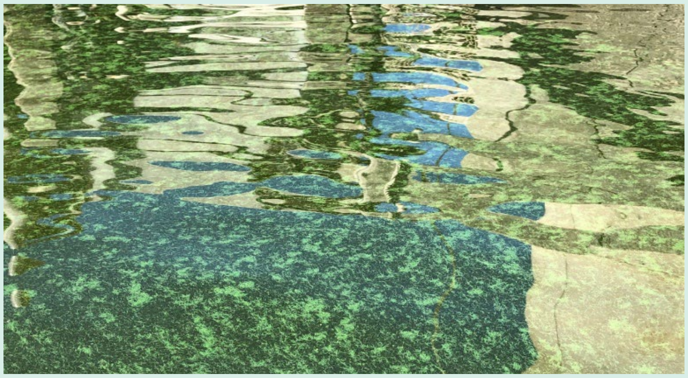
FINAL TECHNICAL REPORT◦ NOVEMBER 2022

Lake Henshaw and Lake Wohlford Harmful Algal Blooms Management and Mitigation Plan



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Cover photos: Lake Henshaw looking south and east (top left) with shoreline cyanobacteria accumulation (bottom left). Lake Wohlford cyanobacteria bloom (top right) and area near boat dock (bottom right). Lake Wohlford bloom photo taken by City of Escondido. Other photos taken in March 2021 by Stillwater Sciences.

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EXECUTIVE SUMMARY

Introduction

Located in northern San Diego County, California, Lake Henshaw and Lake Wohlford experience seasonal cyanobacteria blooms that can affect municipal, domestic, agricultural, and recreational water supply for use by the Vista Irrigation District (District), the City of Escondido (Escondido), and the La Jolla, Rincon, San Pasqual, Pauma, and Pala Bands of Indians (the Bands). In lakes and reservoirs, excessive cyanobacteria can result in low dissolved oxygen (DO), high pH, high un-ionized ammonia, and problematic levels of one or more cyanotoxins, including microcystin, cylindrospermopsin, anatoxin-a, and saxitoxin. At elevated concentrations, cyanotoxins can cause public health concerns and bioaccumulate in the tissue of aquatic biota, such as shellfish, fish, and marine mammals, potentially harming these organisms as well as the humans that consume them. Cyanobacteria blooms containing toxins are often referred to as harmful algal blooms (HABs).

Currently there are no state or federal standards for HABs with respect to recreational water or drinking water uses; however, California has developed guidance for recreational water bodies and for drinking water and the World Health Organization (WHO) has developed provisional guidelines for both water uses. While several species of cyanobacteria are known to produce specific toxins (e.g., *Microcystis aeruginosa* can produce microcystin), not all blooms of a particular species produce toxins at all times. When produced, toxins typically are released into the surrounding environment after a bloom senesces (dies) or cells are ruptured, at which point the toxins can be taken in by higher organisms through their diet or by ingesting contaminated water.

In March 2020, the District began monitoring for the presence of cyanobacteria and cyanotoxins in Lake Henshaw. In July of 2020, laboratory analysis confirmed the presence of elevated levels of the cyanotoxin microcystin in the water released from Lake Henshaw. Cyanobacteria and cyanotoxins have been measured in Lake Wohlford since July 2020.

This iteration of the *Lake Henshaw and Lake Wohlford HABs Mitigation and Monitoring Plan* represents the first phase of a project to manage HABs and cyanotoxins in Lake Henshaw and Lake Wohlford. Phase I objectives include the following:

- Develop short-term solutions for mitigating or treating HABs;
- Screen potential long-term alternatives for preventing or minimizing HABs;
- Develop a HABs Mitigation and Monitoring Plan that includes a Water Quality Monitoring Plan; and
- Gather relevant data to inform future phases of the project.

Here, short-term is defined as occurring prior to 2024 and long-term is defined as occurring in the year 2024 or later.

Knowledge gained during Phase I will inform future phases of the project, including further ranking and prioritization of long-term prevention/minimization alternatives, selection of the preferred long-term alternative(s), and updates to the Water Quality Monitoring Plan and the HABs Management and Mitigation Plan, as needed.

Local System Water Quality

The Warner Ranch Wellfield, local tributaries to Lake Henshaw and the lake itself, the San Luis Rey River (between Henshaw Dam and the Diversion Dam), the Escondido Canal, and Lake Wohlford comprise the Local Water System. Based on available data, several factors influence water quality in the Local Water System and HABs production in Lake Henshaw and Lake Wohlford in particular.

At the upstream end of the Local Water System, watershed runoff into Lake Henshaw during spring, summer, and fall (typically March–November) is generally negligible, although runoff can occur in March and April. Watershed runoff is highest in winter (December–February) although it is variable by year and information regarding associated nutrient inputs is not currently available. Pumped groundwater transfers from the Warner Ranch Wellfield in all seasons contribute nitrate (NO_3^-), orthophosphate (PO_4^{3-}), and iron (Fe) to Lake Henshaw in concentrations and at loading rates sufficient to stimulate cyanobacterial blooms.

Information regarding seasonal stratification patterns for water temperature and dissolved oxygen [DO] in Lake Henshaw is limited. Lake bottom waters and the sediment-water interface are assumed to regularly undergo periods of low DO concentrations during spring, summer, and fall, where low DO at the sediment-water interface facilitates biogeochemical processes that release bioavailable nutrients from organic-rich bottom sediments. Based on experimental sediment chamber studies conducted as part of this project, ammonia (NH_4^+) and orthophosphate release rates from Lake Henshaw sediments are high compared to those reported for Lake Wohlford and other hypereutrophic (i.e., very high algal productivity) lakes in California. Internal nutrient loading (i.e., from lake sediments), particularly in deeper areas of the lake, is likely to contribute a much larger proportion of bioavailable nutrients to the lake than does external loading (i.e., from groundwater transfers or runoff) under existing conditions.

High concentrations of bioavailable nutrients in the Lake Henshaw water column and warm water temperatures support large cyanobacteria blooms throughout the shallow water column. As a measure of cyanobacteria biomass, chlorophyll-a concentrations in Lake Henshaw have increased relative to historical conditions (although data are limited), as have nutrient concentrations, and the lake appears to have shifted from a nitrogen-limited system to one that is phosphorus-limited on a total nutrient basis. On a bioavailable nutrient basis, nitrogen is relatively less available than phosphorus, although both are present at high levels throughout the year.

Both microcystin and anatoxin-a have been detected in Lake Henshaw, with the highest concentrations tending to coincide with the highest cyanobacteria cell counts (using remote sensing data). In August 2020, microcystin concentrations in the reservoir peaked above 1,000 μg/L, exceeding the CCHAB warning (6.0 μg/L) and danger (20.0 μg/L) thresholds for multiple weeks in 2020. Concentrations did not consistently decline below the CCHAB caution threshold of 0.8 μg/L until March 2021. In 2021, the peak microcystin concentration reached 11 μg/L in late fall/early winter (November). Anatoxin-a was detected occasionally from October 2020 through June 2021 and consistently from November 2020 through January 2021 in Lake Henshaw; thus this toxin also exceeded the CCHAB caution threshold of detection, (i.e., less than or equal to 0.15 μg/L) for multiple weeks during recent monitoring. The primary microcystin producer in Lake Henshaw may be *Microcystis sp.* and the primary anatoxin-a producer may be *Dolichospermum sp.*, although genetic studies are necessary to confirm the primary producers of both microcystin and anatoxin-a in Lake Henshaw. It is currently unknown whether benthic algae are contributing to cyanotoxin production in Lake Henshaw. Following the peak of the cyanobacteria bloom, algae cells settle to the bottom sediments, contributing organic matter, nutrients, and potentially persistent cyanotoxins to the sediments, and contributing to future internal loading.

Water released from Lake Henshaw transports orthophosphate, ammonia, nitrate, and other bioavailable nutrients downstream into the San Luis Rey River and Escondido Canal, but these nutrients do not appear to undergo substantial transformation during the short transit time (i.e., hours to days) to Lake Wohlford. Cyanobacteria and cyanotoxins, including microcystin and anatoxin-a, produced in Lake Henshaw are transported downstream when water is released from Lake Henshaw, but do not appear to be altered during the short transit time (i.e., hours to days) to Lake Wohlford.

Watershed runoff into Lake Wohlford during spring, summer, and fall (typically March–November) is generally negligible. Watershed runoff is highest in winter (December–February) although it is variable by year. Information regarding nutrient inputs associated with runoff into Lake Wohlford is not currently available. Water transfers from the Lake Henshaw releases, via the San Luis Rey River and Escondido Canal, contribute ammonia, nitrate, orthophosphate, and iron to Lake Wohlford in concentrations sufficient to stimulate additional cyanobacterial bloom formation in this downstream waterbody. Water transfers also contribute cyanobacteria and cyanotoxins to Lake Wohlford when these constituents are present in the Lake Henshaw release water.

Occasional brief periods of water temperature stratification can occur in Lake Wohlford throughout the year, although the majority of the time the water column is well-mixed. The regular lack of water temperature stratification in the deepest Lake Wohlford waters is likely due to the artificial aeration system that, while not directly transferring much oxygen to the lake, mixes the water column near the dam for 19 hours a day, 7 days a week. The current condition and efficiency of the aeration system is unknown, although systems of this type generally supply little oxygen directly to the water since compressed air contains only 21% oxygen and the coarse air bubbles rise rapidly through the water column and exit to the atmosphere before oxygen can dissolve into the surrounding water. Accordingly, DO concentrations do exhibit stratification in Lake Wohlford throughout the year, with bottom water DO at low or zero concentrations and surface water DO at moderate to high concentrations, particularly in the spring. Multiple brief periods of low DO (< 4 mg/L) in most or all of the water column can occur in summer and fall despite artificial aeration system operations. Low DO concentrations are not typical during winter. Based on experimental sediment chamber studies conducted as part of this project, ammonia and orthophosphate release rates from Lake Wohlford sediments are similar to other hypereutrophic (i.e., very high algal productivity) lakes in California. Internal nutrient loading (i.e., from lake sediments), particularly in deeper areas of the lake, is likely to contribute a much larger proportion of bioavailable nutrients to the lake than does external loading (i.e., from Lake Henshaw transfers or runoff), although in years with heavy runoff external loading from runoff may be the highest.

Moderate concentrations of nitrate in the Lake Wohlford water column and warm water temperatures contribute to large cyanobacteria blooms throughout the lake. There are currently no available data to characterize phosphorus concentrations in Lake Wohlford. The lake is periodically treated with a chelated-copper based algaecide to control HABs. Escondido currently treats the lake when concentrations of microcystin exceed 0.3 μg/L. There was one treatment conducted in early July 2021, in response to elevated concentrations of microcystin in June 2021, although by the time of treatment microcystin concentrations had decreased in Lake Wohlford to below 0.3 μg/L. Since June 2021, when anatoxin-a concentrations began to be measured at multiple locations in Lake Wohlford, they have been consistently below the analytical laboratory reporting limit (i.e., <0.03 μg/L). Note that the laboratory reporting limit for anatoxin-a in the Lake Wohlford samples is five times lower than that of the Lake Henshaw samples. Genetic studies are necessary to confirm the primary producers of microcystin in Lake Wohlford. It is currently unknown whether benthic algae are contributing to cyanotoxin production in the lake. Following the peak of the cyanobacteria bloom, algae cells settle to the bottom sediments, contributing organic matter, nutrients, and potentially persistent cyanotoxins, to the sediments, and contributing to future internal loading.

Consideration of Potential Short-Term Mitigation Methods

Based upon a site visit and screening workshop, the Project Team[1] evaluated 18 potential in-lake management methods for their applicability in addressing HAB occurrences in Lake Henshaw in the short term, and to narrow the list of potential mitigation and treatment methods to one or two approaches that would be most suitable for implementation in 2021. Initial consideration of a broad list of potential management methods was undertaken to ensure that the project did not inadvertently overlook possible viable approaches for short term application.

Screening of short-term HAB mitigation and treatment alternatives discussed at the April 2021 workshop focused on implementation in Lake Henshaw because existing data suggest that while Lake Wohlford can produce cyanobacteria and cyanotoxins *in situ*, HABs in Lake Henshaw can produce high levels of cyanotoxins that are transported downstream through the San Luis Rey River, Escondido Canal, and into Lake Wohlford, including water that is intended to meet scheduled deliveries to the Indian Water Authority. Algaecides were selected by the Project Team as the most feasible short-term HABs control method for Lake Henshaw for the following reasons:

- Algaecide application is a well-proven mitigation method for HABs. Approved algaecide chemicals act quickly (i.e., minutes to hours) and can prevent the formation of and interrupt an ongoing HAB and to stop cyanotoxin production. Some active ingredients can also destroy cyanotoxins in the water column (e.g., hydrogen peroxide).
- Little to no capital investment is required for algaecide application, since licensed applicators can be hired by the District to apply the chemicals and undertake monitoring needed to meet permit requirements.
- Costs are generally predictable and there are multiple algaecide products available on the market.
- In June 2021, the District obtained a Statewide Aquatic Weed Control Permit for application of copper sulfate, chelated copper, and sodium carbonate peroxyhydrate (peroxide) to control algae/cyanobacteria in Lake Henshaw.

The District is currently obtaining experience with the use of both copper- and peroxide-based algaecides in the lake.

[1] The Project Team included staff from the District and Escondido involved in managing Lake Henshaw, the Escondido Canal, and Lake Wohlford, members of the Stillwater Sciences' Team, and Dr. David Caron, as a technical expert representing the Indian Water Authority. The Stillwater Sciences' Team included Stillwater Sciences, Brown and Caldwell, Alex Horne Associates, Robertson-Bryan Inc., University of California at Merced, Marine Biochemists, Water Quality Solutions, and Mr. Bill Taylor.

Consideration of Potential Long-Term Prevention and Mitigation Methods

Following additional data review and analysis, the Project Team participated in a two-part workshop to screen alternatives for preventing or minimizing HABs in Lake Henshaw and Lake Wohlford in the long term, where the latter is defined as occurring in the year 2024 or later. The Project Team evaluated the 18 potential in-lake management methods that were previously evaluated for short-term applicability in Lake Henshaw. The screening of long-term alternatives also considered five potential out-of-lake management methods. Based on feedback from the screening workshop for long-term alternatives, the Project Team undertook further evaluation of a subset of selected management methods judged to have the greatest applicability and suitability for long-term water quality improvements in each lake, given the available information. The subset of selected methods is listed below for each lake. Estimated costs are presented in Table ES-1 and Table ES-2.

Lake Henshaw

- Selected HABs Prevention Methods
  - Out-of-lake: ***source water nutrient control*** to prevent HABs by inactivating bioavailable phosphorus (i.e., orthophosphate) in Warner Ranch Wellfield inflows to Lake Henshaw, which will reduce external loading of this nutrient to the lake.
  - In-lake: ***phosphorus inactivation/chemical sediment sealing*** to prevent HABs by removing bioavailable phosphorus (i.e., orthophosphate) from the water column and minimizing or eliminating orthophosphate release from lake sediments during low DO conditions through application of a chemical (e.g., alum, lanthanum) to the lake.
  - In-lake: ***oxygenation via Speece Cone or SDOX***[2] to prevent HABs by maintaining positive DO concentrations in the water column and at the sediment/water interface, which will decrease the release of orthophosphate, ammonia, and other oxidation reduction potential (ORP)-sensitive constituents from the reservoir sediments during periods of low DO.
- Selected HABs Mitigation Methods
  - Out-of-lake: ***bypass pipeline*** to mitigate HABs by providing cyanotoxin-free water downstream of the Henshaw Dam spillway through the rerouting of groundwater from the Warner Ranch Wellfield around Lake Henshaw and into the San Luis Rey River downstream of Henshaw Dam, thus providing the District, Escondido and the Bands with a reliable and consistent delivery method, added flexibility regarding the timing of water deliveries, and reductions in evaporative losses that occur within Lake Henshaw. Note that the financial viability of the Local Water System relies on runoff into Lake Henshaw as well as wellfield production, hence the construction of a bypass pipeline would also require a strategy to prevent or mitigate in-lake HABs production.
  - In-lake: ***algaecide treatment*** to mitigate HABs by controlling the formation and growth of nuisance algae blooms (filamentous, planktonic, benthic, or cyanobacteria) by killing the organisms responsible for poor water quality through the application of a chemical (i.e., copper- or peroxide-based) to the lake.

[2] The two most common types of oxygenation systems for lakes that create water supersaturated with dissolved oxygen include the non-pressurized Speece Cone technology and the pressurized supersaturated dissolved oxygen (SDOX) system. The ECO_2 System is a Speece Cone system manufactured by ECO_2 (Indianapolis, IN). The SDOX O2® is a patented system manufactured by ChartWater BlueInGreen (Fayetteville, AR).

