-24	0.55	0.04	0.53	0.38	0.64	0.84
	Lake Avg.	0.55	0.03	0.48	0.34	0.59	0.71

Sodium concentrations were stable and exhibited the highest concentrations of the four base cations during the season on a meq/L basis. Calcium had the next highest season average concentrations and exhibited more variability. Magnesium concentrations were lower than sodium and calcium concentrations. Potassium concentrations were the lowest of the four and stable.

The concentrations of alkalinity anions were collectively the greatest of all ions measured and exhibited the greatest variability. Concentrations increased in a unidirectional manner by more than 30% at NB and CL, and by 50% at SB between April and October. The percentage increases for calcium and magnesium during the season were also high and between 25 and 35%.

VIII. Discussion

A. 2024 Trophic Assessment

Bantam Lake was best characterized as eutrophic in 2024. Average summer (July – September) Secchi disk transparency and chlorophyll-*a* concentration were indicative eutrophic conditions. The season average epilimnetic total phosphorus concentration was within the late mesotrophic range, but the August through October average was within the eutrophic range. The May, June, and July total phosphorus averages were characteristic of mesotrophic conditions. Season average total nitrogen was also characteristic of mesotrophic conditions but the epilimnetic and metalimnetic algal productivity were limited by the eutrophic-level phosphorus based on Redfield ratios.

The lake exhibited many signs of internal nutrient loading. These included elevated total phosphorus, total nitrogen, ammonia, and alkalinity in samples collected near the bottom of NB and CL sites. These and other signs, including intrusions of anoxic / highly reduced waters above the thermocline, generally became more pronounced in mid- to late summer. The highest cyanobacteria concentrations were observed after the breakdown of stratification and the mixing of the water column in early August.

The SB site did not exhibit these signs presumably because it was rarely stratified (once in April), and anoxic conditions never developed in waters overlying the bottom. The shallow depth (~4 meters) at SB and wind-driven mixing kept oxygen concentrations high enough to prevent the environment from becoming highly reduced (see *Oxidation-Reduction Potential* above).

The shift from low to high cyanobacteria productivity, i.e., mesotrophic-like to eutrophic conditions occurred after mid-July as reflected in Secchi disk transparencies (Fig. X), cyanobacteria cell counts (Fig. Y), and chlorophyll-*a* concentrations (Fig. Z). This timing corresponded with the timing of the breakdown of stratification and water column mixing which likely resulted in the enrichment of the upper levels of the water column with nutrients. Even though internal loading was not an important factor at SB, wind driven horizontal mixing resulted in shifts at the site to higher chlorophyll and lower Secchi transparencies consistent with those at NB and CL.

B. Historical Change

Table 7. Average water quality characteristics in the epilimnion of Bantam Lake in the 1930s (Deevey 1940), 1970s (Frink and Norvell 1984), early 1990s (Canavan & Siver 1994, 1995), and 2018 through 2024.

Parameter	Units	Bantam Lake									
		1930s	1970s	1990s	2018	2019	2020	2021	2022	2023	**2024**
Total Nitrogen	µg/L		774	714	550	276	469	459	---	693	**464**
Total Phosphorus	µg/L	18.0	33.2	42	25.7	22.7	22.8	23.6	13.1	19.9	**27.0**
Chlorophyll-*a*	µg/L	14.7	31.3	19.7	9.1	9.8	8.5	8.0	11.8	9.1	**16.7**
Secchi Disk	meters	2.3	2.1	1.7	2.63	2.66	2.10	2.27	2.37	1.90	**2.28**
pH	SU	---	---	7.8	7.8	8.0	8.4	8.0	8.0	8.0	**7.9**
Sp. Conductivity	µS/cm	---	96	122	192	176	187	192	194	188	**154**
Alkalinity	mg/L	28.5	25.0	30.5	40.7	39.4	39.7	42.6	41.3	41.3	**36**
Chloride	mg/L	---	10.0	10.3	35.6	27.9	28.5	27.4	33.2	32.9	**20.4**
Calcium	mg/L	---	9.0	8.2	11.9	10.5	11.6	11.7	11.0	11.0	**9.6**
Magnesium	mg/L	---	3.9	7.8	4.8	4.2	5.1	4.9	4.9	4.8	**4.2**
Sodium	mg/L	---	6.2	7.4	17.2	15.5	15.8	14.7	17. 0	15.2	**12.6**
Potassium	mg/L	---	1.2	1.2	1.9	1.5	1.2	ND	1.3	0.9	**1.3**

Table 7 provides historical water quality data collected from Bantam Lake over the last 90 years, including that collected in the last seven years. Average total nitrogen levels in the last seven years were lower than the averages from the 1970's and 1990's. Average total phosphorus and chlorophyll-*a* levels in 2024 were the highest in the last seven years but were not higher than averages from the 1970's and 1990's datasets.

While average specific conductance and concentrations of ion over the last seven years were higher than 1970's and 1990's levels, the recent 2024 averages indicated a reduction in dissolved salts. Average specific conductance, most base cation levels, chloride and alkalinity levels were all at a seven-year low in 2024 (Table 7). Mild winters in recent years may have played a role if less deicing road salts were used.

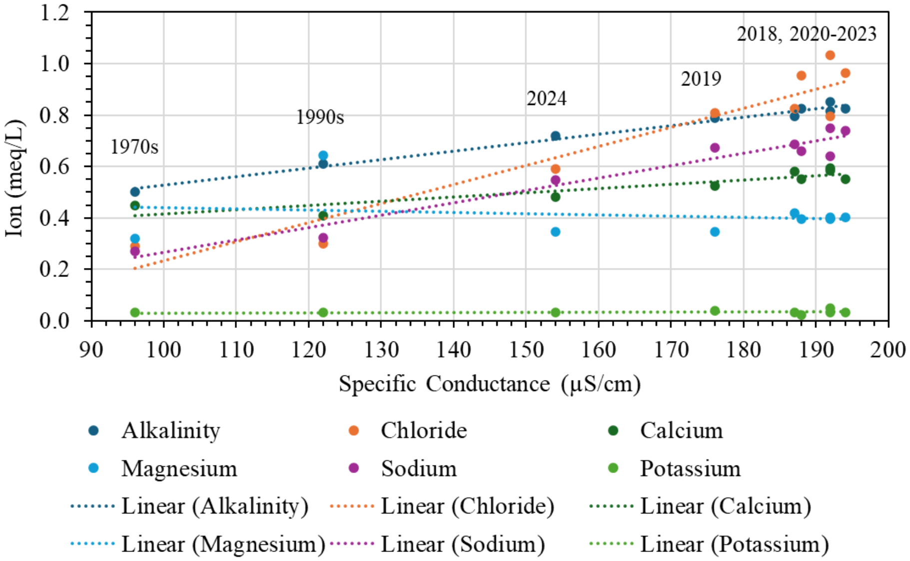

Figure 15. Regressions of average base cations (sodium, potassium, calcium, and magnesium), chloride, and alkalinity against average specific conductance in the epilimnion of Bantam Lake between 2018 and 2024.

To explore the relationship between specific conductance and ion concentrations, the annual averages of the base cations, chloride and alkalinity between 2018 and 2024 were regressed against corresponding averages of specific conductance and displayed (Fig. 15). Coefficients of determination (R^2), which characterize how well a statistical model (i.e., regression line) fits data and predicts conditions, were determined for each regression (Table 8). R^2 values can range from 0 (points do not fit the linear model well) to 1 (points fit the linear model perfectly). Additionally, p values for each regression were determined. The p value is a statistical calculation that predicts how likely the relationship between the two variables (e.g., specific conductance and sodium concentration) does or does not exist. The <u>smaller</u> the p-value, the more likely the relationship between the two variables exists. The threshold for significance is 0.05, meaning that the relationship between two variables with p values lower than 0.05 are statistically significant, i.e., the relationship does exist. For p values >0.05, the relationship is not significant.

Table 8. Coefficients of Determination (R^2) and p-values for regressions of specific conductance against alkalinity, sodium, chloride, calcium, potassium, and magnesium.

Variable	R^2	p-value
Alkalinity	0.98	2.042E-07
Sodium	0.95	0.000009681
Chloride	0.91	0.00006742
Calcium	0.81	0.0009191
Potassium	0.05	0.5458
Magnesium	0.03	0.6507

Based upon those analyses, potassium and magnesium appear to have very little impact on specific conductance levels at Bantam Lake. The relationships of specific conductance with alkalinity, sodium, chloride, and calcium are highly significant. This implies that while the use of sodium chloride-based deicing road salts may have increased specific conductance, there are additional factors contributing to the increase. The specific conductance / alkalinity regression had the highest R^2 value, had the lowest p-value, and is also a variable that can be generated internally from the biologically mediated reduction of iron, manganese, and sulfate via cellular respiration, and denitrification of nitrate to elemental nitrogen in the anoxic lake sediments (Wetzel 2001). This implies that some of the increase in specific conductance over time may have been driven from within the lake and not from the watershed.

C. Change Since 2018

One of the goals of the annual lake monitoring program was to develop a database which could be used to detect changes in water quality. A robust database provides the ability to detect statistically significant trends that may be occurring. Below, two statistical approaches were used to assess whether the lake and specific variables were trending, i.e., significantly increasing or decreasing, or not significantly changing since 2018 (see Appendix C). The first approach pooled 2018 – 2024 annual epilimnetic and hypolimnetic data into one dataset; the second approach leveraged the annual epilimnetic and hypolimnetic datasets independently. The result was three independent datasets.