Table ES-1. Summary of estimated costs for implementation of selected HABs prevention and mitigation methods in Lake Henshaw.

Type of Control	Method	Out-of-lake	In-lake	Potential to Implement in Short term (2021–2023)	Recommended for Long Term (post-2024)	Estimated Costs
HABs Prevention						
Chemical	Source water nutrient control	X			X	$50K–$350K per year chemical costs; $5K–$25K application equipment
	Phosphorus inactivation/chemical sediment sealing		X	X	X	$160K–$4.5M depending on area of treatment and chemical used
	Oxygenation via Speece Cone or SDOX		X		X	$2M–$7M design and implementation; $100K–$200K annual operation and maintenance (O&M)
HABs Mitigation						
Physical	Bypass pipeline	X			X	Capital cost: $22M–$43M O&M: $220K per year
Chemical	Algaecide treatment		X	X	X	$75K–$400K per year

Lake Wohlford

- Selected HABs Prevention Methods
  - In-lake: ***oxygenation via Speece Cone or SDOX***[2] to prevent HABs by maintaining positive DO concentrations in the water column and at the sediment/water interface, which will decrease the release of orthophosphate, ammonia, and other ORP-sensitive constituents from the reservoir sediments during periods of low DO.
- Selected HABs Mitigation Methods
  - In-lake: ***selective withdrawal*** to mitigate HABs by allowing Escondido operations staff to withdraw water of consistent quality from within the lake through the use of multiple outlets located at different water column depths.
  - In-lake: ***algaecide treatment*** to control the formation and growth of nuisance algae blooms (filamentous, planktonic, benthic, or cyanobacteria) by killing the organisms responsible for poor water quality through the application of a chemical (i.e., copper- or peroxide-based) to the lake.

Table ES-2. Summary of estimated costs for implementation of selected HABs prevention and mitigation methods in Lake Wohlford.

Type of Control	Method	Out-of-lake	In-lake	Potential to Implement in Short term (2021–2023)	Recommended for Long Term (post-2024)	Estimated Costs
HABs Prevention						
Chemical	Oxygenation via Speece Cone or SDOX		X		X	\$6.3M–\$11M design and implementation; \$185K–\$360K annual O&M
HABs Mitigation						
Physical	Selective withdrawal		X		X	Capital cost: \$1.8M–\$2.6M
Chemical	Algaecide treatment		X	X	X	Up to \$45K per year

Details regarding implementation considerations, anticipated implementation schedule, compatibility, estimated costs, permit requirements and additional information needs to further rank and prioritize the subset of selected long-term alternatives for Lake Henshaw and Lake Wohlford are provided in Section 4.

Water Quality Monitoring Plan

The ability to track progression of a potentially cyanotoxin-causing bloom requires sampling early in HAB development, when water concentrations of cyanobacteria and cyanotoxins may still be relatively low. Because HABs can develop rapidly (i.e., over the course of several days), early warning that a HAB may be forming in Lake Henshaw and/or Lake Wohlford is critical for successful application of algaecides as the current short-term mitigation strategy. Although licensed applicators can be hired to apply the chemicals and undertake monitoring needed to meet permit requirements, the contracting, selection and procurement of the appropriate chemical(s), transport of the chemical(s) to the site, and deployment of specialized equipment and personnel to treat a large lake, require lead time that typically extends two to four weeks. Thus, appropriate triggers are needed to allow the District and Escondido to move from an *operational strategies window*, before a HAB occurs and when multiple options for reservoir operation are still available, to an *early warning window*, when monitoring data suggest that a HAB may be developing, and, as needed, to a *treatment window*, when algaecide application would occur prior to a HAB becoming out of control.

For short-term HABs mitigation, this monitoring plan uses operational triggers to transition from routine (weekly) monitoring at a small number of index sites during the reservoir operational strategies window, to rapid response monitoring at a greater number of sites and at a higher frequency (sub-weekly) during the early warning window, and, as needed, to algaecide treatment. The triggers for these transitions involve specified increases in water quality parameters (including cell counts [also referred to as cell density], cyanotoxin concentrations, and/or other *in situ* water quality parameters) between consecutive samples collected at one or more monitoring sites, with greater relative increases required to move from the early warning window to the treatment window. The operational triggers also indicate whether it is too late to apply algaecides to Lake Henshaw or Lake Wohlford because the bloom is too dense to be effectively treated with an algaecide, and when to reduce monitoring efforts because the HAB has been eliminated.

Effective long-term management of complex water resources, including the Local Water System, requires that the analysis and interpretation of water quality monitoring data occur periodically in order to support the scientific "learning while doing" inherent to adaptive management. Knowledge gained during previous monitoring periods should be used to evaluate whether monitoring goals and objectives are being met through the monitoring program and this evaluation may result in recommendations for revisions. While this water quality monitoring plan has been developed to assist the District and Escondido in the long-term management of the Local Water System using a detailed process of existing data compilation and analysis (Section 2), consideration of potential short-term HAB mitigation strategies for Lake Henshaw and the selection of algaecides as the most appropriate short-term strategy (Section 3), and screening of potential long-term HAB prevention strategies for Lake Henshaw and Lake Wohlford (Section 4), implementation of this plan will provide new information about the physical, chemical, and biological processes controlling HABs in the water system, and this new information may change future monitoring efforts. For example, data collected to characterize HAB development and the Lake Henshaw response to algaecide application may change the number of monitoring sites, the specific parameters being monitored, and/or the established operational triggers needed to effectively manage HABs in this waterbody and its receiving waters.

Additionally, while multiple water quality monitoring techniques are included in this plan, including synoptic, routine, rapid response, algaecide effectiveness, and permit-related monitoring (Table ES-3), open source cyanobacterial remote sensing data and new approaches for less expensive and/or less logistically complicated analyses of cyanotoxins are becoming more available with time. This water quality monitoring plan assumes that water quality monitoring data collected by the District and Escondido will be reviewed regularly to aid in adaptive decision making, and the data will be compiled, analyzed, and included in periodic reports. Consistent with an adaptive management approach, results and recommendations for plan revisions, as applicable and justified by knowledge gained during the previous monitoring period, will be included as part of future reporting for the Local Water System.

Table ES-3. Estimated annual monitoring costs for monitoring associated with the use of algaecides as a mitigation strategy in Lake Henshaw and Lake Wohlford.

Monitoring Type	Labor Cost	Field Expenses Cost	Contract Laboratory Expenses	Total Cost
Lake Henshaw				
Synoptic[1]	$61,500	$14,400	$5,700	$81,600
Routine[2]	$26,000	$0	$75,600	$101,600
Rapid Response[3]	$6,000	$3,400	$26,100	$35,500
Algaecide Effectiveness[4]	$8,000	$1,400	$11,900	$21,300
Statewide Aquatic Weed Control Permit Monitoring[5]	Costs included in algaecide application contract			
Total	*$101,500*	*$19,200*	*$119,300*	*$240,000*
Lake Wohlford				
Synoptic[1]	$45,300	$14,400	$4,700	$64,400
Routine[2]	$26,000	$0	$12,800	$38,800
Rapid Response[3]	$6,000	$0	$9,500	$15,500
Algaecide Effectiveness[4]	$8,000	$0	$3,200	$11,200
Statewide Aquatic Weed Control Permit Monitoring[5]	Costs included in algaecide application contract			
Total	*$85,300*	*$14,400*	*$30,200*	*$129,900*

1 To establish a set of representative monitoring sites that characterize the range of spatial and seasonal variability of common HAB indicators and cyanotoxins.
2 To provide the District with evidence that a HAB may be developing.
3 To allow sufficient response time for the successful implementation of algaecide as a short-term HAB mitigation strategy.
4 To determine whether algaecide application was effective at meeting water quality improvement objectives.
5 To determine whether algaecide application met the permit requirements.

1 INTRODUCTION

1.1 Background

Located in northern San Diego County, California, Lake Henshaw is a 52,000-acre foot (AF) surface water impoundment of the upper San Luis Rey River, north and east of the city of San Diego (Figure 1-1). The Vista Irrigation District (District) operates Lake Henshaw and groundwater wells within the 43,000-acre Warner Ranch surrounding the lake to provide municipal, domestic, agricultural, and recreational water supply for use by the District, the City of Escondido (Escondido), and the La Jolla, Rincon, San Pasqual, Pauma, and Pala Bands of Indians (the Bands). The lake also stores local runoff from several perennial tributaries including the west fork and mainstem of the San Luis Rey River, the Agua Caliente, San Ysidro, Buena Vista, Matagual, and Carrista creeks, and several unnamed creeks (Figure 1-1). Other designated beneficial uses for Lake Henshaw water include industrial process and service supply; freshwater replenishment; rare, threatened, or endangered species habitat; hydropower generation; warm freshwater habitat; contact and noncontact recreation; and wildlife habitat (San Diego Regional Water Quality Control Board 2021).

Water released from Lake Henshaw flows for 10 miles in the San Luis Rey River before being diverted into the 14-mile-long Escondido Canal via the Escondido Canal Diversion Dam. The Diversion Dam and the start of the Escondido Canal are located within the La Jolla Reservation (Figure 1-1). The canal supplies the 6,500 AF Lake Wohlford, owned by Escondido. Escondido Canal also supplies water to meet the annual entitlement for the Rincon Indian Reservation, which is diverted from the canal back into the San Luis Rey river at a small measurement flume just downstream of the Diversion Dam. The Warner Ranch Wellfield, local tributaries to Lake Henshaw and the lake itself, the San Luis Rey River (between Henshaw Dam and the Diversion Dam), the Escondido Canal, and Lake Wohlford comprise the Local Water System. Water released from Lake Wohlford is blended with imported water delivered by San Diego County Water Authority before treatment at the Escondido-Vista Water Treatment Plant (EVWTP) as a potable water supply for Escondido and the District. Other designated beneficial uses for Lake Wohlford water include agricultural water supply, hydropower generation, warm freshwater habitat, cold freshwater habitat, contact and noncontact recreation, and wildlife habitat (San Diego Regional Water Quality Control Board 2021).

The District was alerted to the possibility of Harmful Algal Blooms (HABs) in Lake Henshaw by remote sensing data made available through the California HABs portal (https://fhab.sfei.org/) (California Water Quality Monitoring Council 2022a). In lakes and reservoirs, excessive seasonal cyanobacteria can result in low dissolved oxygen (DO), high pH, high un-ionized ammonia, and problematic levels of one or more cyanotoxins, including microcystin, cylindrospermopsin, anatoxin-a, and saxitoxin. At elevated concentrations, cyanotoxins can cause public health concerns and bioaccumulate in the tissue of aquatic biota, such as shellfish, fish, and marine mammals, potentially harming these organisms as well as the humans that consume them. Cyanobacteria blooms containing toxins are often referred to as HABs.

Currently there are no state or federal standards for HABs with respect to recreational water or drinking water uses; however California has developed guidance for recreational water bodies through the California Cyanobacteria Harmful Algal Bloom (CCHAB) Network and for drinking water through the Office of Environmental Health Hazard Assessment (OEHHA) (Table 1-1). The World Health Organization (WHO) has developed provisional guidelines for both recreational and drinking waters (Table 1-1). While several species of cyanobacteria are known to produce specific toxins (e.g., *Microcystis aeruginosa* can produce microcystin), not all blooms of a particular species produce toxins at all times. When produced, toxins typically are released into the surrounding environment after a bloom senesces (dies) or cells are ruptured, at which point the toxins can be taken in by higher organisms through their diet or by ingesting contaminated water.

In March 2020, the District began monitoring for the presence of cyanobacteria and cyanotoxins in the lake. In July of 2020, laboratory analysis confirmed the presence of elevated levels of the cyanotoxin microcystin in the water released from Lake Henshaw. Cyanobacteria and cyanotoxins have been measured in Lake Wohlford since July 2020.

1.2 Project Objectives

This iteration of the *Lake Henshaw and Lake Wohlford HABs Mitigation and Monitoring Plan* represents the first phase of a project to manage HABs and cyanotoxins in Lake Henshaw and Lake Wohlford for the District and Escondido. Phase I objectives include the following:

- Develop short-term solutions for mitigating or treating HABs;
- Screen potential long-term alternatives for preventing or minimizing HABs;
- Develop a HABs Mitigation and Monitoring Plan that includes a Water Quality Monitoring Plan; and
- Gather relevant data to inform future phases of the project.

Here, short-term is defined as occurring prior to 2024 and long-term is defined as occurring in the year 2024 or later.

Knowledge gained during Phase I will inform future phases of the project, including further ranking and prioritization of long-term prevention/minimization alternatives, selection of the preferred long-term alternative(s), and updates to the Water Quality Monitoring Plan and the HABs Management and Mitigation Plan, as needed.

Table 1-1. Cyanotoxin recreational and drinking water thresholds.

Criteria	Recreational Water					Drinking Water		
	CCHAB Trigger Levels for Human and Animal Health				WHO 2020 Provisional Guideline Value	OEHHA 2021 Recommended Notification Level [g,h]	WHO 2020 Provisional Guideline Value	WHO 2020 Provisional Reference Value
	No Advisory [a,b]	Caution (TIER 1) [a]	Warning (TIER 2) [a]	Danger (TIER 3) [a]				
Total microcystins [c]	< 0.8 µg/L	0.8 µg/L	6 µg/L	20 µg/L	24 µg/L [e]	ST NL: 0.03 µg/L for up to 3 months	1 µg/L (lifetime) 12 µg/L (short-term) [e]	–
						Acute NL: 3 µg/L for 1 day		
Anatoxin-a	Non-detect [d]	Detected [d]	20 µg/L	90 µg/L	60 µg/L [f]	ST NL: 4 µg/L for up to one month	–	30 µg/L (short-term) [f]
						Acute NL: 8 µg/L for 1 day		
Cylindrospermopsin	< 1 µg/L	1 µg/L	4 µg/L	17 µg/L	–	ST NL: 0.3 µg/L for up to 3 months	–	–
						Acute NL: 3 µg/L for 1 day		
Saxitoxins	–	–	–	–	–	–	–	–
						Acute NL: 3 µg/L for 1 day		
Cell density of potential toxin producers	< 4,000 cells/mL	4,000 cells/mL	–	–	–	–	–	–

Criteria	Recreational Water					Drinking Water		
	CCHAB Trigger Levels for Human and Animal Health				WHO 2020 Provisional Guideline Value	OEHHA 2021 Recommended Notification Level [g,h]	WHO 2020 Provisional Guideline Value	WHO 2020 Provisional Reference Value
	No Advisory [a,b]	Caution (TIER 1) [a]	Warning (TIER 2) [a]	Danger (TIER 3) [a]				
Site-specific indicator(s)	No site-specific indicators present	Discoloration, scum, algal mats, soupy or paint-like appearance. Suspected illness	–	–	–	–	–	–

[a] Action levels are met when one or more criteria are met.

[b] For de-posting, all criteria for no advisory must be met for a minimum of two weeks. General awareness sign may remain posted and healthy water habits are still recommended. Source: My Water Quality: California Harmful Algal Blooms (HABs) (California Water Quality Monitoring Council 2022b).

[c] Microcystins refers to the sum of all measured microcystin congeners. Source: My Water Quality: California Harmful Algal Blooms (HABs).

[d] Must use an analytical method that detects ≤ 1µg/L anatoxin-a. Source: My Water Quality: California Harmful Algal Blooms (HABs).

[e] Cyanobacterial toxins: microcystins. Background document for development of WHO Guidelines for drinking-water quality and Guidelines for safe recreational water environments. Geneva: World Health Organization; 2020 (WHO/HEP/ECH/WSH/2020.6) (WHO 2020b).

[f] Cyanobacterial toxins: anatoxin-a and analogues. Background document for development of WHO Guidelines for drinking-water quality and Guidelines for safe recreational water environments. Geneva: World Health Organization; 2020 (WHO/HEP/ECH/WSH/2020.1) (WHO 2020a).

[g] Zeise, L. 2021. Recommendations for interim notification levels for saxitoxins, microcystins, and cylindrospermopsin. OEHHA memorandum to D. Polhemus. Deputy Director, Division of Drinking Water State Water Resources Control Board.

[h] Zeise, L. 2022. Recommendations for acute notification levels for anatoxin-a, cylindrospermopsin, microcystins and saxitoxins. OEHHA memorandum to D. Polhemus. Deputy Director, Division of Drinking Water State Water Resources Control Board.

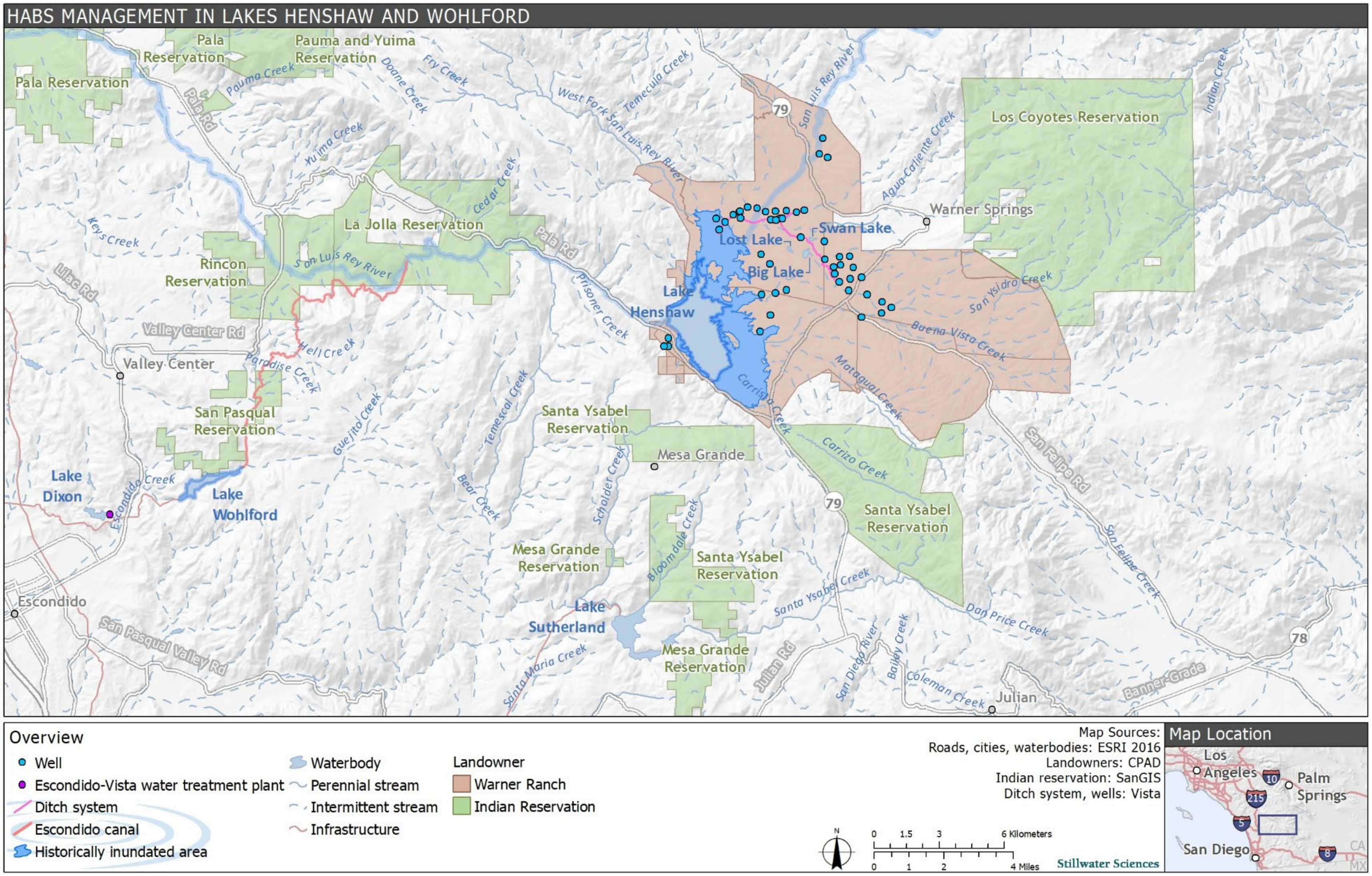

Figure 1-1. Project overview.

1.3 Watershed Context

Lake Henshaw is fed by surface runoff as well as pumped inflow from the District's Warner Ranch Wellfield. Water pumped from the wellfield flows by gravity into an approximately 1,800-ft unnamed stream reach before entering the ephemeral San Luis Rey River upstream of Lake Henshaw, and then flows for approximately one mile before entering the lake. Natural, intermittent inflows to Lake Henshaw occur from the San Luis Rey River, West Fork San Luis Rey River, Matagual Creek, Carrizo Creek, Carrista Creek, Buena Vista Creek, and Agua Caliente Creek. The 52,000 AF Lake Henshaw produces an average annual yield of 13,500 AF of water for entitlements held by the District, Escondido, and the Rincon Band of Indians. The La Jolla Band of Indians also benefits from the passive recreational use of water released from Lake Henshaw to the San Luis Rey River as it flows through a campground operated on their reservation lands. As described above, the Diversion Dam and the start of the Escondido Canal also are located within the La Jolla Reservation (Figure 1-1), and the canal supplies water to the 6,500 AF Lake Wohlford.

1.4 Lake Henshaw Bathymetry

At full pool (spillway elevation 2,690.59 feet [ft] above sea level), Lake Henshaw has a surface area of 2,256 acres, a volume of 51,832.2 AF, and a maximum depth of 49 ft (Figure 1-2). The reservoir is broad and shallow, sloping gently from its northern, eastern, and southern edges towards a narrow, deeper channel near the dam. The typical average depth is roughly 8 to 20 ft, as the reservoir is rarely at maximum capacity. In the period 2016 through 2021, the reservoir elevation has varied between 2,656 ft and 2,672 ft, which corresponds with approximately 490 and 1,450 acres, and approximately 1,910 and 17,720 AF respectively (Figure 1-2). The wind fetch[3] of Lake Henshaw is approximately 1 to 2 miles.

[3] The unobstructed distance along the lake water surface for wave development from wind.

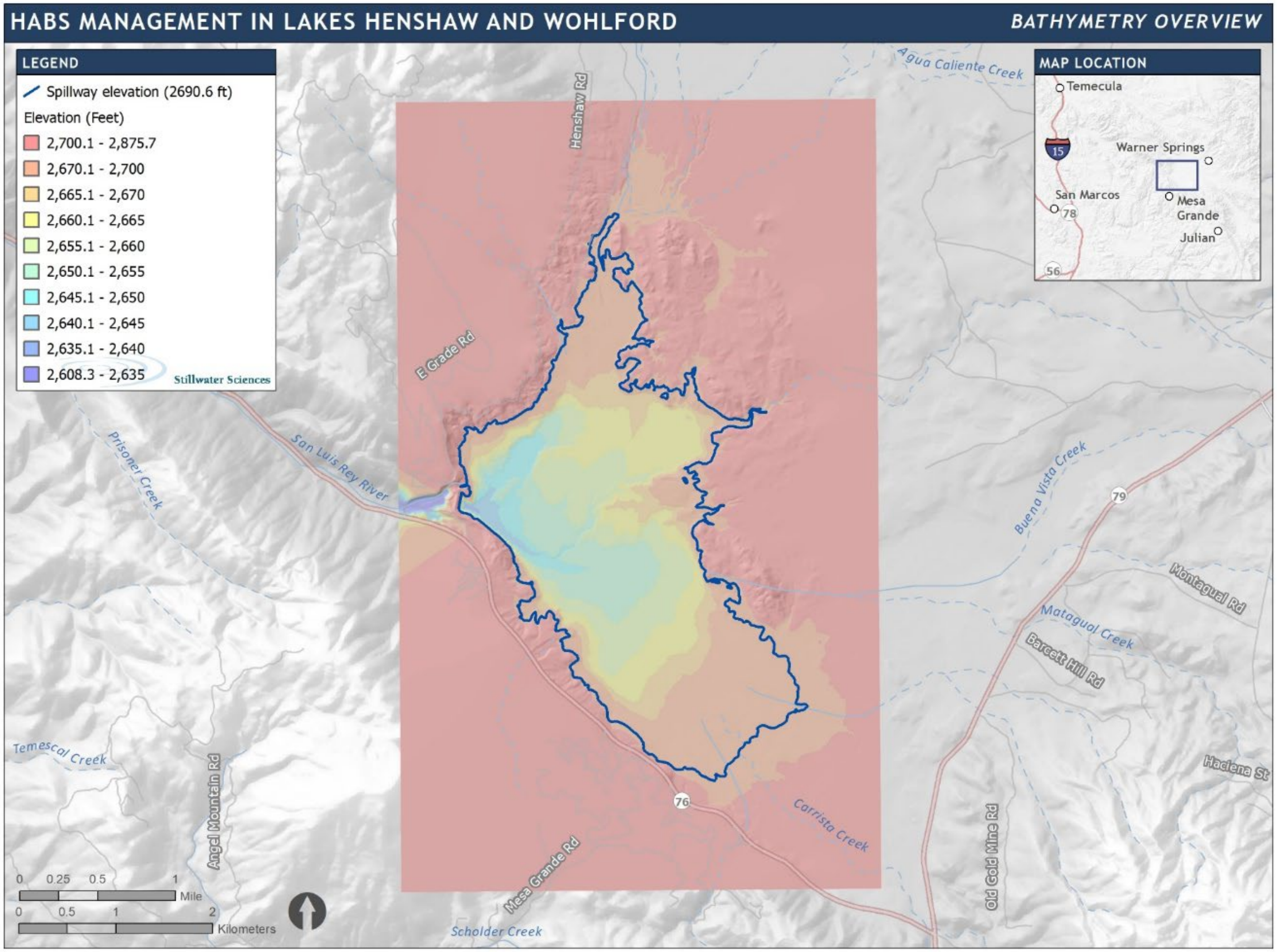

Figure 1-2. Lake Henshaw bathymetry.

2 LOCAL SYSTEM WATER QUALITY

2.1 Warner Ranch Wellfield

2.1.1 Background

In 1946, the District bought the Warner Ranch from San Diego County Water Company, which previously had built the Henshaw Dam, including overlying water rights for Warner Basin. In 1951, the District was experiencing a five-year drought that caused Lake Henshaw to reach a low of 200 AF. To provide a new source of water for the Local Water System, the District constructed 31 wells and associated facilities to extract Warner Basin groundwater and convey it via open ditches to Lake Henshaw (Vista Irrigation District 2019). Although the District upgraded the Warner Ranch wells in the 1980s, only 16 extraction wells are currently active and 52 wells are inactive (Figure 2-1). Some wells are artesian wells which flow under natural pressure without pumping; other wells require pumping. Groundwater, whether artesian or pumped, flows from the wellfield and into approximately six miles of ditches that convey the water by gravity flow through a dissipation structure (Figure 2-2) into an unnamed tributary of the San Luis Rey River and subsequently to Lake Henshaw. Recent renovations to the Warner Ranch have converted approximately ¾-miles of the existing ditches to pipelines (VID 2021). Future wellfield improvements may convert additional sections of ditch to pipelines.

The original Spanish land grants (circa 1800s) delineated the Warner Ranch and established the land use as cattle grazing, which continues to be the primary land use today. Most of the Warner Ranch continues to be available for grazing under grazing licenses issued by the District, and typical head counts vary between 1,500 and 2,500 head of cattle. Since the 1990s, fencing has prevented cattle from having direct access to Lake Henshaw, and the District has recently installed fencing to prevent cattle from having access to pumped well water. Between 1981 and 1986, the District allowed farming of potatoes on about 600 acres of land, and there has been no other agricultural activity on the Warner Ranch (Vista Irrigation District 2021).

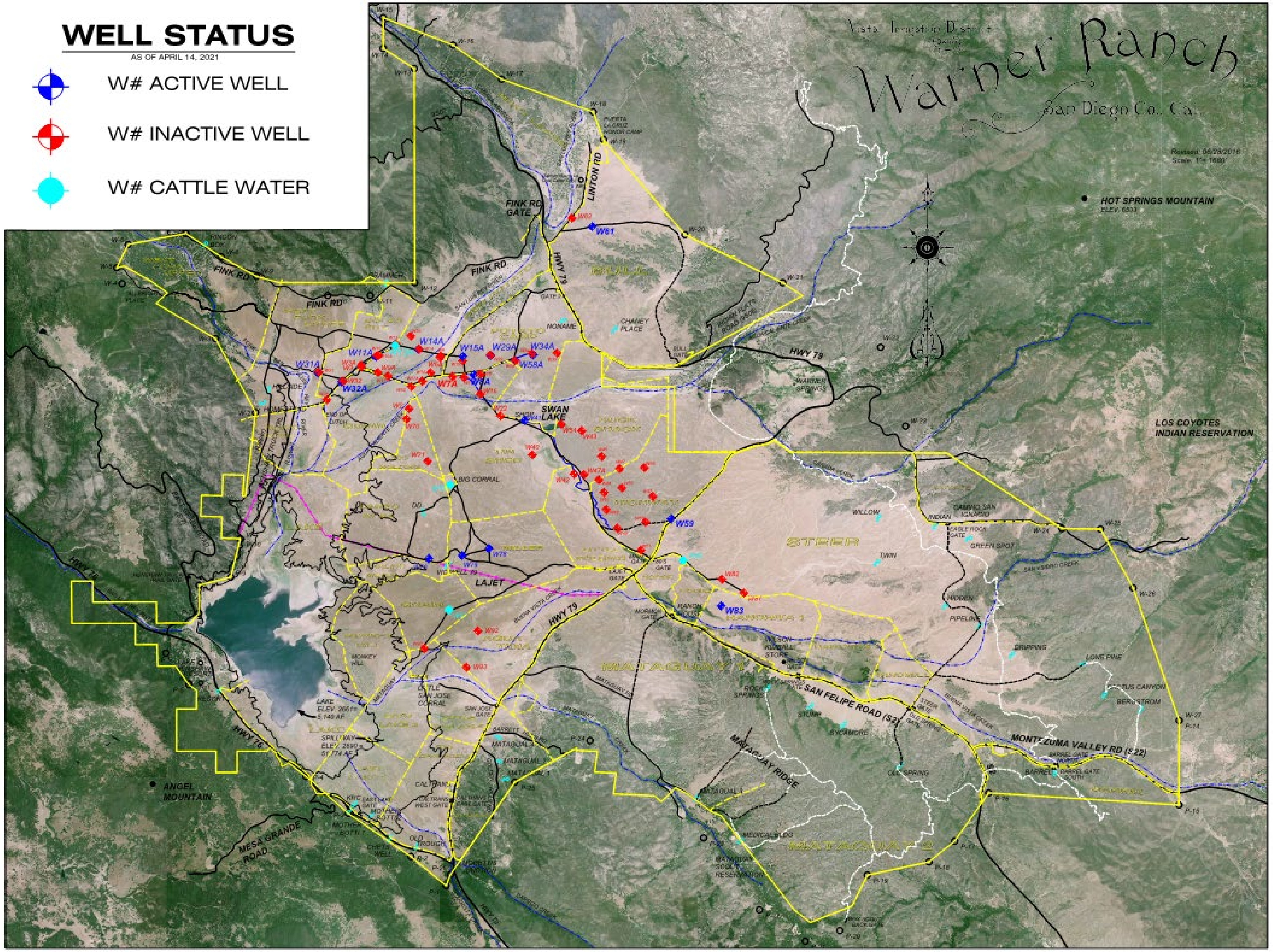

Figure 2-1. Warner Ranch Wellfield Extraction Wells.

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Figure 2-2. Warner Ranch Wellfield ditch system terminus.

2.1.2 Wellfield production

Warner Ranch Wellfield groundwater pumping operations are based on multiple factors within Lake Henshaw and the wellfield itself, including whether Lake Henshaw contains sufficient water to deliver annual entitlements and requests from downstream water users. The main consumptive users are Escondido and the District. Additionally, the Rincon Reservation is entitled to water for consumptive use each year under the Rincon Entitlement. The quantity of Rincon's entitlement fluctuates and is based on the previous two years of natural runoff occurring downstream of Henshaw Dam. Based on historical data, average flow to the Rincon Reservation is 2,900 AF/year (AFY) with a range of 1,100 to 4,300 AFY. However, the Rincon Entitlement may be curtailed in periods of extreme water shortage. The La Jolla Band is not a consumptive water user and has limited scheduling rights for recreational use (Smith 2021).

Other factors that the District considers when determining the need for pumping from Warner Ranch Wellfield include expected evaporative losses from Lake Henshaw, the minimum storage goal in Lake Henshaw of 2,500 AF on October 1, the condition of the wellfield, and the wellfield's groundwater elevation compared to historical years and the average well pump setting. During years when groundwater augmentation of natural runoff is needed, wellfield pumping begins in the spring with continuous operation until October 1 (i.e., start of water year) (Smith 2021).

The type of water year also influences wellfield production: dry year, normal year, wet year, or peak wet year. Pumping operations are minimal to nonexistent during a wet year or peak wet year due to high amounts of runoff supply to Lake Henshaw (Figure 2-3; see also Section 2.2.1). During normal and dry years, the lake requires significant pumping from the wellfield to maintain water levels and replenish evaporative losses.