Two statistical methods were applied to each of the three datasets. The first method, *Multiple Linear Regression* (MLR), was employed to determine if the epilimnion, hypolimnion, and/or the whole lake (pooled data) had changed significantly based on the multiple variables in each dataset. A p-value statistic was calculated to determine whether the epilimnion, hypolimnion, or whole lake was experiencing change or not with $p<0.05$ indicating a statistically significant change. Variables used were epilimnetic and hypolimnetic total phosphorus, total nitrogen, specific conductance, alkalinity, and pH. Additionally, Secchi disk transparencies and chlorophyll concentrations were included in the epilimnetic dataset.

The second method, *Analysis of Variance* (ANOVA), was utilized to examine those variables independently to determine whether a change occurred in a statistically significant manner over time. The F-statistic was used to calculate the probability (i.e., p-value) that a dataset's variable pattern differed from a random distribution of values, i.e. exhibited a statistically significant change.

Results from MLR analyses indicated that there had been changed in the lake over the seven-year period based on the pooled data and epilimnetic data. Variables that exhibited the greatest influence on the combined data MLR model were total phosphorus, specific conductance, and alkalinity. Specific conductance was the primary variable driving the change in the epilimnetic MLR model.

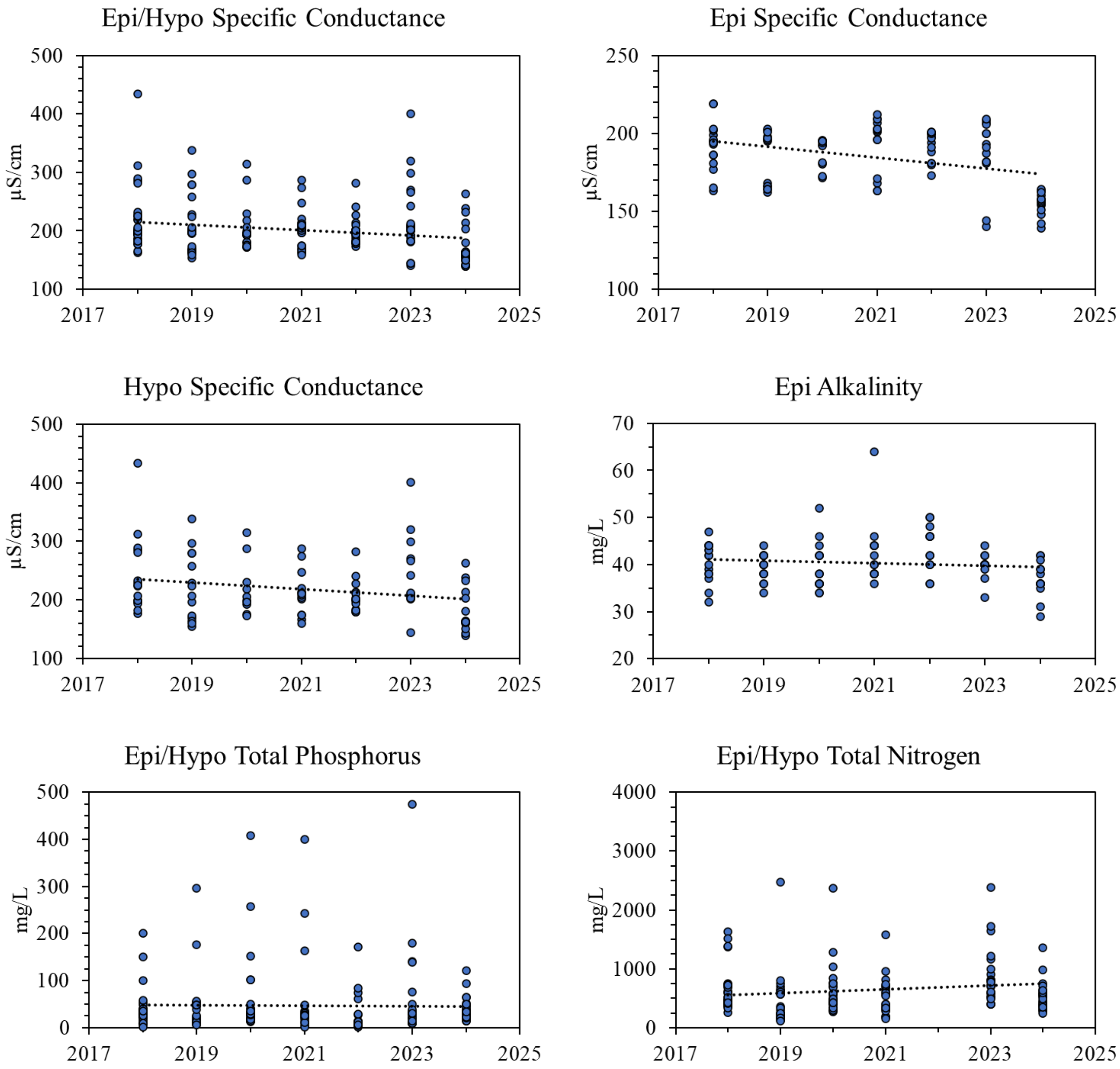

Figure 16. Linear regressions of selected data collected from 2018 to 2024. Datasets include combined epilimnetic and hypolimnetic (Epi/Hypo) specific conductance, epilimnetic (Epi) specific conductance, hypolimnetic (Hypo) specific conductance, epilimnetic alkalinity, combined epilimnetic and hypolimnetic total phosphorus, and combined epilimnetic and hypolimnetic total nitrogen.

Results from ANOVA indicated that all three specific conductance datasets (epilimnetic, hypolimnetic, and combined) have significantly decreased since 2018 ($p<0.05$; Fig. 16). The change in the combined and epilimnetic datasets were highly significant ($p<0.001$). Combined epilimnetic and hypolimnetic alkalinity also decreased in a significant manner ($p<0.05$). Season averages of the other variables, including total phosphorus, total nitrogen, chlorophyll, and Secchi transparency, have not changed in a statistically significant manner since 2018.

D. Cyanobacteria Growth and 2024 Peroxide-Based Treatment

Several approaches were used to assess whether the hydrogen peroxide-based algicide used in 2024 resulted in conditions that differed from those in seasons when copper sulfide was used (e.g. 2018 to 2022) or when no algicides was used (e.g. 2023) at Bantam Lake. First, data that characterized or was a surrogate for cyanobacteria growth from 2020 to 2024 was compiled (Appendix C) and displayed on an annual basis (Fig. 17). Only dates when data from all four variables existed were used. Season averages for chlorophyll-*a*, Secchi disk transparency, cyanobacteria cell concentration, and relative phycocyanin concentration were calculated (Table 9). ANOVA and Kruskal Wallis statistical analyses were performed for each variable. <u>No statistically significant differences were observed between years for each variable</u>.

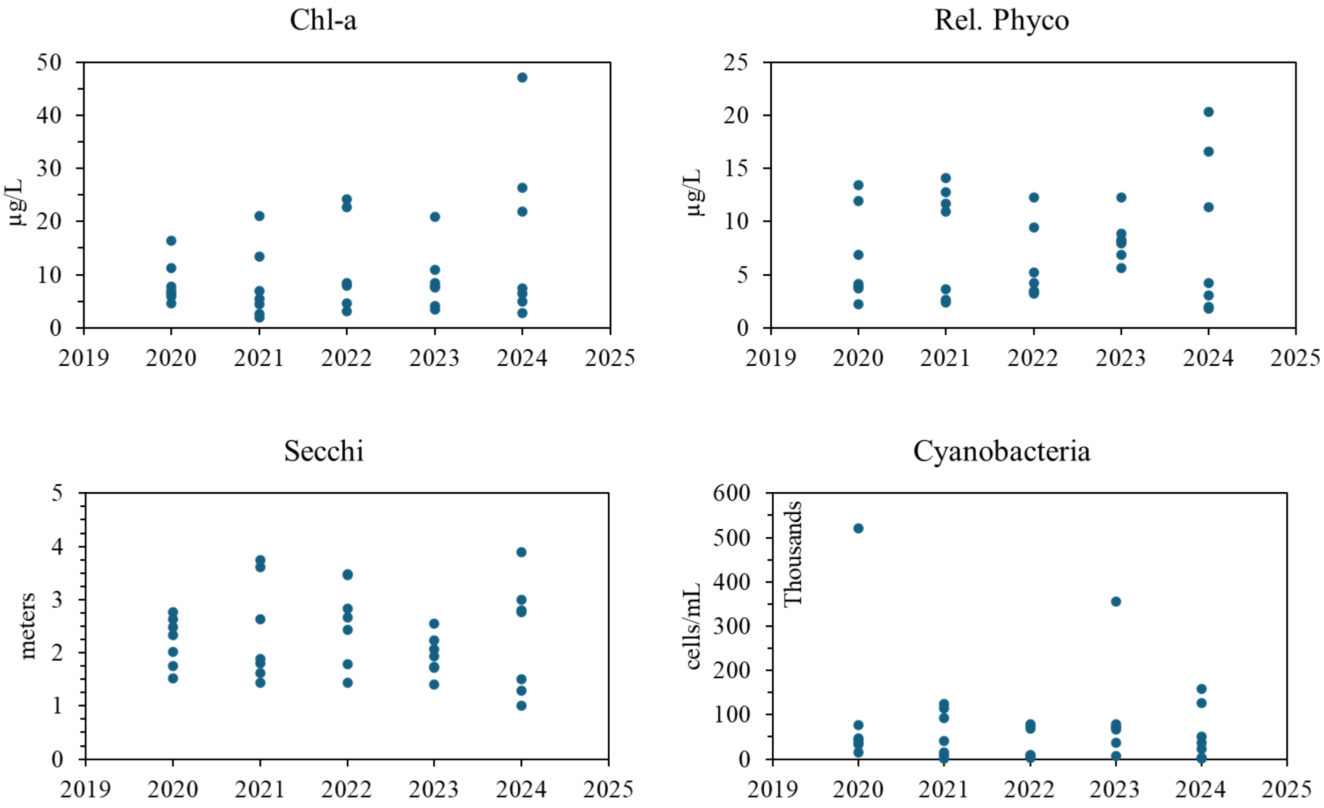

Figure 17. Monthly measurements of chlorophyll-a (Chl-a), relative phycocyanin, (Rel. Phyco) Secchi disk transparency (Secchi), and cyanobacteria cell concentration (Cyanobacteria) for the years 2020 to 2024. Each point represents the lake average for a given date.

Table 9. Season averages for chlorophyll-a (Chl-a), Secchi disk transparency (Secchi), cyanobacteria cell concentration (Cyano) and relative phycocyanin (Rel. Phyco) from 2020 to 2024.

Year	Chl-a (µg/L)	Secchi (meters)	Cyano (cells/mL)	Rel. Phyco (µg/L)
2020	8.49	2.22	110402	6.59
2021	7.97	2.39	56709	8.32
2022	11.83	2.59	35507	5.89
2023	9.05	1.95	98656	8.28
2024	16.69	2.32	57503	8.47

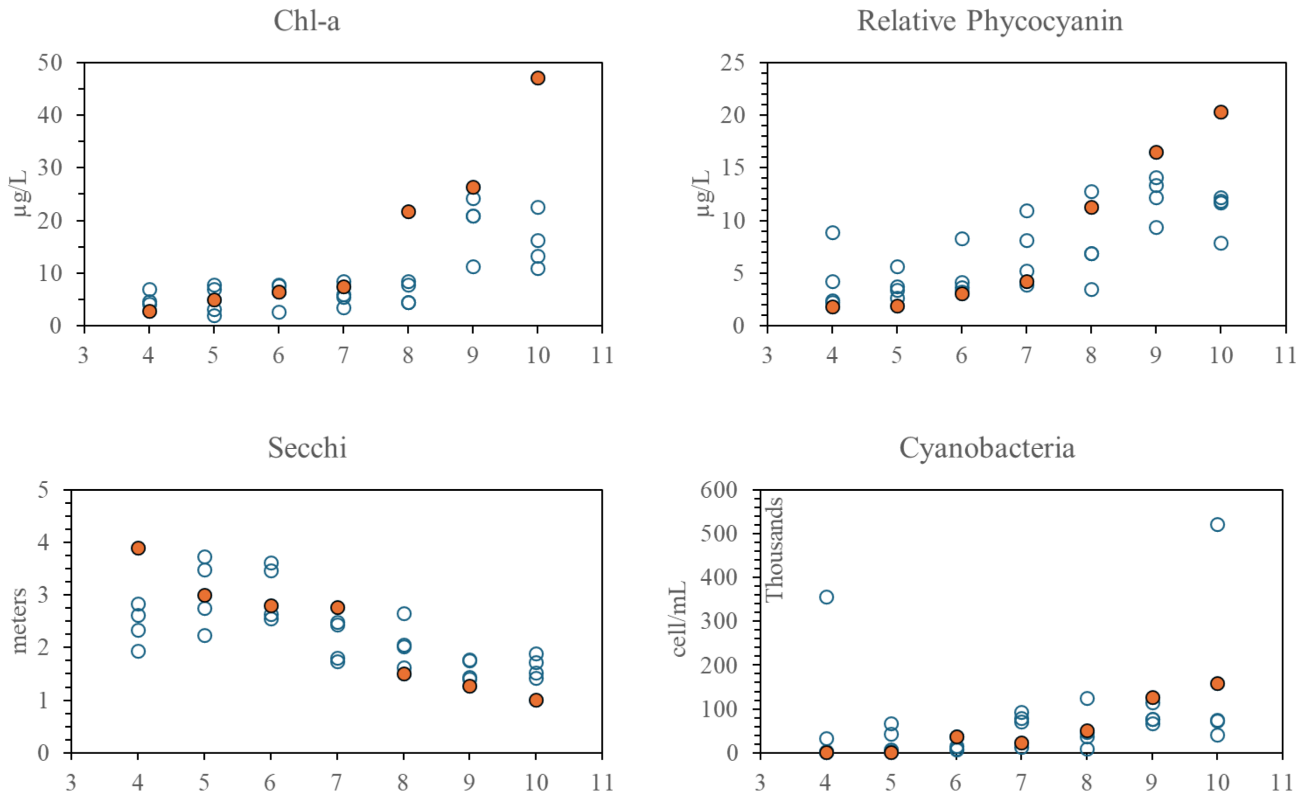

Figure 18. Figure 19. Comparisons of monthly chlorophyll-a concentrations (Chl-a), relative phycocyanin concentrations, Secchi disk transparency, and cyanobacteria cell concentration monthly from 2020 to 2024. Red points are the 2024 data. The hollow points are data from 2020 to 2023. Months are represented numerically, i.e., 4 = April, 5 = May, 6 = June, 7 = July, 8 = August, 9 = September, and 10 = October.

The same dataset was sorted by month and displayed (Fig. 18). Visual assessment suggested that April through July relative phycocyanin and cyanobacteria cell concentrations in 2024 were low compared to April of other years. April of 2024 chlorophyll-*a* concentration and Secchi disk transparency indicated that the cyanobacteria levels for that month were the lowest in the last five years. However, August through October data in 2024 portrayed some of the poorest conditions over the five-year period, e.g., highest chlorophyll concentrations and lowest Secchi transparencies since 2020.

A second approach used to assess differences in cyanobacteria productivity over the last five years utilized modified CyanoMonitoring protocols (CMC 2024). CyanoMonitoring is an ongoing, nationwide program to collect data on cyanobacteria populations across the country and to better understand the increases in bloom frequency and intensity. It utilizes lake scientists and lake managers, as well as citizen scientists, to collect samples and analyze them with fluorimetry (CMC 2021), like that discussed in earlier sections of this report.

CyanoMonitoring utilizes the following formula to calculate cyanobacteria growth rates between two dates:

$$\mu\ d^{-1} = \ln(F_2/F_1)/(t_2-t_1)$$

where $\mu\ d^{-1}$ is growth in micrograms phycocyanin per day; ln is the natural logarithm; F_2 is the fluorometrically measured mass of phycocyanin at day 2 (t_2); and F_1 is the fluorometrically measured mass of phycocyanin at day 1

(t_1). Whereas CyanoMonitoring utilizes in-lab fluorimetry to analyze phycocyanin in different cyanobacteria size fractions in the same sample, (e.g., large bloom-forming genera vs small picoplankton genera), we assessed relative phycocyanin growth utilizing in-lake fluorimetry in the top three meters of the water column at the CL site using the biweekly measurements. We plotted phycocyanin levels, the natural log of those, and the calculated growth rates across the season for each year from 2020 to 2024. Plots for all years are provided in Appendix E. Below, we provided plots for the last three years: in 2022 when copper sulfate was used on July 17th and August 30th; in 2023 when no treatments occurred; and in 2024 when hydrogen peroxide was used on April 10th (Fig. 20).