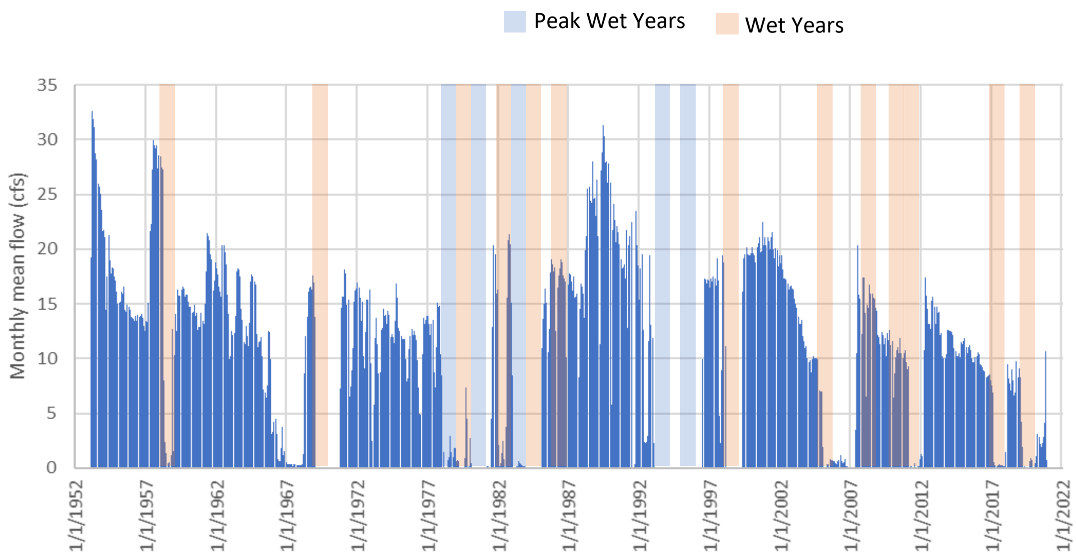

Figure 2-3. Warner Ranch Wellfield production with peak wet and wet years. Wet year (light orange) and peak wet year (light blue) categories reflect runoff experienced throughout San Diego County.

2.1.3 Wellfield water quality

2.1.3.1 Historical data

Groundwater sampling began in 1951 at the start of the wellfield operations with regular sampling conducted by the District for field water quality indicators, major and minor ions, nutrients, and trace elements. The United States Geological Survey Groundwater Ambient Monitoring and Assessment program (USGS GAMA) also sampled the Warner Ranch groundwater and analyzed samples for general parameters, major ions, trace elements, nutrients, dissolved organic carbon (DOC), microbiology, volatile organic compounds (VOCs), wastewater compounds, and contaminants of emerging concern (CECs). This analysis focuses on nitrate, orthophosphate, and iron in Warner Ranch groundwater as these constituents can affect lake primary productivity by stimulating algal and cyanobacterial growth. This analysis also includes manganese because it is an indicator of redox conditions in productive lakes and it can affect downstream operations at the EVWTP. Groundwater well pumping rates and chemical concentrations are used to calculate the monthly average loading of chemicals from the Warner Ranch Wellfield to Lake Henshaw by multiplying the monthly wellfield production rate by the monthly average chemical concentration.

Based on the period February 1951 to December 2019, nitrate, orthophosphate, iron, and manganese data from 1951 to 1999 are inconsistent with more recent (post-2000) trends, a pattern that is likely due to differing land use over time, lowering of the Lake Henshaw spillway (in 1981) and a concomitant decrease in reservoir size, and the upgrading of Warner Ranch groundwater wells. Although most of these activities occurred in the late 1970s to 1980s, this analysis assumes that it took over a decade to flush nutrients from the groundwater to result in concentrations and loadings from the wellfield to Lake Henshaw that align with current day operations as shown in Figure 2-4 through Figure 2-7.

Therefore, this analysis assumes that the most applicable water quality data for understanding existing conditions are from the period 2000 to 2019; however, the full period of record dataset is presented graphically for wellfield production and estimated chemical loadings to show trends over time (or lack thereof). When a sampling event occurred across multiple extraction wells, we present an average monthly loading (mass [kg] per month) from the wellfield into Lake Henshaw.

Table 2-1 presents summary chemical concentration data used in the calculations of estimated wellfield loading to Lake Henshaw, with details discussed in Sections 2.1.3.2 to 2.1.3.5.

Table 2-1. Warner Ranch Wellfield chemical concentration summary.

Constituent	Units	2000–Present					1950+	
		n	Average	Median	Minimum	Maximum	n	Peak Maximum
Nitrate (NO_3^-)	mg/L	256	1.1	0.9	0.0	3.2	452	28.0
Orthophosphate (PO_4^{3-})	mg/L	8	0.05	0.05	0.01	0.09	8	0.09
Iron (Fe)	µg/L	9	7.0	6.4	< 0.1	19.9	25	360
Manganese (Mn)	µg/L	9	0.6	0.4	0.1	2.99	24	50

Note: Groundwater well pumping rates and chemical concentrations were used to calculate monthly loadings of chemicals from the Warner Ranch Wellfield to Lake Henshaw by multiplying the monthly wellfield production rate by the monthly average chemical concentration.

2.1.3.2 Nitrate

Nitrate (NO_3^-) is a compound that forms naturally when nitrogen combines with oxygen (O_2) or ozone (O_3). Nitrate occurs naturally in surface and groundwaters at safe levels (typically up to 1 mg/L). Nitrate in excessive amounts can cause water quality problems. Sources of nitrate in groundwater include fertilizers, leaking septic systems, animal feeds or waste, and food or industrial wastes; note that the latter are likely irrelevant potential sources for the Warner Basin. The District has collected numerous nitrate (reported as nitrogen; NO_3-N) samples at the Warner Ranch Wellfield, more than any other constituent. In total, available data show 470 samples at 24 different well locations during sampling events between 1951 and 2019. During 10 sampling events, the extraction wells were not operating and thus these data are not included in below nitrate loading estimates.

Historically, data show an apparent increase in nitrate concentrations in the wellfield during the mid-1980s to early 1990s, possibly related to potato farming activities that were occurring during that period within Warner Ranch. Nitrogen contributed by other historical land uses such as agriculture, mining, or construction activities would likely have been flushed out of groundwater over the intervening decades. Nitrate concentrations during the years 2000 to 2019 range from non-detect[4] to 3.2 mg/L, with an average of 1.1 mg/L (Table 2-1), which is well below the water quality objective of 22 mg/L nitrate as nitrogen in the Water Quality Control Plan for the San Diego Basin. Combined with wellfield pumping rates, the summary nitrate concentration data result in a nitrate loading rate to Lake Henshaw that ranges from zero to a maximum of 1,800 kg NO_3-N/month, and a period average of 850 kg NO_3-N/month for the period 2000 to present (see also Appendix A).

[4] Non-detection limits for historical data are not known. Current nitrate non-detection limits are generally 0.1 mg/L.

Calculated nitrate loading trends over time using monthly wellfield pumping rates suggest a historical spike in loading to the lake during the period 1985–1992, with estimated loading rates increasing from below 2,000 kg NO_3-N/month to potentially as high as approximately 14,000 kg NO_3-N/month (Figure 2-4). From 2002 to present, decreasing nitrate loading trends from 2002–2005, 2008–2011, and 2013–2019 correspond to decreasing pumping rates, suggesting that nitrate concentrations in groundwater remain relatively constant and the primary factor controlling loading is the amount of pumping (Figure 2-5).

Note that the existing data provide insufficient information to estimate a nitrogen budget for Lake Henshaw for either the historical period or current conditions, nor do they shed light on the historical or current importance of nitrogen-fixing algae to the nitrogen budget in Lake Henshaw. Additionally, due to gaps in nitrate concentration data and inconsistencies in the number and location of wells sampled over time, the trends presented in Figure 2-4 should be considered suggestive rather than definitive. For the same reason, the period average monthly loading rate should not be translated to a period average annual loading rate using a factor of 12. Additional and regular sampling for nitrate in Warner Ranch groundwater wells, coupled with accurate pumping rates by well, would inform a future, more detailed analysis of nitrate loading potential to Lake Henshaw.

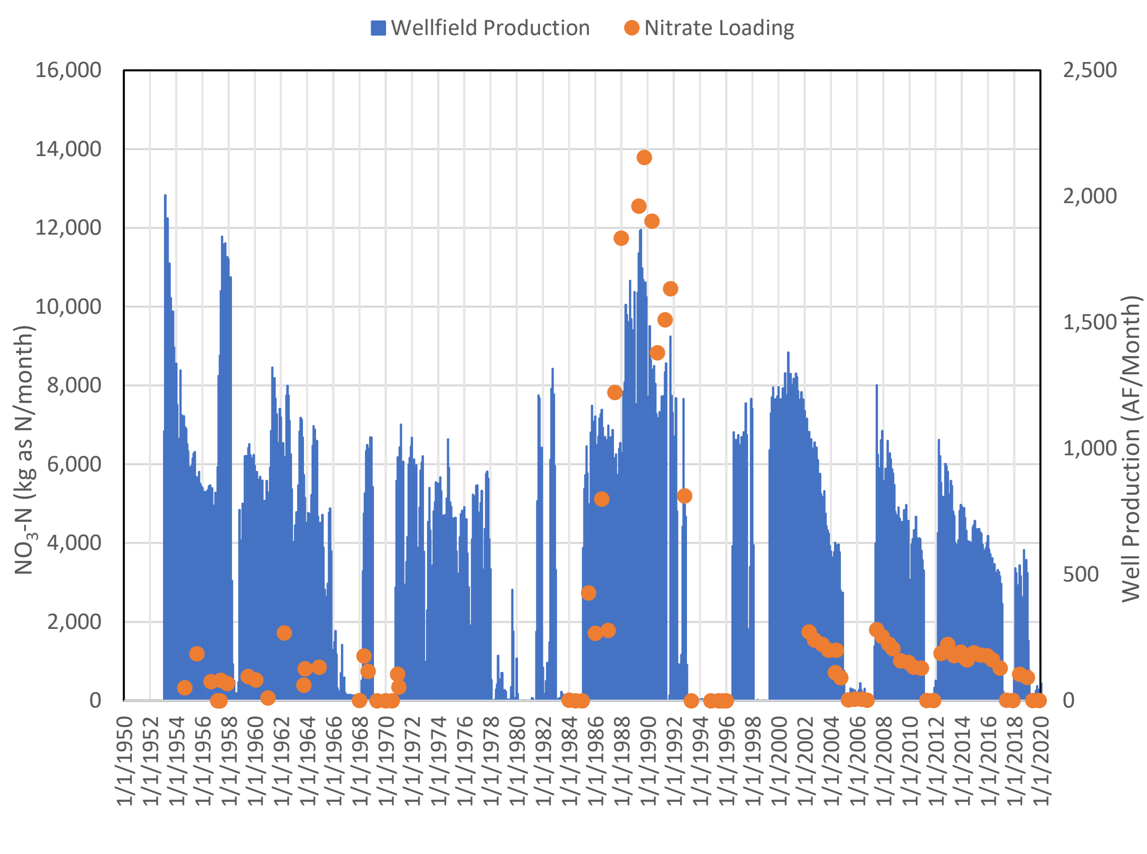

Figure 2-4. Warner Ranch Wellfield estimated nitrate as nitrogen (NO_3-N) loading to Lake Henshaw using measured nitrate concentrations and well production data.

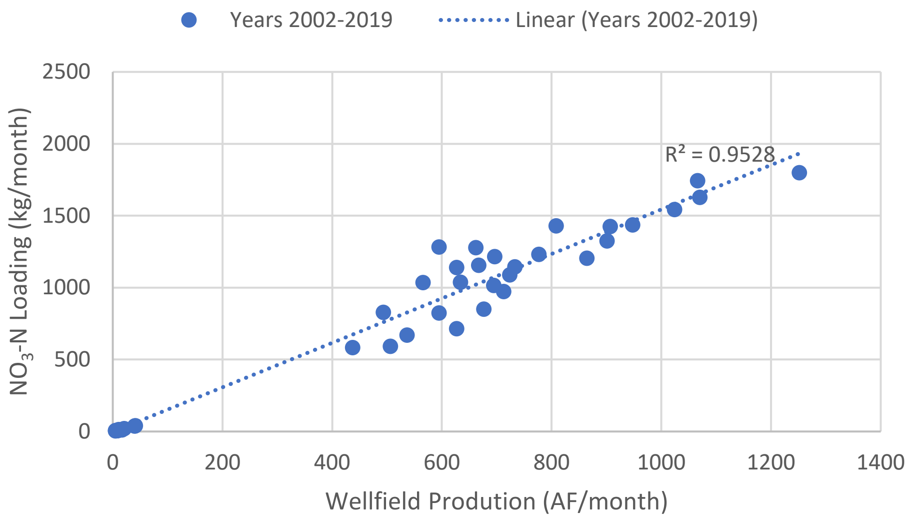

Figure 2-5. Correlation of Warner Ranch Wellfield nitrate as nitrogen (NO_3-N) loading to Lake Henshaw vs. wellfield production levels during production years 2002 through 2019.

2.1.3.3 Orthophosphate

Phosphorus is an important analyte in water quality since it is often the limiting nutrient for primary productivity in surface waters; however, it is usually not seen in high amounts in groundwater. Phosphorus may occur in groundwater under certain conditions, such as karst geology, glacial moraines, or septic tanks leaking into groundwater. Historically, potato farming activities that occurred between 1981 and 1986 within Warner Ranch may have been a source of phosphorus. Despite generally low housing density (MWH 2016), septic tanks may be a current source of elevated levels of phosphorous in groundwater in the Warner Basin since the local geology is not generally characterized by high background phosphorous. Orthophosphate (PO_4^{3-}) is a common dissolved form of phosphorus in water and consists of phosphorus bound to four oxygen atoms.

The District has sampled minimally for phosphorus at the Warner Ranch Wellfield. In total, the District took eight samples at four different well locations during sampling events in 2004, 2014, and 2019. During the years 2000 to 2019, phosphorous concentrations ranged from 0.01 mg/L to 0.09 mg/L with an average of 0.05 mg/L (Table 2-1). These summary data translate to estimated phosphorous loading rates to Lake Henshaw ranging from 0.32 to 68.4 kg PO_4-P/month, with a period average of 31.8 kg PO_4-P/month.

The District has added phosphorous to pumped groundwater by dosing sodium hexametaphosphate, an anti-scalant, into the wells. The District used an orthophosphate-containing anti-scalant from February 2015 to January 2021. Based on the amount of sodium hexametaphosphate purchased by the District each year and groundwater pumping rates during the 2015–2021 period, orthophosphate loading from the anti-scalant contributed approximately 13 kg PO_4-P/month to Lake Henshaw (Figure 2-6; see also Appendix A).

Given the extremely limited number of phosphorus concentration data in groundwater well samples over the period of record, it is not possible to identify an orthophosphate loading trend for Lake Henshaw due to groundwater pumping. For the same reason, the period average monthly loading rate should not be translated to a period average annual loading rate using a factor of 12. The sodium hexametaphosphate addition could account for additional phosphorus seen when discharged into Lake Henshaw but does not correlate with data seen from the groundwater. Additional and regular sampling for orthophosphate in Warner Ranch groundwater wells, coupled with accurate pumping rates by well, would inform a future analysis of orthophosphate loading potential to Lake Henshaw.