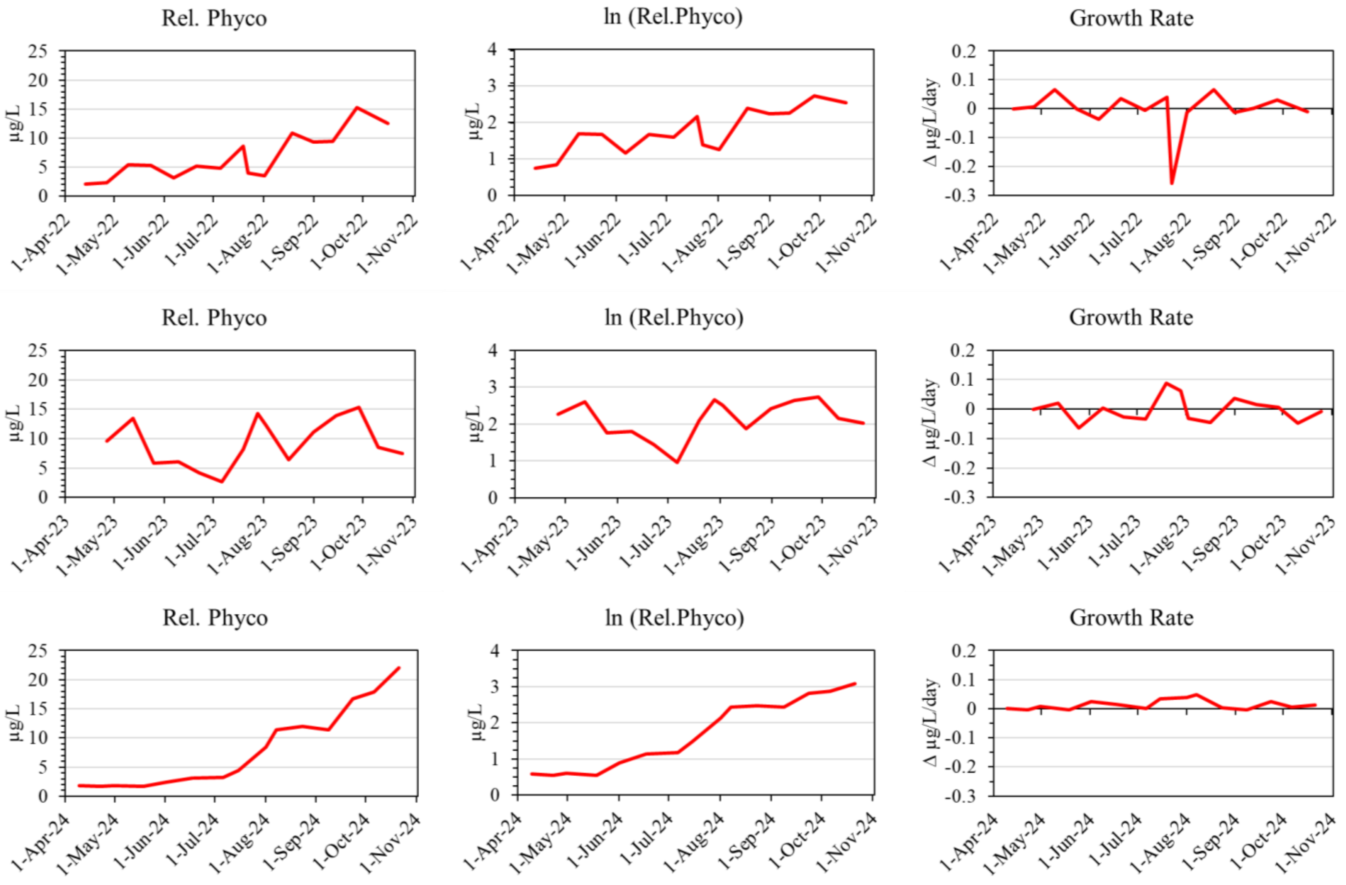

Figure 20. Time-series plots of relative phycocyanin concentrations (Rel. Phyco), the natural log of the relative phycocyanin concentrations (ln Rel. Phyco), and cyanobacteria growth rates for 2022 (top), 2023 (middle), and 2024 (bottom).

Although not identical, 2022 plots were similar to the 2020 and 2021 plots in that there was variability in relative concentrations but with levels trending up over the season. Growth rates varied up and down with notable decreases following the first copper sulfate treatments, followed by notable increases in growth in August. Shifts in phycocyanin concentrations in 2023, when no treatment occurred, were greater than those in 2022 and 2024 and concentrations early in the season were high. The 2024 plots differed from the 2023 and 2022 plots since phycocyanin levels, although trending upwards, exhibited much less variability over time and remained low through mid-July. Growth rates in 2024 were more subtle (Fig. 20).

It is important to note that cyanobacteria concentrations are influenced by many factors and not just whether a treatment was implemented or not, or what type of algicide was used. Nutrient levels and seasonal climate are very important variables impacting cyanobacteria concentrations and bloom formations and those can greatly vary from year to year. Signs of the hydrogen peroxide treatment being effective were observed, but those signs did not persist for the entirety of the season and season averages for the variables used to assess cyanobacteria productivity were not significantly different over the years of 2020 to 2024. Therefore, the continued use of the hydrogen peroxide-based algicide, with its much higher cost, does not currently appear warranted. We will continue to monitor research efforts on this alternative method of cyanobacteria bloom mitigation in scientific and lake management literature.

E. 2023 and 2024 Conditions vs TMDL Goals

The Bantam appendix of the State's TMDL provided in-lake nutrient goals and projected what chlorophyll-*a* concentrations and Secchi disk depths should be once attaining those nutrient goals. Annual 2023 and 2024 averages for those variables were compiled along with the TMDL goals and projections (Table 9).

The 2023 and 2024 data were inconsistent relative to TMDL goals and projections. The 2023 average total phosphorus was nearly at the TMDL goal while the 2024 average was higher. The reverse was true for total nitrogen. Chlorophyll-*a* concentrations in 2023 were on average lower than the TMDL projections, but the 2024 average was higher. Average Secchi disk transparency in 2023 was lower than the projection but the 2024 average was higher. It is worth noting that there was a change to the UCONN CESE laboratory in August of 2023.

Table 10. Comparisons of average 2023 and 2024 epilimnetic total phosphorus, total nitrogen, and chlorophyll-*a* concentrations, and Secchi disk transparencies to the goals and projections of the Bantam TMDL (CT DEEP 202X).

Variable	Units	2023 Avg	2024 Avg	TMDL Goals or Projections
Total Phosphorus	µg/L	19.9	27.0	20
Total Nitrogen	µg/L	693	464	400
Chlorophyll-*a*	µg/L	9.1	16.7	11.8
Secchi Transparency	meters	1.90	2.28	2.1

The combined 2023-2024 averages were also determined and compared to the TMDL goals and projections. The 2-year averages for total phosphorus, total nitrogen, and chlorophyll-a concentrations, and Secchi disk transparency were 23.5 µg/L, 648.2 µg/L, 12.9 µg/L, and 2.14 meters, respectively. Recent nutrient levels were higher than TMDL goals, average chlorophyll was slightly above the TMDL projection, and Secchi disk transparency was at the TMDL projected level.

Within the TMDL, the current phosphorus loading to the lake was estimated at 1,614 kg/yr with approximately 35% of that generated internally from within the lake. A target phosphorus loading rate was established at 1,211 kg/yr.

For comparative purposes, the phosphorus loads in the lake were estimated based on the concentrations on the 2023 and 2024 sampling dates. Phosphorus concentrations measured throughout the water column were applied to the estimated volumes of the corresponding layer (e.g. epilimnion, metalimnion, and/or hypolimnion) which were determined using temperature, RTRM calculations, and a hypsographic curve for Bantam Lake. The total was determined by adding the phosphorus mass in each layer for a specific date. It is important to note that in-lake phosphorus loads are rough estimates since concentrations are only determined at specific levels of the water column. For example, the phosphorus concentration near the bottom might be applied to the entire hypolimnion which may extend up several meters. The assumption being made is that the concentration is consistent throughout the layer.

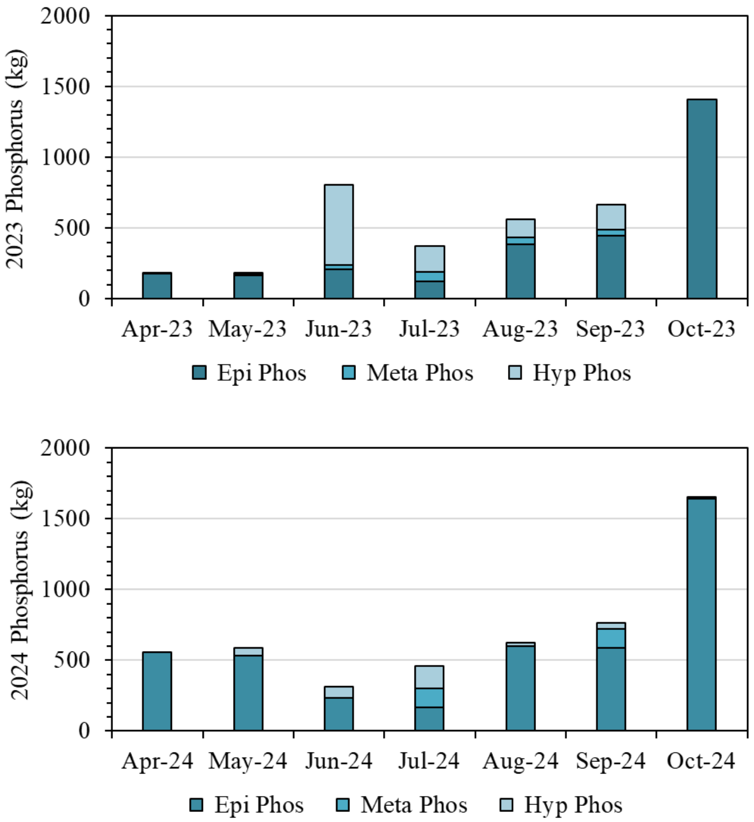

Figure 21. Phosphorus mass (load) in the water column of Bantam Lake based on data from Center Lake in 2023 and 2024.

In 2023 and 2024, the total phosphorus loads in the lake at the end of each season were 1,410 kg and 1,656 kg, respectively. In both years, the total load increased starting in the mid-summer and peaked by October when most, if not all, of the water column is mixed. This timing was concurrent with nutrient loading from the lake sediments. Late spring / early summer total loads, when the lake is just starting to stratify, were much lower (Fig.

IX. Conclusions and Recommendations

A. 2024 Conditions

On average, Bantam Lake in 2024 was best characterized as eutrophic, but that characterization was greatly influenced by the August through October data. April through July data depicted a mesotrophic lake with some of the lowest cyanobacteria cell counts and relative phycocyanin concentrations for those months in the last five years. We hypothesize that the hydrogen peroxide-based algicide was effective for the part of the season.