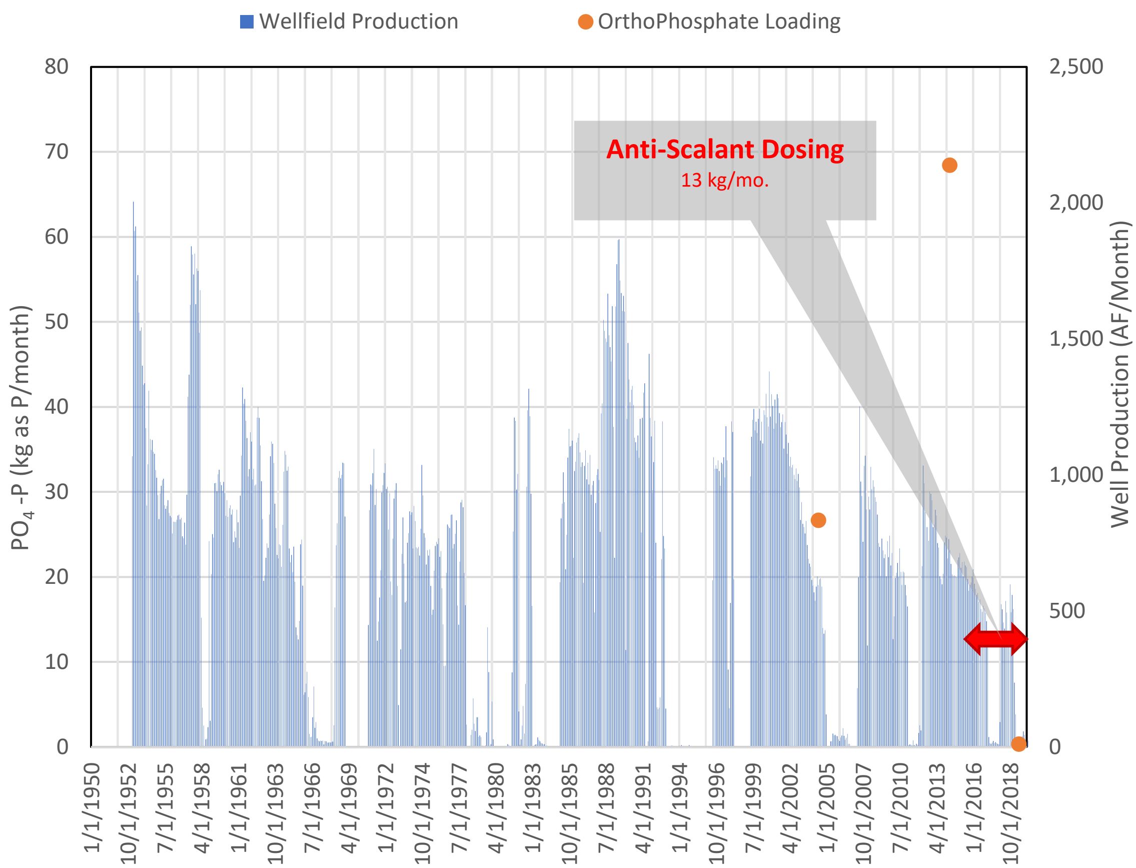

Figure 2-6. Warner Ranch Wellfield estimated orthophosphate as phosphorus (PO_4-P) loading to Lake Henshaw using measured orthophosphate concentrations (n=3) and well production data.

2.1.3.4 Iron

Iron (Fe) is a common metal found within the earth's crust, and it is found in surface and groundwater. Organisms need iron to metabolize nitrogen; iron availability may limit algae growth in water bodies; however, some algae are capable of producing organically bound iron when it is scarce (Horne and Goldman 1994).

Few iron samples exist for groundwater from the Warner Ranch Wellfield. In total, there were 25 iron-in-water samples collected between 1957 and 2019 from 15 different well locations. During three of the sampling events, the extraction wells were not in operation and hence loading rates into Lake Henshaw cannot be estimated from these data. From 2000 to 2019, the iron concentrations ranged from non-detect to 19.9 µg/L, with an average of 7.0 µg/L. These summary data indicate an iron loading rate to Lake Henshaw that ranges from zero to 173.8 kg Fe/month, with a period average of 15.1 kg Fe/month (Figure 7; see also Appendix A).

Given the sparse number of iron concentration data in well samples over the period of record, it is not possible to identify an iron loading trend for Lake Henshaw due to groundwater pumping. Additional and regular sampling for iron in Warner Ranch groundwater wells, coupled with accurate pumping rates by well, would inform a future analysis of iron loading potential to Lake Henshaw.

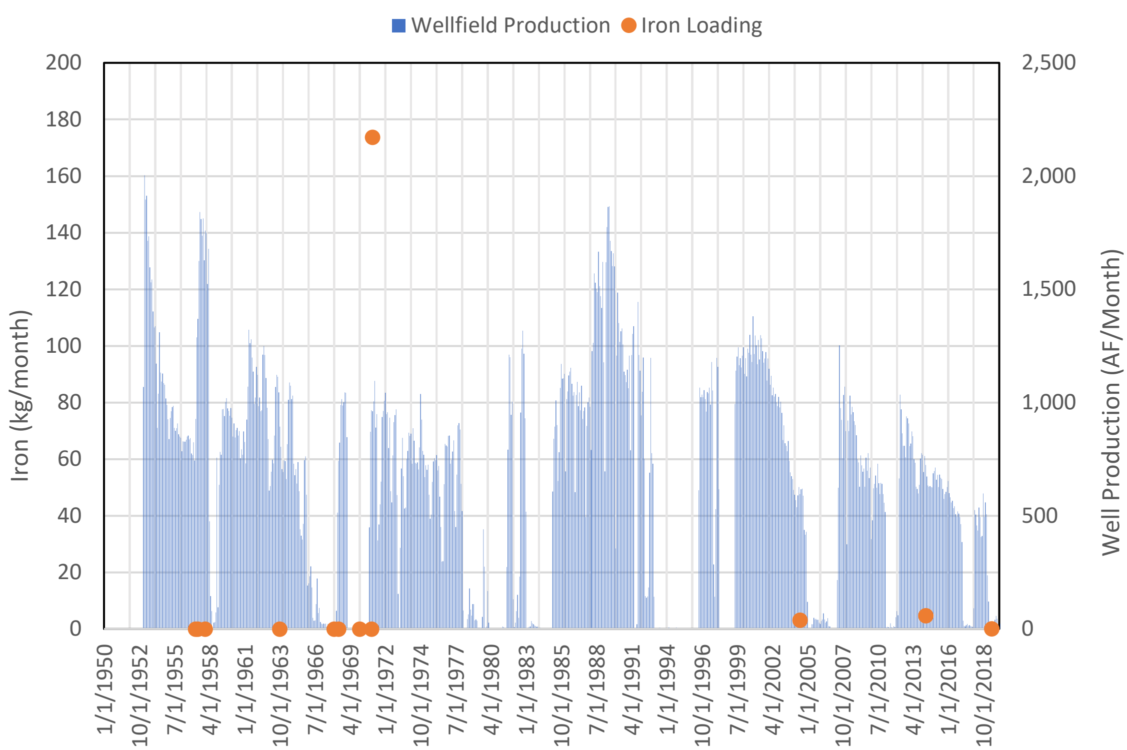

Figure 2-7. Warner Ranch Wellfield estimated iron loading to Lake Henshaw using measured iron concentrations (n=17) and well production data.

2.1.3.5 Manganese

Manganese (Mn) concentrations do not typically impact bodies of water; however, if the body of water becomes anaerobic (or anoxic) and manganese becomes soluble, it may affect downstream operations at the EVWTP. Raw water with high manganese concentrations that feed WTPs requires an oxidation process before filtering. High soluble manganese (i.e., the chemically reduced form of manganese [Mn^{+2}]) causes several adverse impacts on filtration: elevated chlorine (or other oxidant) demands, filter clogging, overly frequent filter backwashing, and frequent need to replace, rehabilitate, or clean filter media.

Similarly to iron, this analysis found few data for manganese samples collected at the Warner Ranch Wellfield. In total, 24 samples were taken between 1951 and 2019 at 14 different well locations. During three of the sampling events, the extraction wells were not in operation and hence loading rates into Lake Henshaw cannot be estimated from these data. From 2000 to 2019, the manganese concentrations ranged from 0.1 µg/L to 2.99 µg/L, with an average of 0.6 µg/L. These summary data indicate a manganese loading rate to Lake Henshaw that ranges from zero to 0.36 kg Mn/month, with a period average of 0.04 kg Mn/month (Figure 8; see also Appendix A).

Given the limited number of manganese concentration data in well samples over the period of record, it is not possible to identify a manganese loading trend for Lake Henshaw due to groundwater pumping. EVWTP personnel report no operational issues associated with soluble manganese and are currently dosing up to 0.5 ppm of potassium permanganate at the EVWTP as a manganese pre-oxidant. Additional and regular sampling for manganese in Warner Ranch groundwater wells, coupled with accurate pumping rates by well, would inform a future analysis of manganese loading potential to Lake Henshaw.

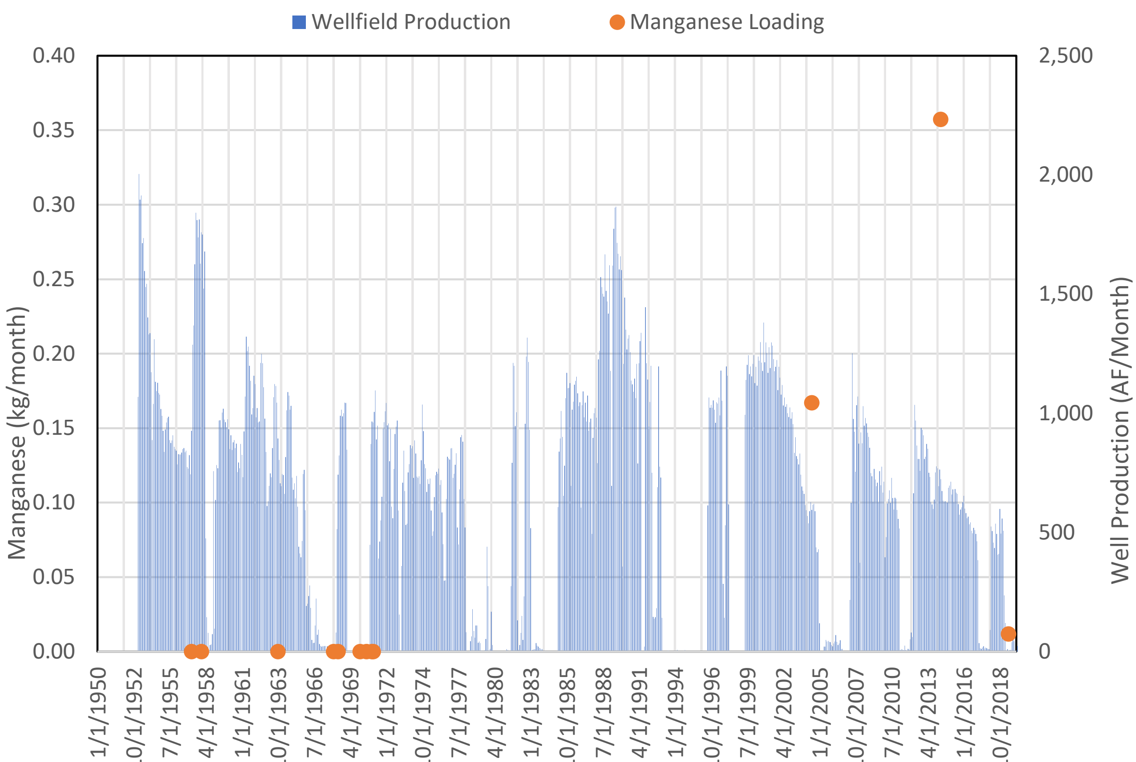

Figure 2-8. Warner Ranch Wellfield estimated manganese loading to Lake Henshaw using measured manganese concentrations (n=24) and well production data.

2.2 Lake Henshaw

2.2.1 Hydrology

Originally built in 1923, the Henshaw Dam was lowered in 1981 such that the historical capacity (200,000 AF) and inundated area of Lake Henshaw (approximately 3,834 acres) extended farther than the present day (Figure 1-1). Lake Henshaw did not spill until the dam was lowered in 1981, after which it has only spilled twice (Figure 2-9). Inflow to the lake is dependent on intermittent, seasonal surface runoff from several ephemeral creeks, the San Luis Rey River (Figure 1-1), and seasonally variable wellfield production (e.g., Figure 2-10, see also Section 2.1). Surface runoff from the 206 square mile (mi^2) catchment area tends to be greatest during January through April (e.g., Figure 2-10), with median monthly surface runoff for the period 1953–2020 at approximately 100 AF (Table 2-2). Although median monthly wellfield production is higher than the latter for the period of record, at 695 AF, wellfield production generally occurs at a low level and more continuously than the infrequent runoff events driven by precipitation (e.g., Figure 2-10). Releases from Lake Henshaw to the San Luis Rey River are dependent on water availability, downstream water needs, and water quality within the reservoir. Releases and evaporative losses from Lake Henshaw are generally highest in May through October (e.g., Figure 2-10), which is also the period of lowest rainfall (e.g., Figure 2-11). On average, monthly evaporative losses can comprise a large fraction of monthly wellfield production (Table 2-2).

USEPA estimated the mean hydraulic retention time of Lake Henshaw as 5.7 years (USEPA 1978), where annual storage during the analysis period was approximately 2,000 to 7,000 AF, which is generally similar to current conditions.

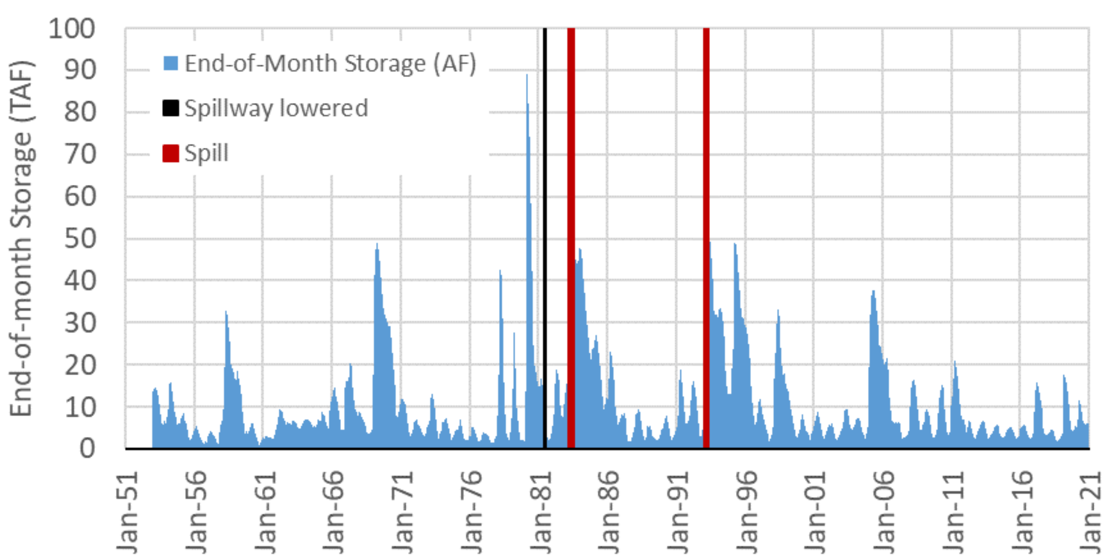

Figure 2-9. Lake Henshaw end-of-month storage in thousand acre feet (TAF) for the period of record (1951 to current), including spillway lowering in 1981 (black vertical line) and two spill events (red vertical lines) in March 1983 and February 1993. Note that while the dam was constructed in 1923, storage records began in 1951.

Table 2-2. Lake Henshaw monthly inflows and outflows for the period of record 1953-2020.

	Reservoir Inflows		Reservoir Outflows		
	Wellfield Production (AF/mo)	Computed Surface Runoff (AF/mo)	Evaporation (AF/mo)	Release (AF/mo)	Spill (AF/mo)
Minimum	0	0	3	0	0
Median	695	100	335	779	0
Maximum	2,004	64,376	1,560	17,460	15,703

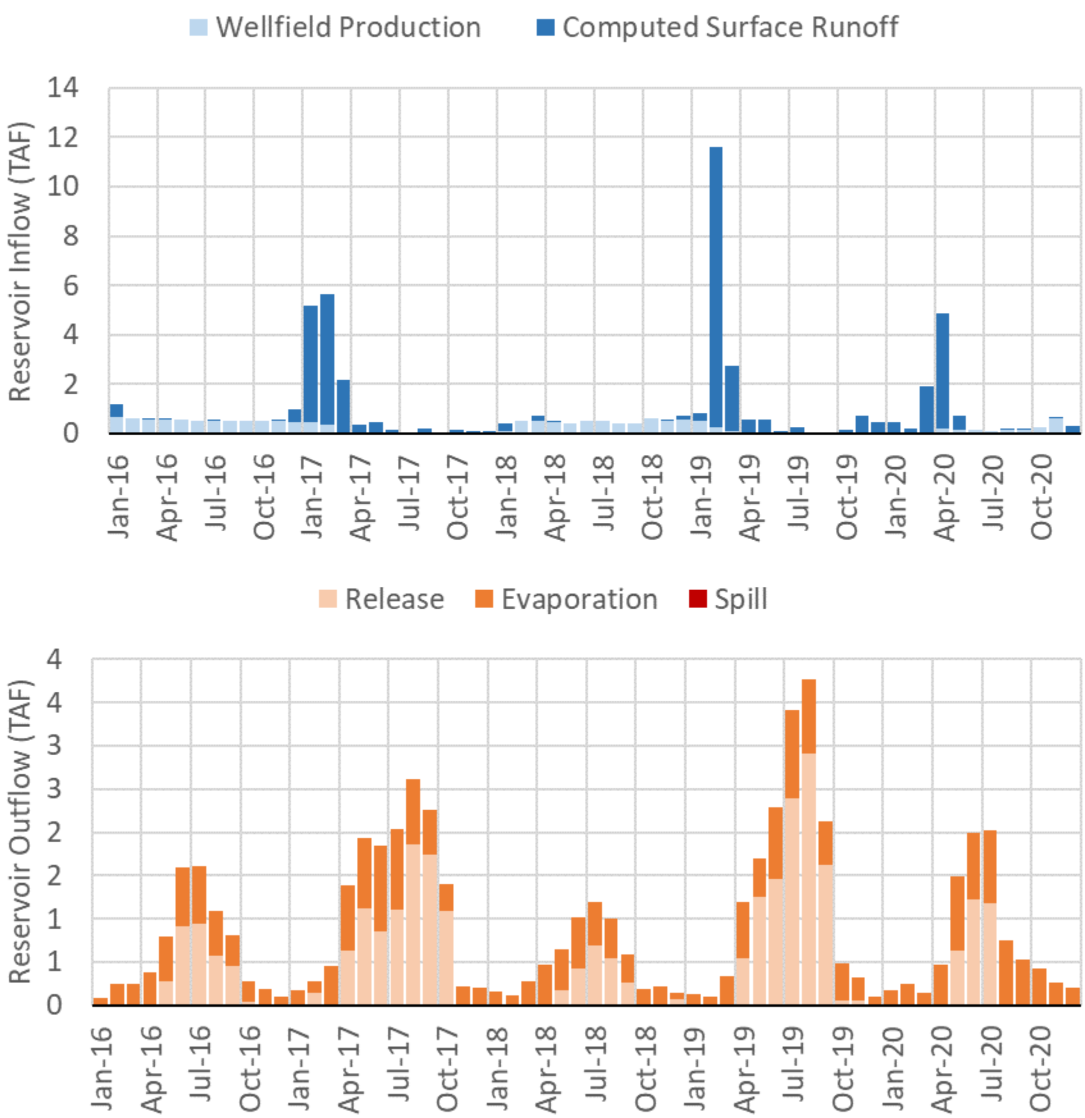

Figure 2-10. Lake Henshaw components of inflow (top) and outflow (bottom) in thousand acre feet (TAF) during 2016-2020.

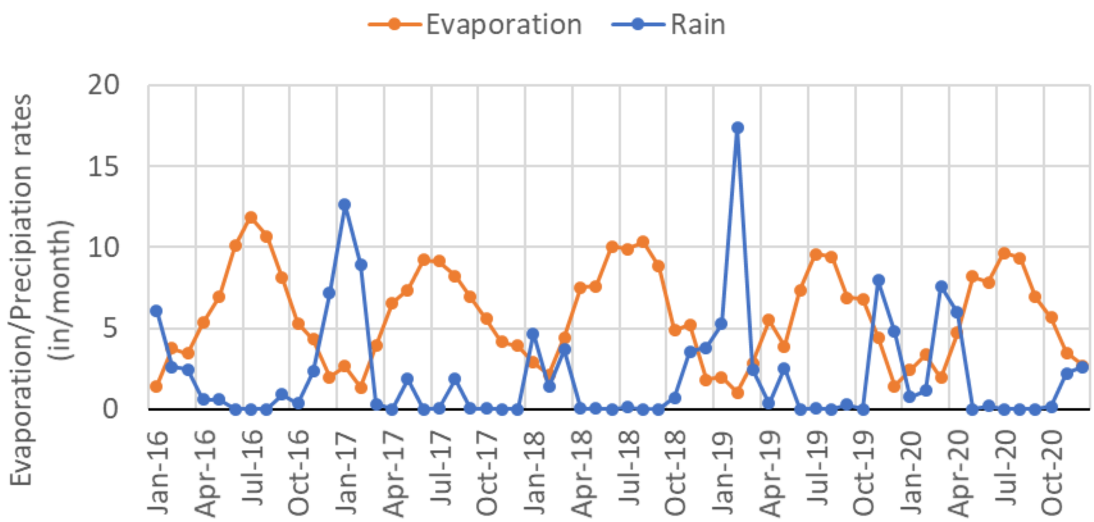

Figure 2-11. Lake Henshaw evaporation and precipitation amounts in inches per month during 2016-2020.

2.2.2 Water quality

Lake Henshaw was sampled three times in 1978 as part of the USEPA National Eutrophication Study. Water quality parameters (i.e., DO, conductivity, pH, alkalinity, nutrients, chlorophyll-a) were collected as integrated depth samples at two sites (one site near the dam and one site in the shallow southern portion of the lake) and composited across sites for analysis. Secchi depth was collected at both sites.

Lake water quality samples have been collected by the District approximately semi-annually (e.g., June and December) since 1984 for numerous minerals (i.e., magnesium, potassium, sodium), metals/metalloids (i.e., aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, calcium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, zinc), nutrients (nitrate), and other constituents (alkalinity, bicarbonate, carbonate, hardness, chloride, conductance, fluoride, hydroxide, pH, sulfate, surfactants, total dissolved solids, turbidity).

Recently, the District has collected additional water quality data at three sites within Lake Henshaw, and one site immediately downstream of the dam (Figure 2-12;Table 2-3). Beginning in March 2020, cyanotoxin (microcystin, anatoxin-a) samples were collected from at least two of these four sites. Nutrient (total phosphorus [TP], total nitrogen [TN]) samples were collected at two sites beginning in August 2020. The District eventually included both a surface and bottom water sample at the buoy line site (i.e., H-BL, H-BLS), and the frequency of monitoring for both cyanotoxins, TN, and TP increased from approximately biweekly to weekly. Beginning in January 2021, nutrient analyses were expanded beyond TN and TP to include orthophosphate (PO_4^{3-}), nitrate (NO_3^-), and ammonia (NH_4^+). Additional details are provided in Sections 2.2.2.1 and 2.2.2.5.

Table 2-3. Lake Henshaw additional water quality monitoring sites.

Site ID	Location	Latitude	Longitude
H-S	Southwestern shoreline at beach adjacent to fishing dock	33.23496°N	116.75617°W
H-FD	Southwestern shoreline at the in-water end of the fishing dock	33.23544°N	116.75568°W
H-BLS	Buoy line at dam in surface waters	33.23963°N	116.76174°W
H-BL	Buoy line at dam in bottom waters	33.23963°N	116.76174°W
H-R	Dam release channel approximately 10 ft upstream of flow measurement weir (point of release to San Luis Rey River)	33.23923°N	116.76594°W

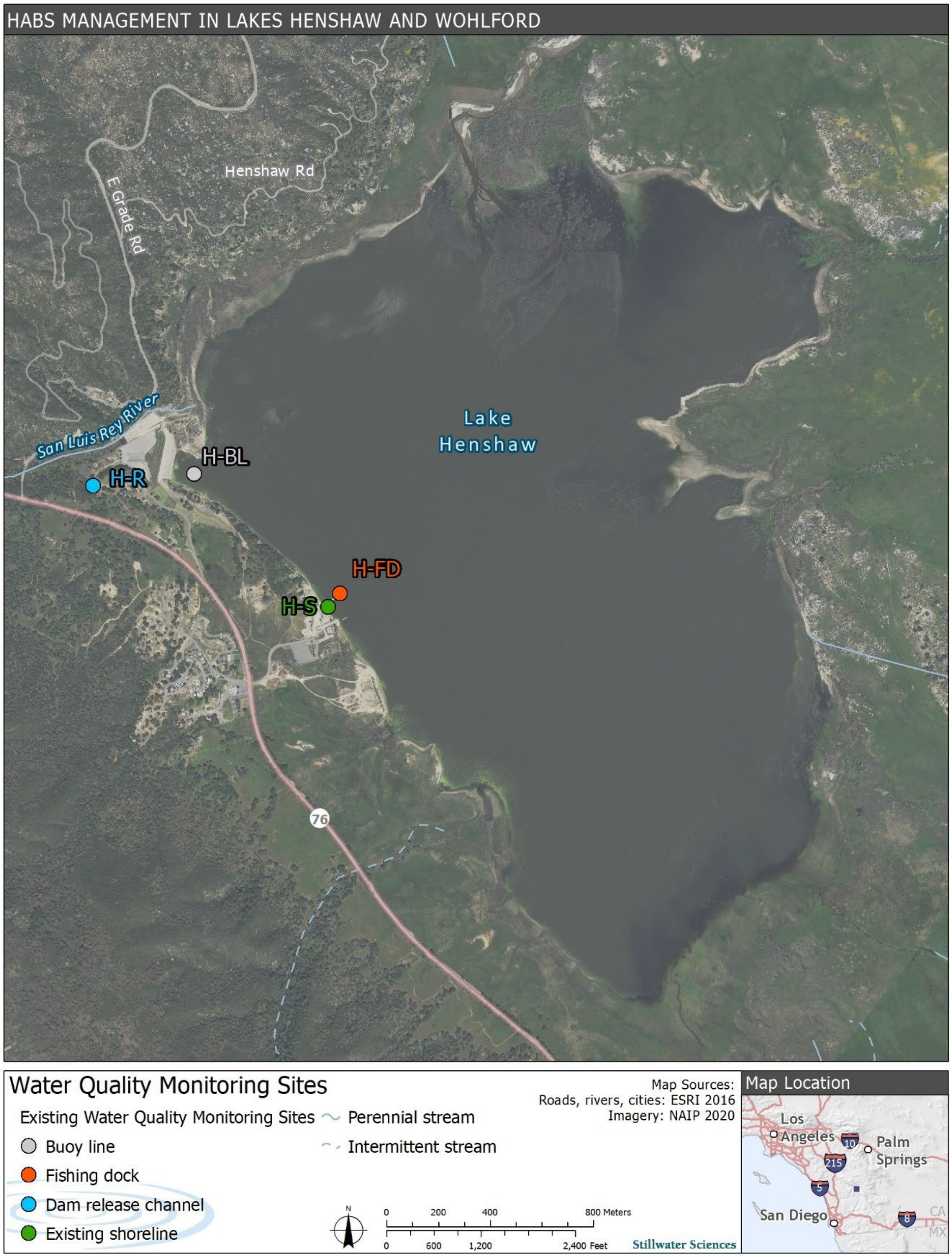

Figure 2-12. Lake Henshaw water quality monitoring sites 2020 to present.

2.2.2.1 Seasonal stratification

Available data characterizing water column thermal and chemical stratification in Lake Henshaw are limited to several dates in March 2022 (Appendix B). Seasonal and occasional daily mixing is expected in the reservoir, driven primarily by differential heating/cooling of surface waters and wind, but the timing and duration of such events are currently unknown. Although the 1978 USEPA National Eutrophication Study involved collection of DO data in Lake Henshaw, reported values are composites from two sites on three separate dates (ranging 5.2 to 8.7 mg/L) rather than vertical profiles that would elucidate chemical stratification patterns (USEPA 1978).

2.2.2.2 pH

Semi-annual pH data in Lake Henshaw range 7.0–9.1 standard units (s.u.) (Figure 2-13). Values can exceed the San Diego Regional Water Quality Control Board Wate Quality Control Plan (Basin Plan) instantaneous maximum water quality objective of 8.5 s.u. (RWQCB 2016) in either June or December, suggesting that relatively high levels of photosynthesis can occur during summer and winter months. There is no apparent pattern in pH with lake storage (Figure 2-13).

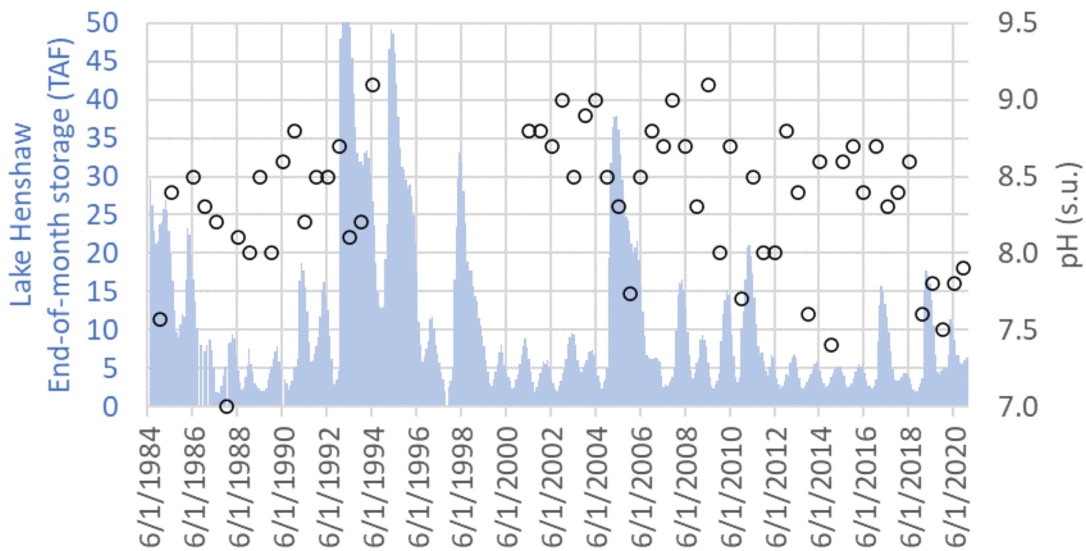

Figure 2-13. Monthly storage (thousand acre feet [TAF]; blue shading) and pH (s.u.; open circles) collected in June and December each year from Lake Henshaw during the period 1984-2020.

2.2.2.3 Nutrients

Nitrogen

Lake Henshaw TN concentrations reported in 1975 ranged 0.6–0.8 mg/L based on integrated-depth composite samples collected from one deep and one shallow open water site (Table 2-4). During August 2020 through January 2021, TN concentrations collected as surface grabs (which tend to exhibit higher concentrations than integrated depth samples) were highest at the shoreline site (H-S), peaking at just over 11 mg/L (Figure 2-14). Relatively high TN concentrations at the shoreline site in 2020 are likely due to accumulations of algal surface scums. TN concentrations during this period at the fishing dock (H-FD) and the buoy line surface site (H-BLS) tended to be lower, ranging from approximately 2 to 5 mg/L (Figure 2-14). The most recent data record at the

Lake Henshaw buoy line indicates that TN is highest in the summer to late summer months and is generally consistent between surface and bottom samples, with occasional differences, but no clear vertical pattern (Figure 2-15). The lack of difference between TN concentrations in surface and bottom waters on most dates suggests that algae in the water column in the deepest areas of the lake (i.e., > 15-ft deep) were distributed somewhat evenly, although *in situ* vertical profile data would be needed to confirm this was the case.

Table 2-4. Lake Henshaw nutrients collected during USEPA National Eutrophication Survey[1].

Sampling Date	Total N (mg/L)	Nitrite + Nitrate (mg/L)	Ammonia (mg/L)	TP (mg/L)	Ortho-phosphorus (mg/L)
3/7/1975	0.62	0.02	0.044	0.12	0.063
6/23/1975	0.788	0.022	0.032	0.262	0.218
11/13/1975	0.605	0.08	0.022	0.134	0.075

[1] Samples were collected as integrated depth samples at two sites (i.e., one site near the dam and one site in the shallow southern portion of the lake) and composited for each sampling date.

In 1975, nitrate ranged from 0.02 to 0.08 mg/L based on integrated-depth composite samples collected from one deep and one shallow open water site (Table 2-4). Regularly collected nitrate samples in the lake began to be collected by the District in 1984. Since this time, concentrations have generally remained below detection, although the analytical detection limit has ranged from < 0.04 to < 0.44 mg/L, where the latter limit would be considered moderately high nitrate. Higher nitrate levels in the late 1980s and 1990s (0.5 to 0.9 mg/L) may have been related to agriculture in the Lake Henshaw watershed, or increased groundwater pumping rates; they do not appear to be correlated with lake storage (Figure 2-16, see also Section 2.1.3.2). During more recent monitoring efforts at the Lake Henshaw buoy line, most nitrate samples have been at or below the laboratory quantification limit (QL) of 0.25 mg/L, with no apparent seasonal pattern or differences between surface and bottom waters (Figure 2-15), although nitrate levels of 0.25 mg/L would be considered more than sufficient for algae growth.