Algal productivity became eutrophic-like once stratification broke down and the water column mixed across the waterbody, which occurred between mid-July and early August. At that time, high concentrations of phosphorus sequestered below the thermocline were released throughout the water column. Combined with warmer temperatures, the cyanobacteria that survived the April treatment and those that germinated from the sediments after the treatment, reached their exponential growth phase creating the high concentrations observed from August through October.

The decrease in specific conductance and the ions that are highly correlated with it in 2024 was a good indication that the watershed plays a role in Bantam's chemistry, including nutrient chemistry. In this case, we surmise that milder winters in recent years reduced the salt load in the watershed which has been reflected in concentrations and specific conductance in the lake. Additionally, specific conductance, in 2023 decreased between early July through the end of October and likely reflected the accelerated lake flushing rate due to record rainfall in the area. This undoubtedly contributed to the lower conductivity and ion concentrations in the beginning of the 2024 season. Climate will continue to play a role, perhaps a larger role, in the water quality of Bantam Lake in the future.

B. Cyanobacteria Management

We believe that hydrogen peroxide-based treatments can be an effective tool at managing cyanobacteria blooms but is currently cost-prohibitive. Additionally, more in-lake research is needed. We recommend returning to the copper sulfate treatments for now. Based on our five-year assessment of cyanobacteria productivity and treatment, we believe the first treatment should occur in mid-July like it did in 2022. We also recommend sharing portions of this report focusing on cyanobacteria management with the algicide applicator, SOLitude, and meeting with them this spring to discuss treatment options.

C. TMDL Goals

The 2024 average total phosphorus and total nitrogen concentrations and the 2023/2024 averages did not meet the TMDL goals. The same was true for average Secchi transparency and chlorophyll-a concentration. However, none of the mitigation measures in the Watershed-based Plan for Bantam Lake have been implemented, although the White Memorial Foundation has received grant funding for several of those projects proposed in the plan and may have started planning for those. Better communications with the watershed partnering entities are recommended. Grant funding for several more projects has been applied for.

Lake and watershed monitoring will continue to be an important component of the Bantam Lake management. However, all the costs of monitoring, lake and watershed, fall on the BLPA. We recommend engaging several professional grant writing companies to assist with the development of grants applications for monitoring and future non-point source projects in the Bantam Watershed Based Plan.

D. Sediment Phosphorus Sequestering – Alum

The important role of internal phosphorus loading in cyanobacteria dynamics at Bantam Lake is irrefutable. Multiple lines of evidence have been provided in this and past reports. The BLPA has invested considerable resources in understanding sediment phosphorus content and alum dosing rates for Bantam Lake. The reason for not moving forward with an alum treatment is the cost that was projected to be 1.8 million dollars across several years. We recommend developing a strategy to acquire the necessary funds for the project. This effort might be aided with assistance from a professional grant writing company and other financial professionals.

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Appendix A. Nutrient and Chemical Data in 2024

NH4 = ammonia; TN = total nitrogen; TN:TP is the Redfield ratio of total nitrogen to total phosphorus; TP = total phosphorus; Alk – alkalinity, Chl = chlorophyll-a; Epi = epilimnion; Meta = metalimnion; Hypo = hypolimnion; Na = sodium; K = potassium; Ca = calcium; Mg = magnesium; Cl = chloride.

North Bay

Site 1 (NB)	NH_4	NH_4	NH_4	TN	TN	TN	TN:TP	TN:TP	TN:TP
	Epi	Meta	Hypo	Epi	Meta	Hypo	Epi	Meta	Hypo
	µg/L								
9-Apr-24	7	5	5	331	278	242	10	10	12
18-May-24	17	25	38	292	340	387	19	26	11
17-Jun-24	0	0	32	380	357	354	15	19	8
15-Jul-24	0	11	153	410	470	555	19	13	9
23-Aug-24	8	12	21	574	551	487	18	21	21
23-Sep-24	19	26	45	586	602	592	16	18	17
21-Oct-24	4	4	114	619	699	638	21	19	13

Site 1	TP	TP	TP	Alk	Alk	Alk	Chl	pH	pH	pH
	Epi	Meta	Hypo	Epi	Meta	Hypo	Epi	Epi	Meta	Hypo
	µg/L			mg/L			µg/L	Standard Units		
9-Apr-24	32	27	21	31	30	31	1.9	7.62	7.48	7.43
18-May-24	15	13	34	35	34	35	4.1	7.51	7.43	6.87
17-Jun-24	26	19	45	36	36	41	6.2	7.67	6.67	7.5
15-Jul-24	22	35	64	38	37	44	7.4	7.71	7.02	7.44
23-Aug-24	32	26	23	39	39	37	24.0	7.93	7.96	7.11
23-Sep-24	36	34	34	42	43	42	25.5	7.45	7.39	7.23
21-Oct-24	29	36	50	41	39	42	43.6	8.25	7.9	7.06

Site 1	Na	K	Ca	Mg	Cl
	Epi	Epi	Epi	Epi	Epi
	mg/L				
9-Apr-24	12.7	1.1	8.3	3.7	20.8
18-May-24					
17-Jun-24					
15-Jul-24	12.9	1.3	10.4	4.6	20.3
23-Aug-24					
23-Sep-24					
21-Oct-24	12.8	1.4	10.8	4.7	22.1

Center Lake

Site 2 (CL)	NH_4	NH_4	NH_4	TN	TN	TN	TN:TP	TN:TP	TN:TP
	Epi	Meta	Hypo	Epi	Meta	Hypo	Epi	Meta	Hypo
	µg/L								
9-Apr-24	7	5	8	445	290	287	14	11	13
18-May-24	10	34	144	397	313	524	12	24	13
17-Jun-24	0	0	90	310	298	458	22	17	7
15-Jul-24	0	0	657	359	376	1361	21	19	11
23-Aug-24	21	87	106	532	532	569	16	21	6
23-Sep-24	0	0	227	643	660	986	14	20	33
21-Oct-24	0	0	102	752	653	707	20	17	14

Site 2	TP	TP	TP	Alk	Alk	Alk	Chl	pH	pH	pH
	Epi	Meta	Hypo	Epi	Meta	Hypo	Epi	Epi	Meta	Hypo
	µg/L			mg/L			µg/L	Standard Units		
9-Apr-24	31	26	22	29	30	32	3.5	7.55	7.5	7.46
18-May-24	32	13	41	36	37	45	5.0	7.64	7.16	7.41
17-Jun-24	14	18	64	36	37	45	5.4	7.62	6.72	6.97
15-Jul-24	17	20	122	36	36	61	6.4	7.95	7.07	7.44
23-Aug-24	34	25	94	36	40	40	19.9	7.86	7.46	7.41
23-Sep-24	47	33	30	42	40	39	29.8	8.2	7.83	7.75
21-Oct-24	38	38	51	39	41	44	63.8	8.41	7.6	7.43

Site 2 (CL)	Na	K	Ca	Mg	Cl
	Epi	Epi	Epi	Epi	Epi
	mg/L				
9-Apr-24	12.1	1.2	8.1	3.6	18.2
18-May-24					
17-Jun-24					
15-Jul-24	13.0	1.2	10.2	4.4	20.2
23-Aug-24					
23-Sep-24					
21-Oct-24	12.5	1.4	10.3	4.5	21.7

South Bay

Site 4 (SB)	NH_4	NH_4	NH_4	TN	TN	TN	TN:TP	TN:TP	TN:TP
	Epi	Meta	Hypo	Epi	Meta	Hypo	Epi	Meta	Hypo
9-Apr-24	5	5	5	288	294	632	12	12	11
18-May-24	7	7	10	324	349	350	27	27	27
17-Jun-24	0	0	0	307	343	337	24	20	15
15-Jul-24	0	0	0	363	405	364	17	21	19
23-Aug-24	7	7	6	537	558	567	21	21	21
23-Sep-24	0	0	0	661	687	688	21	22	17
21-Oct-24	0	0	5	638	611	546	19	19	16

Site 4 (SB)	TP	TP	TP	Alk	Alk	Alk	Chl	pH	pH	pH
	Epi	Meta	Hypo	Epi	Meta	Hypo	Epi	Epi	Meta	Hypo
	µg/L			mg/L			µg/L	Standard Units		
9-Apr-24	25	25	59	28	29	28	2.9	7.55	7.51	7.46
18-May-24	12	13	12	33	33	33	5.7	7.6	7.56	7.52
17-Jun-24	13	17	22	36	36	37	7.8	7.62	6.62	6.87
15-Jul-24	21	19	19	35	36	37	8.4	7.97	7.42	7.04
23-Aug-24	26	26	27	32	34	38	21.5	8.41	8.43	7.71
23-Sep-24	31	31	40	39	37	40	23.9	8.33	8.28	8.06
21-Oct-24	34	33	34	42	41	42	33.8	8.04	7.85	7.08

Site 4 (SB)	Na	K	Ca	Mg	Cl
	Epi	Epi	Epi	Epi	Epi
	mg/L				
9-Apr-24	12.0	1.1	7.9	3.5	18.5
18-May-24					
17-Jun-24					
15-Jul-24	12.6	1.3	9.7	4.3	20.0
23-Aug-24					
23-Sep-24					
21-Oct-24	12.6	1.4	10.7	4.6	22.0

Appendix B. Algae Genera by Taxonomic Group Observed in Bantam Lake in 2024

Cyanobacteria

Aphanizomenon sp.
Aphanocapsa sp.
Aphanothece sp.
Chroococcus sp.
Dolichospermum sp.
Gomphosphaeria
Microcystis sp.
Planktothrix sp.
Pseudanabaena sp.
Snowella sp.
Woronichinia sp.

Bacillariophyta

Asterionella sp.
Aulocoseria sp.
Cyclotella sp.
Fragilaria sp.
Rhizosolenia sp.
Stephanodiscus sp.
Synedra sp.
Tabellaria sp.

Chlorophyta

Anikistrodesmus sp.
Arthrodesmus sp.
Chlamydomonas sp.
Closterium
Coelastrum sp.
Cosmarium sp.
Crucigenia sp.
Dictyosphaerium sp.
Elakatothrix sp.
Eudorina sp.
Gloeocystis sp.
Golenkinia sp.
Gonium sp.
Kirchneriella sp.
Micractinium sp.
Mougeotia sp.
Nephrocytium sp.
Oocystis sp.
Padorina sp.
Pediastrum sp.
Quadrigula sp.
Scenedesmus sp.
Selenastrum sp.
Sphaerocystis sp.
Staurastrum sp.
Tetraedron sp.
Ulothrix sp.
Volvox sp.

Pyrrhophyta

Ceratium sp.
Glenodinium sp.
Gymnodinium sp
Peridinium sp.

Cryptophyta

Cryptomonas sp.

Euglenophyta

Euglena sp.
Phacus sp.
Trachelomonas sp.

Chrysophyta

Chrysosphaerella sp.
Dinobryon sp.
Mallomonas sp.
Synura sp.
Uroglenopsis sp.

Appendix C. Multiple Linear Regression (MLR) and Analysis of Variance (ANOVA)

Whole Lake

MLR

	Estimate	Std. Error	t value	Pr (>|t|)	
Intercept	2.03E+03	3.27E+00	619.223	<2e-16	***
TP	8.87E-03	3.97E-03	2.233	0.0272	*
TN	5.35E-04	2.73E-04	1.961	0.0518	.
SpCond	-1.04E-02	4.89E-03	-2.124	0.0355	*
Alk	-8.53E-02	3.87E-02	-2.207	0.029	*
pH	-8.32E-02	3.74E-01	-0.222	0.8243	
SE	2.13				
R	0.12				
F	3.72				

EPI

MLR

	Estimate	Std. Error	t value	Pr (>|t|)	
Intercept	2.03E+03	5.90E+00	344.263	< 2e-16	***
Secchi	-4.22E-01	4.71E-01	-0.895	0.37433	
Chloro	-1.68E-03	3.27E-02	-0.051	0.9592	
TP	-2.53E-02	2.58E-02	-0.982	0.32993	
TN	1.17E-03	1.23E-03	0.953	0.34399	
SpCond	-4.21E-02	1.58E-02	-2.658	0.00996	**
Alk	-8.68E-02	6.81E-02	-1.274	0.20727	
pH	2.10E-01	6.57E-01	0.32	0.7498	
SE	2.0280				
R	0.2637				
F	3.2230				

HYPO

MLR

	Estimate	Std. Error	t value	Pr (>|t|)	
Intercept	2.02E+03	7.83E+00	258.168	<2e-16	***
TP	6.06E-03	4.88E-03	1.24	0.219	
TN	4.67E-04	2.92E-04	1.599	0.114	
SpCond	-8.38E-03	5.74E-03	-1.459	0.149	
Alk	-6.44E-02	5.19E-02	-1.241	0.219	
pH	6.43E-01	1.01E+00	0.638	0.525	
SE	2.1830				
R	0.1092				
F	1.6880				

Whole Lake

ANOVA	Df	Sum Sq	Mean Sq	F value	Pr (>F)	
TP	1	0	0	0	0.9949986	
TN	1	7.371	7.371	1.6231	0.2047897	
SpCond	1	54.847	54.847	12.0764	0.0006815	***
Alk	1	22.023	22.023	4.8492	0.0293079	*
pH	1	0.225	0.225	0.0495	0.8242814	

Epi

ANOVA	Df	Sum Sq	Mean Sq	F value	Pr (>F)	
Secchi	1	7.416	7.416	1.803	0.1841721	
Chloro	1	4.712	4.712	1.1455	0.2885679	
TP	1	3.553	3.553	0.8639	0.3561978	
TN	1	0.28	0.28	0.068	0.7950777	
SpCond	1	69.297	69.297	16.8473	0.0001186	***
Alk	1	7.129	7.129	1.7331	0.1927822	
pH	1	0.422	0.422	0.1026	0.7498017	

Hypo

ANOVA	Df	Sum Sq	Mean Sq	F value	Pr (>F)	
TP	1	0.0014	0.0014	0.0003	0.98642	
TN	1	5.6321	5.6321	11819	0.28082	
SpCond	1	22.6229	22.6229	4.7473	0.03281	*
Alk	1	9.5348	9.5348	2.0008	0.16178	
pH	1	1.9424	1.9424	0.4076	0.52533	

Appendix D. Lake Average Chlorophyll-a, Secchi disk transparency data, cyanobacteria cell concentrations and relative phycocyanin from 2020 to 2024

Year	Date	Chl-a µg/L	Secchi meters	Cyano cells/mL	Rel. Phyco µg/L
2020	23-Apr-20	4.64	2.33	32721	2.23
2020	21-May-20	6.97	2.76	43799	3.72
2020	17-Jun-20	6.42	2.64	37013	4.16
2020	13-Jul-20	5.93	2.49	14208	3.90
2020	11-Aug-20	7.79	2.02	47358	6.86
2020	9-Sep-20	11.34	1.76	76596	13.39
2020	19-Oct-20	16.36	1.53	521117	11.90
2021	27-Apr-21	6.93	2.63	915	2.43
2021	25-May-21	2.01	3.74	8319	2.68
2021	23-Jun-21	2.63	3.61	15543	3.65
2021	19-Jul-21	5.47	1.81	92391	10.95
2021	16-Aug-21	4.44	1.62	124277	12.76
2021	13-Sep-21	20.98	1.44	114439	14.08
2021	12-Oct-21	13.34	1.88	41079	11.70
2022	13-Apr-22		2.83	3980	4.20
2022	9-May-22	3.20	3.49	2980	3.43
2022	6-Jun-22	7.89	3.47	9545	3.26
2022	5-Jul-22	8.48	2.44	78557	5.20
2022	1-Aug-22	4.56	2.66	9765	3.48
2022	12-Sep-22	24.22	1.78	68337	9.41
2022	16-Oct-22	22.66	1.43	75383	12.24
2023	26-Apr-23	4.11	1.93	356539	8.88
2023	25-May-23	7.80	2.23	67778	5.61
2023	22-Jun-23	7.68	2.56	6770	8.31
2023	19-Jul-23	3.48	1.74	71226	8.11
2023	18-Aug-23	8.40	2.06	37663	6.91
2023	14-Sep-23	20.93	1.41	77955	12.24
2023	10-Oct-23	10.97	1.72	72659	7.92
2024	9-Apr-24	2.77	3.90	2599	1.82
2024	18-May-24	4.93	3.00	2008	1.96
2024	17-Jun-24	6.47	2.80	36536	3.10
2024	15-Jul-24	7.40	2.77	23692	4.21
2024	23-Aug-24	21.80	1.51	50881	11.33
2024	23-Sep-24	26.40	1.28	127469	16.55
2024	21-Oct-24	47.07	1.01	159333	20.35

Appendix E. Time series plots of relative phycocyanin concentrations, their natural log, and growth rates at the Center Lake Site from 2020 to 2024

Appendix F. Preparers' Qualifications

Laurence J. Marsicano

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RELEVANT EXPERIENCE

- Thirty years as a lake ecologist, manager, advocate, educator, and leader in Connecticut. Successful in the academic, public, and private sectors.
- Advanced the mission of the Candlewood Lake Authority from 1998 through 2017 with the last 14 of those as Executive Director. The board and staff of that agency served the five municipalities surrounding Candlewood Lake, the largest lake in the State and one of Connecticut's most important inland water resources.
- Developed meaningful relationships and worked with the general CT lake community, local and state environmental agency staff, academic researchers, elected leaders at all levels of government, and educators from middle school through college/university levels.
- Co-directed an interdistrict grant program that utilized Candlewood Lake as a living, learning laboratory. The program ran for 10+ years and engaged ~150 high school students and teachers each year.
- Have trained and supervised employees and/or students in Limnological and Paleolimnological field and laboratory methods.
- Founding member of the Connecticut Federation of Lakes, have, and served as a volunteer and an officer of Connecticut's lake advocacy, nonprofit organization until 2022.