Results from recent laboratory chamber studies of Lake Henshaw sediments under oxic (DO > 2 mg/L), hypoxic (DO < 2 mg/L), and anoxic (DO = 0 mg/L) conditions indicate that high nitrate fluxes[5] occurred during oxic periods, where the likely mechanism was microbial nitrification[6] (Beutel 2021). Under anoxic and hypoxic conditions, the Lake Henshaw sediment chambers showed nitrate loss from the water column (i.e., negative nitrate fluxes), likely resulting from microbial denitrification[7]. The relatively high releases of nitrate from sediments in the laboratory chamber studies under oxic conditions (approximately 100 to 175 mg NO_3-N/m^2/d) and the negative fluxes under hypoxic conditions (approximately -30 to -90 mg NO_3-N/m^2/d) measured in Beutel (2021) are suggestive that both nitrification and denitrification are occurring in Lake Henshaw, potentially as coupled reactions in response to DO levels that vary throughout the day, season, and location in the lake. High algal productivity during the day combined with wind

[5] Transfer of nitrate from pore waters in the lake sediments to the overlying water column, in units of mass per unit area per unit time.
[6] Biological oxidation of ammonia (NH_4^+) to nitrite (NO_2^-) followed by the oxidation of nitrite to nitrate (NO_3^-) performed by small groups of autotrophic bacteria and archaea under oxic conditions.
[7] Biological conversion of nitrate (NO_3^-) to nitrogen gas (N_2) by heterotrophic bacteria under anoxic conditions.

mixing may help to oxygenate the sediment-water interface in more shallow areas of the lake, whereas deeper areas remain suboxic or anoxic in bottom waters and support denitrification. The overall effect may be to keep concentrations of nitrate generally low in both surface and bottom waters, although *in situ* vertical profile data including DO would be needed to confirm this is the case.

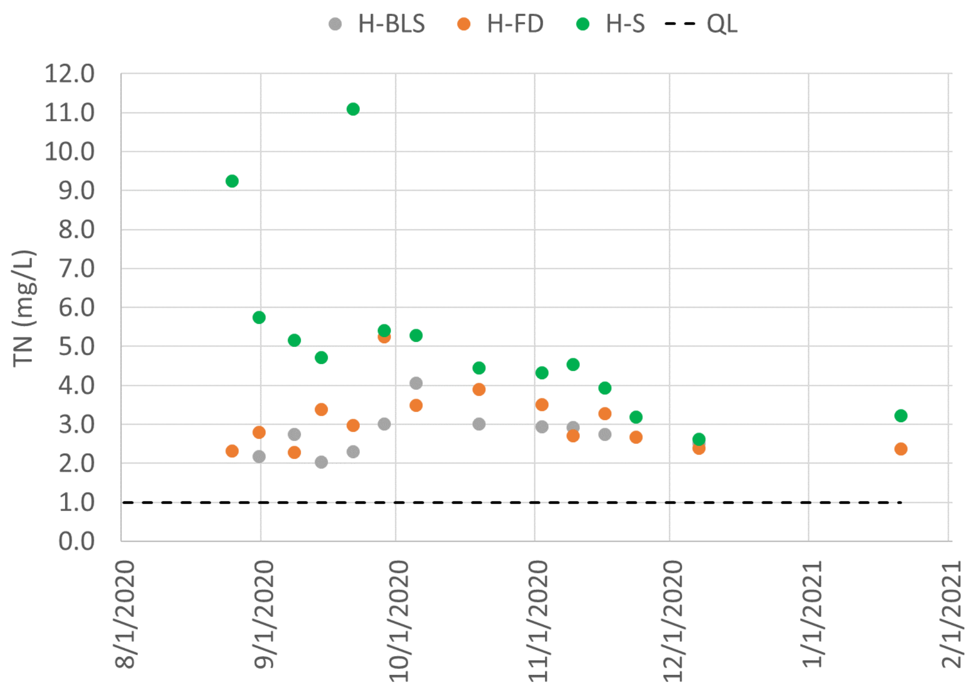

Figure 2-14. Total nitrogen (TN) at buoy line surface (H-BLS; grey), fishing dock (H-FD; orange), and shoreline (H-S; green) sites in Lake Henshaw, August 2020 through January 2021. Analytical laboratory quantification limit (QL) is shown as the dashed line.

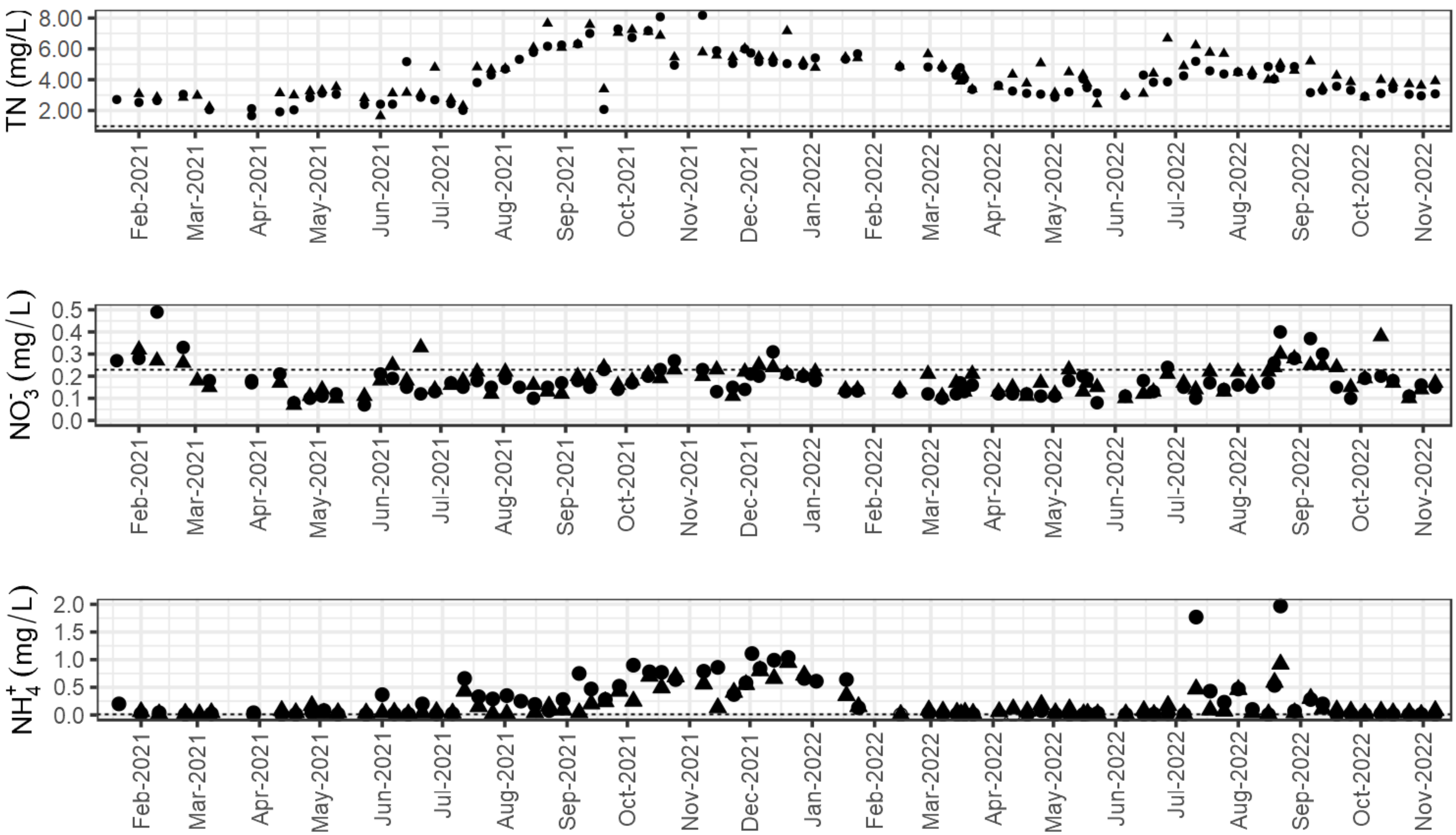

Figure 2-15. Total nitrogen (TN; top), nitrate (NO_3^-; middle), and ammonia (NH_4^+; bottom) at buoy line surface (H-BLS; triangle) and buoy line bottom (H=BL; circle) sites in Lake Henshaw, January 2021 through November 2022. Analytical laboratory quantification limits (QLs) are shown as dashed lines. All results below the quantification limits are considered estimated values.

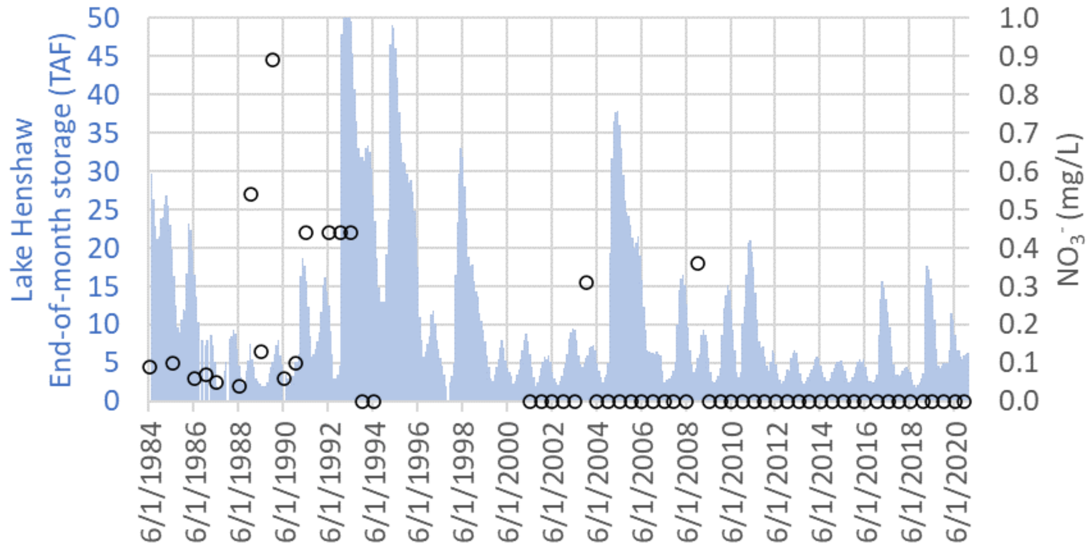

Figure 2-16. Monthly storage (thousand acre feet [TAF]; blue shading) and nitrate (NO_3^- [mg/L]; open circles) collected in June and December each year from Lake Henshaw during the period 1984–2020.

In 1975, ammonia ranged from 0.02 to 0.04 mg/L based on integrated-depth composite samples collected from one deep and one shallow open water site (Table 2-4). Recent (2021–2022) ammonia samples collected from the buoy line site (H-BL, H-BLS) indicate a distinct seasonal pattern, with concentrations at or below the laboratory quantification limit (QL) of 0.05 mg/L in winter and spring months and increases in concentrations in summer through fall months to a peak of approximately 1 mg/L (Figure 2-15). Incidences of elevated bottom water ammonia concentrations compared with surface water concentrations during warmer summer and fall months are suggestive that hypoxic (DO < 2 mg/L) or anoxic (DO=0 mg/L) conditions are occurring in bottom waters and/or lake sediments, which can release ammonia into the overlying water column due to mineralization of organic nitrogen in sediments. While there are no 2021 chlorophyll-a data to confirm assumptions regarding algal biomass in the water column, elevated concentrations of ammonia in surface and bottom waters in early winter 2021 could be the result of a substantial bloom die-off and release of ammonia and/or reduced uptake of this nutrient as the bloom diminished. TN during November and December 2021 remained relatively high during these months but was generally lower than that of September, suggesting that while the bloom strength was likely decreasing in late fall/early winter 2021, it was not completely gone by this point and may still have influenced ammonia concentrations (Figure 2-15). Elevated ammonia concentrations in late August 2022 occurred three to seven days following a copper-based algaecide treatment (0.02–1.97 mg/L), when chlorophyll-a concentrations decreased below 100 ug/L for the first time in two months, suggesting a rapid bloom die-off and ammonia release (Stillwater Sciences *in prep*).

As discussed for nitrate, results from recent laboratory chamber studies of Lake Henshaw sediments are suggestive of nutrient patterns that may be occurring in the lake itself. Beutel (2021) indicates that low ammonia releases from lake sediments occurred during oxic periods, but under anoxic conditions, the magnitude of ammonia release (150 to 264 mg NH_4-N/m^2/d normalized to 20 °C) was the largest ever measured in similar studies of California reservoirs. For comparison, ammonia release rates for other hypereutrophic lakes reported in Beutel (2021) range

30 to 60 mg NH_4-N/m^2/d normalized to 20ºC. While all Lake Henshaw sediment chambers released ammonia under anoxic conditions, the highest release rates were apparent in the chambers containing sediments collected in deep waters nearest the dam. Dam sediments also exhibited relatively high organic matter content that likely promoted ammonia release (Beutel 2021). High algal productivity during the day combined with wind mixing may help to oxygenate the sediment-water interface in more shallow areas of Lake Henshaw and reduce ammonia release from these sediments, whereas deeper sediments near the dam may be a "hot spot" of internal ammonia release. *In situ* vertical profile data including DO would be needed in Lake Henshaw to confirm this is the case.

Phosphorus

In 1975, TP ranged from 0.12 to 0.26 mg/L based on integrated-depth composite samples collected from one deep and one shallow open water site (Table 2-4). During August 2020 through January 2021, TP concentrations were highest at the shoreline site (H-S), peaking at just over 1.5 mg/L (Figure 2-17.). TP concentrations during this period at the open water fishing dock (H-FD) and the buoy line surface (H-BLS) sites tended to be lower, ranging from approximately 0.1 to 0.4 mg/L (Figure 2-17.). The most recent data record at the Lake Henshaw buoy line indicates that, like TN, TP was highest in the late summer months. While surface samples exhibited slightly greater TP earlier in the year, bottom samples often exhibited greater TP later in the year, although the differences are small (Figure 2-18). Peak TP at the buoy line in 2021 (approximately 0.8 mg/L) occurred in lake bottom waters in early November, suggesting a late-season cyanobacteria bloom that may have elevated shoreline surface concentrations of TP even higher at this time. Indeed, shoreline conditions indicated a substantial bloom lasting several weeks in Lake Henshaw in November through mid-December 2021 (Figure 2-19). Peak TP in 2022 occurred in July and August (0.55–0.58 mg/L), prior to a copper-based algaecide treatment, and then decreased following rapid bloom die-off (Stillwater Sciences *in prep*).

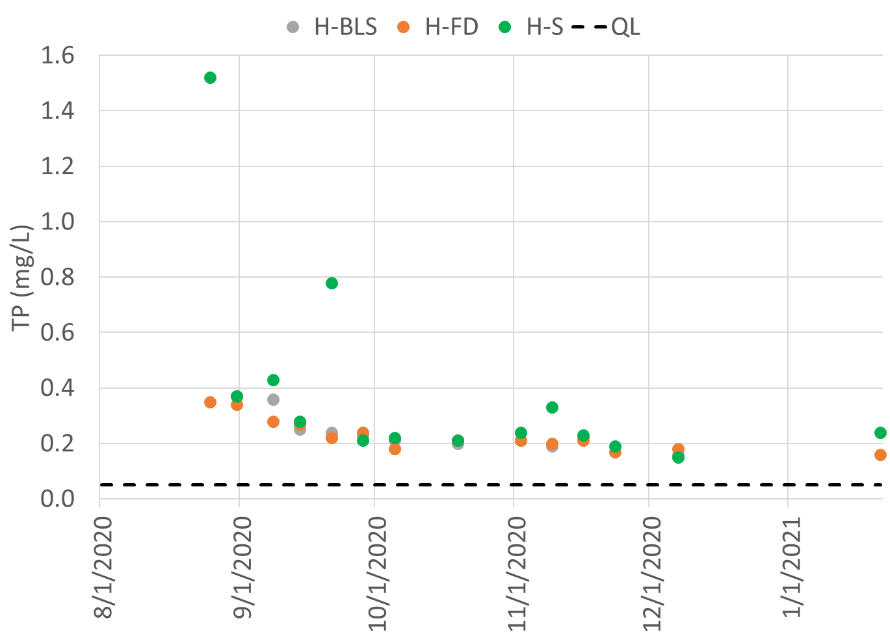

Figure 2-17. Total phosphorus (TP) at buoy line surface (H-BLS; grey), fishing dock (H-FD; orange), and shoreline (H-S; green) sites in Lake Henshaw, August 2020 through January 2021. Analytical laboratory quantification limit (QL) is shown as the dashed line.