PROFESSIONAL EXPERIENCE

- **Principal Limnologist – Brawley Consulting Group, LLC.** 2023 to present
- **Principal Partner – Aquatic Ecosystem Research, LLC.** July 2017 to 2022
- **Adjunct Faculty** – Western Connecticut State University, Biol. and Enviro. Science Dept. August 2011 to present.
- **Executive Director –** Candlewood Lake Authority, Sherman, CT 06784. April 2003 to July 2017
- **Lake Preservation Director –** Candlewood Lake Authority, Sherman, CT 06784. April 1998 to Oct. 2002
- **Academic Research Associate –** Connecticut College, New London, CT 06320. Sept. 1989 to Jan. 1998
- **Visiting Lecturer –** Connecticut College, New London, CT 06320. August 1997 to January 1998
- **Research Assistant** – Western Connecticut State University, Danbury, CT 06810. 1987 to 1989

CERTIFICATION, EDUCATION, AND TRAINING

- **Certified Lake Manager**, North American Lake Management Society, 2017
- **Professional Certification** in GIS, Pace University, 2014
- **Graduate Certification** in GIS Technology, University of New Haven 2001
- **M.A. in Botany**, Connecticut College 1993
- **B.A. in Biology**, Western Connecticut State University 1988

AWARDS

- **Excellence in Environmental Stewardship** from the **Connecticut Outdoor and Environmental Education Association** in 2018
- **Recognition of Service** in the **Congressional Record** by **US Rep. Elizabeth Esty** on June 14, 2017
- **Watershed Conservationist Award** from the **Housatonic Valley Association** in 2011
- **Conservation Professional of the Year** from the **Litchfield County Conservation District** in 2002
- **Honor Award**, **Southern New England Chapter of the Soil and Water Conservation Society** in 2000.
- **Green Circle Award** from the **Connecticut Department of Environmental Protection** in 1999.
- **Conservation Award** from **Housatonic Valley Association** for publication entitled *Candlewood Lake: Watershed Awareness and Lake Preservation* in 1998.

ORGANIZATIONS

- **Connecticut Federation of Lakes** – Founding member 1995; Treasurer from 1995 – 2001; Vice President from 2009 – 2011, 2018 - present; President from 2011 - 2015
- **Connecticut Forest and Park Association** – Board member from 1994 – 2002
- **North American Lakes Management Society** – Member since 1990

SELECTED PUBLICATIONS

PEER-REVIEWED SCIENTIFIC PAPERS

Siver, P.A., Sibley, J., Lott, AM., **Marsicano**, L.J. Temporal changes in diatom valve diameter indicate shifts in lake trophic status. J Paleolimnology 66, 127–140 (2021). https://doi.org/10.1007/s10933-021-00192-y

Siver, P., L. **Marsicano**, A. Lott, S. Wagener, N. Morris. 2018. Wind Induced Impacts on Hypolimnetic Temperature and Thermal Structure of Candlewood Lake (Connecticut, U.S.A.) from 1985-2015. Geo: Geography and the Environment. 5(2). https://doi.org/10.1002/geo2.56

Kohli, P., Siver, P.A., **Marsicano**, L.J., Hamer, J.S., and Coffin, A.M. 2017. Statistical Assessment of Long-term Trends for Management of Candlewood Lake, Connecticut, USA. Journal of Lake and Reservoir Management. 33:280-300

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Moore H.H., Niering W.A., **Marsicano** L.J, and Dowdell M. 1999. Vegetation change in created emergent wetlands (1988–1996) in Connecticut (USA). Wetland Ecology and Management. 7:177-191.

Siver, P.A. A. M. Lott, E. Cash, J. Moss, and L.J. **Marsicano**. 1999. Century changes in Connecticut, USA, lakes as inferred from siliceous algal remains and their relationship to land use changes. Limnology and Oceanography 44:1928-1935.

Siver, P.A. and L.J. **Marsicano**. 1996. Inferring trophic conditions using scaled chrysophytes. Beiheft zur Nova Hedwigia 114:233-246.

Siver, P.A., Canavan, R.W. IV, Field, C., **Marsicano**, L.J. and A.M. Lott. 1996. Historical changes in Connecticut lakes over a 55-year period. Journal of Environmental Quality 25: 334-345

Marsicano, L.J., J.L. Hartranft, P.A. Siver, and J.S. Hamer. 1995. An historical account of water quality changes in Candlewood Lake, Connecticut, over a sixty-year period using paleolimnology and ten years of water quality data. Journal of Lake and Reservoir Management 11:15-28.

Lott, A.M., Siver, P.A., **Marsicano**, L.J., Kodama, K.P. and R.E. Moeller. 1994. The paleolimnology of a small waterbody in the Pocono Mountains of Pennsylvania, USA: reconstructing 19th-20th century specific conductivity trends in relation to changing land use. Journal of Paleolimnology 12: 75-86.

Marsicano, L.J. and P.A. Siver. 1993. A paleolimnological assessment of lake acidification in five Connecticut lakes. Journal of Paleolimnology 9:202-221.

Siver, P.A. and L.J. **Marsicano**. 1993. *Mallomonas connensis* sp. nov., a new species of Synurophyceae from a small New England lake. Nordic Journal Botany. 13: 337-342

Siver, P.A. and L.J. **Marsicano**. 1991. Assessing acidification trends in Connecticut lakes using a paleolimnological approach. CT. Department of Environmental Protection Bulletin, 44 pp. + appendices

POLICY PAPERS AND SUBMITTALS

Marsicano, LJ. 2009. An Examination of Recreational Pressures on Candlewood Lake, CT. Candlewood Lake Authority. Sherman, CT. 68 pp.

Marsicano, L.J., et al. 2000 – 2017. Submittals of the Candlewood Lake Authority to the Federal Energy Regulatory Commission during license renewal and management plan processes for Housatonic Hydro, FERC Docket No. P-2576.

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PROFESSIONAL EXPERIENCE

Owner/Manager, Brawley Consulting Group LLC, Windsor, CT

(January 2008 to present).

Provides land conservation and management services to local land trusts and conservation organizations, including designing and implementing habitat restoration projects, grant writing, trail design and construction, crafting and monitoring conservation easement, boundary posting, Baseline Documentation Reports and developing property management plans. www.brawleycg.com

Land Manager, Naromi Land Trust, Sherman, CT

(March 2004 to present).

Manage all land trust properties and help acquire, monitor and enforce conservation easements. Responsibilities also include designing and building trails, securing funding for habitat restoration projects, and assisting with organizational and administrative tasks. Work cooperatively with the town and other conservation organizations to identify and prioritize lands for future acquisition. www.naromi.org

Land Manager, Kent Land Trust, Kent, CT

(September 2008 to August 2014).

Manage all land trust properties and help acquire, monitor and enforce conservation easements. Responsibilities also include securing funding for habitat restoration projects and preparing Baseline Documentation Reports (BDRs) and property management plans. Addressed backlog of stewardship items required for Kent Land Trust to become the second land trust in Connecticut accredited by the Land Trust Alliance.

Project Manager, Northeast Instream Habitat Program, Amherst MA.

(January 2004 to March 2005).

Coordinated all facets of two fisheries habitat assessment projects working with researcher at the University of Massachusetts, including project planning, data collection, hiring and overseeing seasonal staff, data analysis and report preparation. http://www.neihp.org/index.htm

Executive Director, Pomperaug River Watershed Coalition, Southbury, CT

(July 2001 to May 2003).

Managed all activities of non-profit watershed management organization dedicated to conserving regional water resources, including research, outreach, budgets, grant writing, website development, fundraising, and volunteer relations. www.pomperaug.org

Senior Project Manager, LabLite, LLC, New Milford, CT

(January 2000 to June 2001).

Product development, testing, sales, and customer service for a software company that provides Laboratory Information Management Software (LIMS) to environmental and other laboratories. www.lablite.com

Research Coordinator, The National Audubon Society, Southbury, CT

(March 1998 to January 2000).

Designed and implemented all research on birds and other wildlife at the 625-acre wildlife sanctuary. Conducted natural resources inventory, created checklists of wildlife and plants, established environmental education programs, and coordinated cooperative research projects with state agencies and regional conservation organizations. http://ct.audubon.org/IBA_BOR.html

Environmental Analyst, Land-Tech Consultants, Inc., Southbury, CT

(November 1996 to February1998).

As Project Manager conducted environmental impact statements, wetland assessments, and wildlife surveys; prepared federal, state and local permit applications; designed pond and tidal wetland restoration projects; and conducted lake diagnostic studies. Worked with state agencies and local land use agencies to mitigate impacts of residential and commercial development projects. www.landtechconsult.com

Wetland Ecologist, The Deep River Land Trust, Deep River, CT.

(July to October 1995).

Worked in association with The Nature Conservancy Connecticut Chapter on a conservation project at two freshwater tidal marshes in the lower Connecticut River. Position entailed mapping dominant vegetation communities, identifying potential environmental impacts, researching information on appropriate buffer zones and recommending methods for long-term monitoring of the system.

Research Assistant, The Nature Conservancy CT Chapter, Weston, CT.

(May to July 1995).

Assisted with research on the productivity and survivorship of Worm-eating Warblers at the 1700-acre Devil's Den Preserve in Weston, CT. Responsibilities included mist-netting, bird banding, and locating and monitoring approximately 25 nest sites throughout the breeding season.
http://www.nature.org/wherewework/northamerica/states/connecticut/

Master's Thesis Research, Connecticut College, New London, CT.

(September 1993 to May 1995).

Conducted two-year study investigating relationships between bird populations and environmental conditions in tidal wetlands of Connecticut. Quantified bird use, vegetation, and selected environmental parameters in eight tidal marsh systems on the Long Island Sound to assess the use of birds as indicators of environmental quality.
http://www.conncoll.edu/departments/botany/index.htm

Research Associate, Connecticut College Arboretum, New London, CT.

(Sept. 1992 to January 1994).

Conducted a natural resources inventory of The Harriet C. Moore Foundation property in Westerly, RI, including producing lists of all plants and animals (flora and fauna), conducting a breeding bird census, and identifying and tagging over 100 ornamental trees. Developed a five-year plan for the management and use of this 35-acre public land preserve.
http://arboretum.conncoll.edu/

Principal Investigator, The Nature Conservancy CT Chapter, Middletown, CT

(Summer 1994).

Studied five marshes in the tidelands of the lower Connecticut River to assess the impacts of the spread of common reed (*Phragmites australis*) on bird populations. Designed project that included the systematic collection of data on bird use, vegetation sampling and an analysis of physical site characteristics.
http://www.nature.org/wherewework/northamerica/states/connecticut/

EDUCATION

Connecticut College, New London, CT. Master of Arts in Botany, 1995.

Connecticut College, New London, CT. Bachelor of Arts in American History, 1982.

The Loomis Chaffee School, Windsor, CT. Graduated 1978.

PUBLICATIONS

Brawley, A. H., Zitter, R. and L. Marsicano, Editors. 2005. Candlewood Lake Buffer Guidelines. Candlewood Lake News *Special Edition*, Vol 1:21.

Warren, R.S., P. E. Fell, R. Rozsa, A. H. Brawley, A. C. Orsted, E. T. Olson, V. Swamy and W. A. Niering. 2002. Salt Marsh Restoration in Connecticut: 20 years of Science and Management. *Restoration Ecology* 10 (3) 497-513.

Markow, J. and H. Brawley. 2001. Herpetofaunal and Avifaunal Surveys of Vaughn's Neck Peninsula, Candlewood Lake, Connecticut. Report to the Town of New Fairfield, CT. 32 p.

Brawley, A. H. 1998. A Vegetation Survey and Conservation Analysis of Vaughn’s Neck Peninsula. Report to The Candlewood Lake Authority. The National Audubon Society. 11 p.

Brawley, A. H., R. S. Warren and R. A. Askins. 1998. Bird Use of Restoration and Reference Marshes Within the Barn Island Wildlife Management Area, Stonington, Connecticut, USA. *Environmental Management* 22(4): 625-633.

Marsicano, L. J. and A. H. Brawley. 1997. Land Use, Watersheds, and Aquatic Resources. *Connecticut Woodlands* 62(3): p. 21.

Niering, W. A., and A. H. Brawley. 1996. Functions and Values Assessment of Area A Downstream Wetlands and Watercourses. Naval Submarine Base New London, Groton, CT. Report to Brown & Root Environmental, The Environmental Protection Agency, and The United States Navy. 36 p.

Brawley, A.H. 1995. Pratt and Post Coves: A Vegetation Survey and Conservation Analysis. Report to the Deep River Land Trust, Deep River, CT. 62 p.

Brawley, A.H. 1995. Birds of Connecticut's Tidal Wetlands: Relating Patterns of Use to Environmental Conditions. Master's Thesis, Connecticut College, New London, CT. 87 p.

Brawley, A.H. 1994. Birds of the Connecticut River Estuary: Relating Patterns of Use to Environmental Conditions. Report to the Nature Conservancy Connecticut Chapter Conservation Biology Research Program, Middletown, CT. 23 p.

Brawley, A.H., G.D. Dreyer. 1994. Master Plan for the Future Management and Use of Moore Woods. Connecticut College Arboretum Publication. New London, CT. 65 p.

Brawley, A.H., G.D. Dreyer and W.A. Niering. 1993. Connecticut College Arboretum Phase One Report to the Harriet Chappell Moore Foundation. Connecticut College Arboretum Publication. New London, CT. 100 p.

ACTIVITIES

Forest and Trails Conservation Committee, Connecticut Forest & Park Association (CFPA)

Coverts Project Cooperator, UConn Cooperative Extension System

WILLIAM HENLEY
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EXPERIENCE

Sr. Aquatic Resource Scientist **May 2018 – Present**
South Central Connecticut Regional Water Authority, New Haven, CT

- Supervise source water quality monitoring at a regional water company serving 400,000+ customers in 15 towns
- Coordinate field work on 11 active reservoirs totaling over 2,000 acres as well as watershed lands (26,000 acres)
- Manage environmental programs and initiatives as they relate to source waters (streams, reservoirs, aquifers)
- Administer program budgets as well as plan and manage special projects
- Collect in situ profile data and water samples at reservoirs to monitor reservoir ecosystem health
- Process samples at the Authority's laboratory; analyze, interpret, and report the results
- Conduct aquatic macrophyte surveys and manage downstream release compliance
- Attend public events and participate in environmental outreach and education
- Contribute to ecological restoration by collaborating with various departments and external stakeholders
- Operate/maintain water quality sampling equipment as well as company boat and vehicle

Core accomplishments:

- Played a focal role in the realization of a new stream water quality monitoring initiative
- Worked on the discovery and management of an invasive species in one of the Authorities waterbodies
- Provided management and oversight of downstream release from reservoirs, and determined needs to meet new regulations.

Fisheries Durational Resource Technician **Apr – Nov 2016, July 2017 – Mar 2018**
DEEP Inland Fisheries Division, Marlborough, CT

- **Provided support to fisheries biologists by collecting biological data as it relates to fish and** aquatic ecosystems at over 65 stream sites and 25 lakes and ponds
- Conducted monitoring of stream and lake fish species utilizing electrofishing boats, backpacks, and trap nets
- Performed surveys of freshwater anglers on statewide lakes 2-3 days weekly in open water and ice conditions
- Collected and cataloged freshwater mussels for identification in conjunction with the CT DEEP Wildlife Division
- As a lead observer, facilitated stream crossing and culvert assessments for the North Atlantic Aquatic Connectivity Collaborative (NAACC)
- Collected dissolved oxygen profiles on lakes intended for stocking as part of a long-term dataset
- Provided direction to new technicians and oversaw volunteers

Core accomplishments:

- Served as a leader for sampling crews as well as managed several critical projects
- Participated in the discovery/confirmation of an invasive plant now found in the Connecticut River
- Contributed to a study of winter road sand impact presented at the Connecticut Natural Resource Conference
- Certified as North Atlantic Aquatic Connectivity Collaborative (NAACC) Lead Observer
- Participated in wild Brook Trout PIT tagging initiative

Adjunct Limnologist **July 2015 – January 2023**
Aquatic Ecosystem Research, Branford, CT

- Worked as a technician under the companies' principal limnologist executing field work on over a dozen freshwater bodies of water totaling over 2,500 acres
- Performed water quality monitoring and algal sampling using YSI Water Quality Sondes and Van Dorn samplers

- Conducted aquatic plant community surveys, including invasive species monitoring
- Collected various geospatial data, aquatic plant data, bathymetric data, and infrastructure data to conduct research, analyze trends, and create geospatial products and maps
- Compiled and managed large data sets and generated accurate reports on a regular basis
- Closely collaborated with freshwater and coastal stakeholders on the creation and planning of conservation and management projects

Core accomplishments:

- Developed various geospatial methodologies for assessment of ecological systems
- Implemented new techniques for monitoring aquatic plant communities
- Created standardized templates for company map products
- Integrated new technologies for bathymetric and plant mapping
- Participated in research initiatives for various projects, including authorship on a research paper

Wildlife, Geospatial & Field Technician **May 2016 – May 2018**
Davison Environmental, Chester, CT

- Worked independently on a variety of projects performing various assorted environmental work as a subcontracted environmental technician
- Accountable for geospatial data collection and analysis
- Participated in wetland and plant surveys as well as mapping initiatives
- Conducted wildlife surveys for amphibians, herps, birds, and bats
- Solely developed geospatial techniques and maps

Environmental & Aquatic Field Technician **Summer 2013 – 2015**
All Habitat Services, Branford, CT

- Identified and removed invasive wetland, upland, and aquatic vegetation by applying pesticides
- Conducted aquatic vegetation surveys and water quality sampling as well as produced professional map products
- Implemented new geospatial methodologies to survey sediments and bathymetry

EDUCATION

B.A. in Geography with minor in Wildlife Conservation • *University of Delaware* **Spring 2015**
Graduate Certificate Geographic Information Science • *University of Delaware* **Spring 2015**

VOLUNTEER & COMMUNITY SERVICE

White-tailed Deer Capture & Tracking • University of Delaware, Milton, Delaware **Jan – Apr 2015**
Marsh Bird Surveys • St. Jones Reserve, Dover, Delaware **June – July 2014**
President of National Meteorological Society Student Chapter, University of Delaware
Boy Scouts of America • Rank of Life Scout

PUBLICATIONS

June-Wells M, Simpkins T, Coleman AM, **Henley W**, Jacobs R, Aarrestad P, Buck G, Stevens C, Benson G. (2017) Seventeen years of grass carp: an examination of vegetation management and collateral impacts in Ball Pond, New Fairfield, Connecticut. Lake and Reservoir Management 33:84–100

SKILLS & QUALIFICATIONS

Proficient with Microsoft Word, Excel, and PowerPoint
Proficient with ArcMap, ENVI, and GIS Data Procurement
Basic knowledge of NCL, Python, R, and Unix
Proficient in use of GPS hand and backpack units (Garmin, Trimble)
Proficient in a variety of water quality sampling techniques/equipment
North Atlantic Aquatic Connectivity Collaborative (NAACC) Lead Observer
National Weather Service Skywarn storm spotter training
North American Lake Management Society Certifield Lake Manager