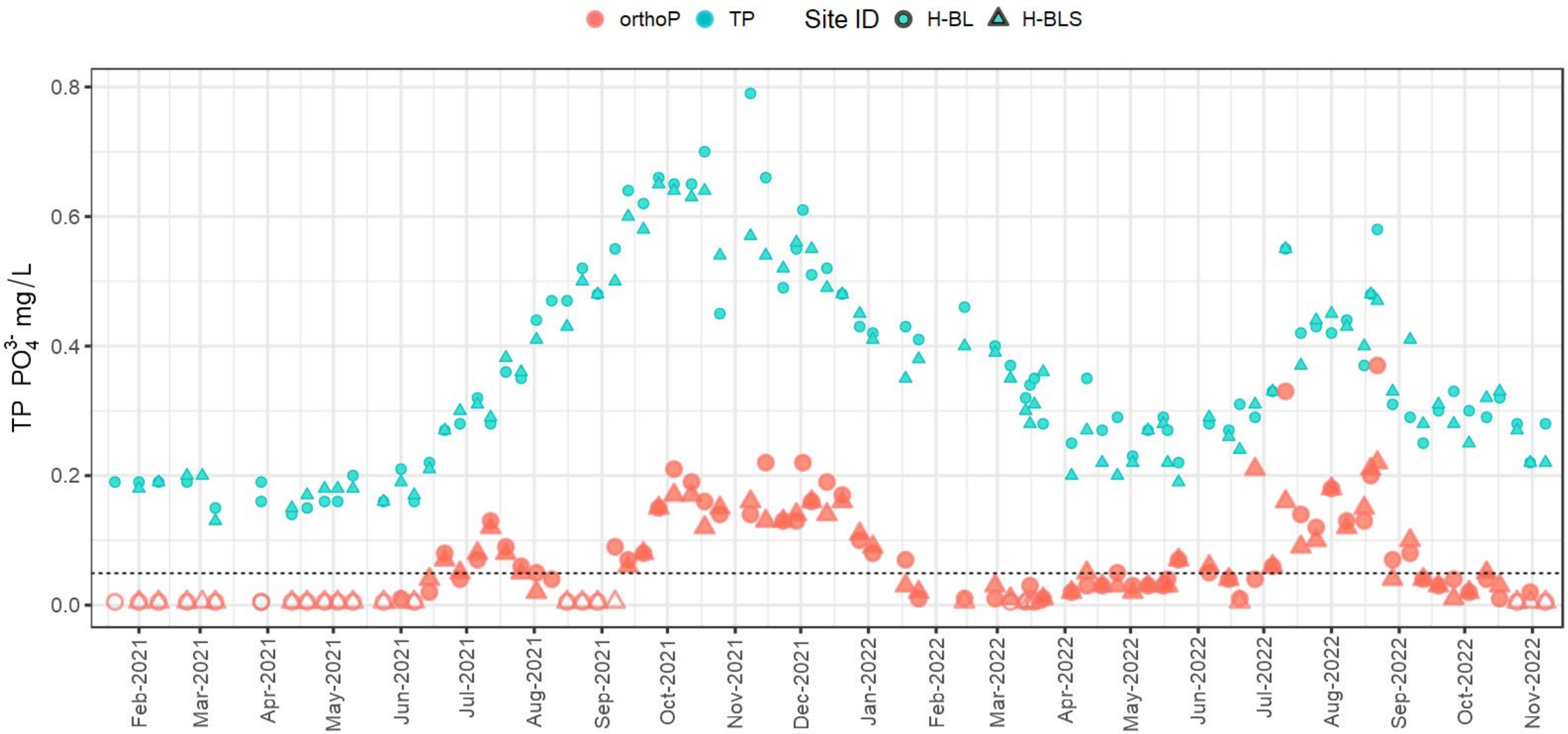

Figure 2-18. Total phosphorus (TP; pink) and orthophosphate (PO_4^{3-}; blue) at buoy line surface (H-BLS; triangle) and buoy line bottom (H=BL; circle) sites in Lake Henshaw, January 2021 through November 2022. Analytical laboratory quantification limits (QLs) are shown as dashed line. All results below the quantification limits are considered estimated values. Results reported as non-detects by the analytical laboratory are shown as 0.5 x instrument sensitivity limit (0.01 mg/L) and represented by open circles and triangles. No method detection limit (MDL) was reported by the analytical laboratory.

Figure 2-19. Cyanobacteria surface accumulation at the Lake Henshaw shoreline site (H-S), December 16, 2021.

In 1975, orthophosphate ranged from 0.06 to 0.22 mg/L based on integrated-depth composite samples collected from one deep and one shallow open water site (Table 2-4). Recent (2021–2022) orthophosphate samples collected from the buoy line site (H-BL, H-BLS) indicate a distinct seasonal pattern, with concentrations at or below the laboratory quantification limit (QL) of 0.05 mg/L in mid-winter and spring months and increases in concentrations to approximately 0.2 mg/L extending from October through December 2021 (Figure 2-18). The decrease in orthophosphate for roughly four weeks in August 2021 may have been due to intensive uptake of this nutrient during bloom growth and potential for a relatively short period of phosphorus limitation. Incidences of elevated bottom water orthophosphate concentrations compared with surface water concentrations during late September through early December 2021 are suggestive that hypoxic (DO < 2 mg/L) or anoxic (DO = 0 mg/L) conditions are occurring in bottom waters and/or lake sediments, which can release orthophosphate into the overlying water column due to iron reduction in the sediments. Orthophosphate concentrations ranged 0.09–0.18 mg/L before a copper-based algaecide treatment in July and August 2022, and 0.02–0.37 mg/L after treatment, suggesting release upon algal bloom die-off and/or release from anoxic bottom sediments following treatment (Stillwater Sciences *in prep*). Although the copper-based algaecide used in Lake Henshaw in August (SeClear) also contains an orthophosphate binding agent, it does not appear that the chemical binding agent was present in sufficient quantities to eliminate the observed increases in this nutrient following algaecide application (Stillwater Sciences *in prep*).

Seasonal orthophosphate patterns in the lake during 2021 and 2022 are generally consistent with results from recent laboratory chamber studies of Lake Henshaw sediments. Beutel (2021) indicates that all Lake Henshaw sediment chambers released orthophosphate under oxic, hypoxic, and anoxic conditions. However, the release rates were highest under anoxic conditions and second highest under hypoxic conditions. Orthophosphate release in the chambers corresponded with iron release, suggesting that the orthophosphate source was a combination of organic matter mineralization and reductive dissolution of iron oxides in the sediments (Beutel 2021). The magnitude of orthophosphate release measured in the Lake Henshaw sediment chambers (approximately 90 to 120 mg PO_4-P/m^2/d) was the largest ever measured in similar studies of California reservoirs. For comparison, reported orthophosphate releases for deeper eutrophic lakes ranged 5 to 20 mg PO_4-P/m^2/d (Nurnberg 1994 as cited in Beutel 2021) and anoxic orthophosphate releases in shallow and broad Lake Elsinore, located in southern California, were approximately 20 mg PO_4-P/m^2/d (Beutel 2000b). As orthophosphate release is mediated by biotic (e.g., manganese- and iron-reducing bacteria) and abiotic (i.e., sorption to metal oxides, precipitation of FeS) processes, the effects of water temperature on the release rates are more difficult to determine and Beutel (2021) does not normalize them to 20 °C. As noted for ammonia above, because sediment in the deeper portion of Lake Henshaw may experience more severe anoxia on a daily and seasonal basis, there may be “hot spots” of internal orthophosphate release (i.e., loading) from bottom sediments.

Iron

As discussed previously, iron availability may limit algae growth in water bodies; however, some algae are capable of producing organically bound iron when it is scarce (Horne and Goldman 1994). Iron concentrations in Warner Ranch Wellfield inputs to Lake Henshaw ranged < 0.1 µg/L to 20 µg/L with a median of 6 µg/L for the period 2000–2019 (n=9; Table 2-1). Across the longer record of lake water samples (i.e., 1984–2020), iron concentrations in the lake ranged from < 0.02 mg/L to 2 mg/L with a median of 0.5 mg/L (n=71), or one to two orders of magnitude greater than wellfield inputs. There was no discernable pattern in iron concentrations with respect to lake storage or season (i.e., June or December) (Figure 2-20).

In recent laboratory chamber studies of Lake Henshaw sediments, iron release from sediments into overlying waters was variable under oxic (DO > 2 mg/L), hypoxic (DO < 2 mg/L), and anoxic (DO = 0 mg/L) conditions. While iron concentrations increased in overlying waters under oxic and anoxic conditions from sediments collected near Henshaw Dam, concentrations increased under hypoxic conditions from sediments collected in the northern portion of the lake, and concentrations increased under hypoxic and anoxic conditions from sediments collected near the fishing dock (Beutel 2021). All other chambers exhibited decreasing iron concentrations. Further, iron release rates, whether positive (dissolution and flux of Fe^{2+} out of the sediments) or negative (precipitation and flux of Fe^{3+} into the sediments), generally exhibited high variability across replicates, indicating that iron and DO dynamics are complex in Lake Henshaw sediments. Orthophosphate release into the water column in the chambers corresponded with iron release, suggesting that iron oxides in Lake Henshaw sediments provide important binding sites for orthophosphate under oxic conditions, whereas under hypoxic and anoxic conditions both are released to the overlying water column. Iron release rates measured in the Lake Henshaw sediment chambers (approximately 0.9 to 3.3 mg Fe/m^2/d) were one to two orders of magnitude lower than release rates measured in two eutrophic northern California reservoirs (i.e., 20−40 mg/m^2/d in Lafayette Reservoir [Beutel 2000a] and 5−20 mg/m^2/d in San Pablo Reservoir [Stillwater Sciences and Brown and Caldwell 2016]), as well as those measured in sediment from the eutrophic Lake Hodges Reservoir (San Diego, CA) at 20−80 mg/m^2/d (Beutel 2000b).

Although sediment release rates from Lake Henshaw measured in the sediment chambers were relatively low, water column concentrations can be relatively high (> 1 mg/L; Figure 2-20), indicating that iron is unlikely to be limiting algal growth in Lake Henshaw.

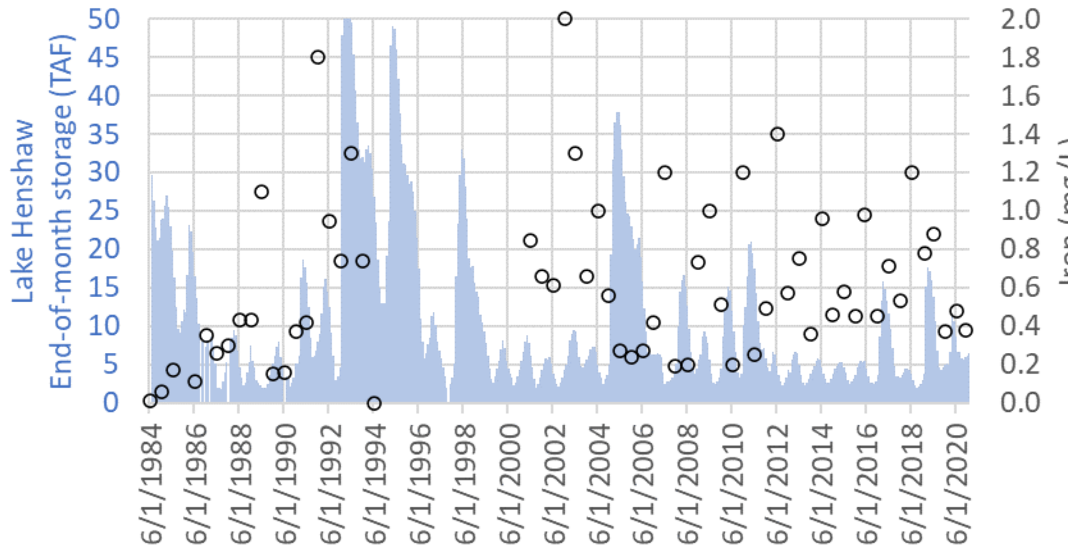

Figure 2-20. Monthly storage (thousand acre feet [TAF]; blue shading) and iron concentrations (mg/L; open circles) collected in June and December each year from Lake Henshaw during the period 1984–2020.

Manganese

While manganese is considered to be an algal micronutrient, it is almost always sufficiently available for growth in freshwater systems. Concentrations in Warner Ranch Wellfield inputs to Lake Henshaw ranged 0.1 µg/L to 3 µg/L with a median of 0.4 µg/L for the period 2000–2019 (n=9; Table 2-1). Across the longer record of lake water samples (i.e., 1984–2020), manganese concentrations in the lake were one to two orders of magnitude higher, ranging from < 10 µg/L to 30 µg/L with a median of 96 µg/L (n=70). Reduced (soluble) manganese (Mn^{+2}) can occur in lake sediments under anoxic (i.e., DO = 0 mg/L) conditions, which could diffuse into the water column and increase total water column manganese concentrations. However, the available data suggest no pattern between winter (December) and summer (June) manganese concentrations, nor between high and low water years, and thus it is not currently possible to determine whether water column stratification and subsequent anoxic conditions cause elevated concentrations of manganese in Lake Henshaw. As noted in Section 2.1.3.5, further downstream in the Local Water System, EVWTP personnel report no operational issues associated with soluble manganese.

In recent laboratory chamber studies of Lake Henshaw sediments, manganese concentrations in the water column decreased with time (i.e., flux from sediments at all sites was negative) under oxic (DO > 2 mg/L) conditions and increased with time (i.e., flux from sediments at all sites was positive) under hypoxic (DO < 2 mg/L) and anoxic (DO = 0 mg/L) conditions (Beutel 2021), which is consistent with typical redox and mineral oxide dissolution patterns for this metal. While there are limited data for comparison, manganese release rates measured in the Lake Henshaw sediment chambers (approximately 1.8 to 16 mg Mn/m²/d) were 3 to 30 times lower than release rates measured in a eutrophic northern California reservoir (i.e., 50−60 mg/m²/d in San Pablo Reservoir [Stillwater Sciences and Brown and Caldwell 2016]).

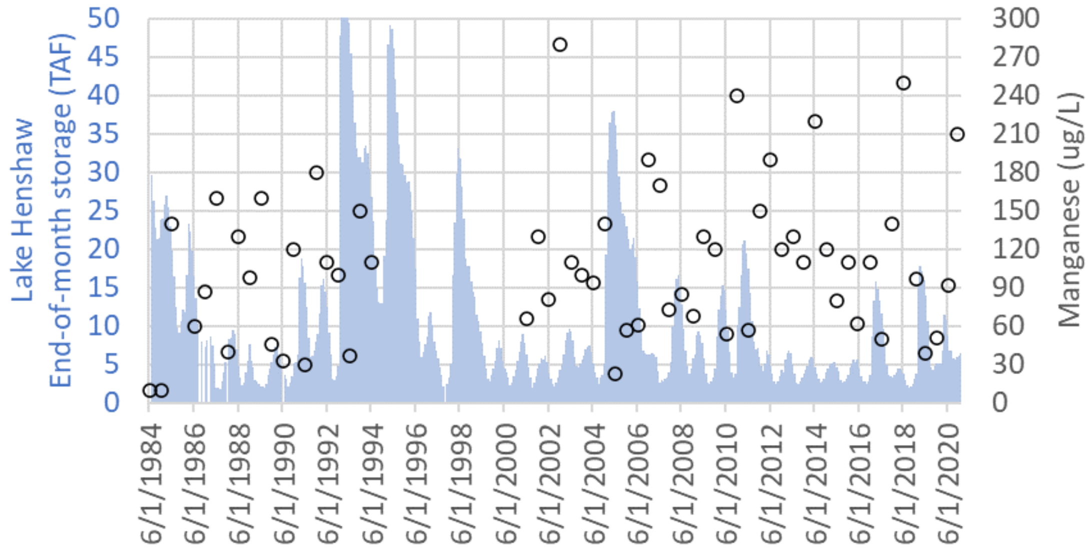

Figure 2-21. Monthly storage (thousand acre feet [TAF]; blue shading) and manganese concentrations (ug/L; open circles) collected in June and December each year from Lake Henshaw during the period 1984-2020.

2.2.2.4 Trophic status and N and P limitation

The Trophic State Index (TSI) developed by Carlson (1977) is one available tool for assessing Lake Henshaw's relative productivity. The trophic state of a lake or reservoir is based on overall system productivity and is a function of both physical features (e.g., latitude and elevation; ratio of watershed to waterbody areas; reservoir depth; hydraulic residence time), chemical features (e.g., nutrients, oxygen) and biological responses (e.g., primary productivity, zooplankton and fish assemblage food webs and biomass). The trophic status of Lake Henshaw was examined using the Carlson TSI for temperate lakes to validate assumptions about the relationships between measured physical and chemical parameters. TSI is a quantitative lake index ranging from 0 to 100 (Table 2-5).

Table 2-5. Carlson TSI associations with water quality.

TSI	Trophic Status and Water Quality Conditions
< 30	oligotrophic; clear water; high DO throughout the year in the entire hypolimnion[1]
30–40	oligotrophic to mesotrophic; clear water; possible periods of limited hypolimnetic anoxia
40–50	mesotrophic; moderately clear water; increasing chance of hypolimnetic anoxia in summer; fully supportive of all swimmable/aesthetic uses
50–60	mildly eutrophic; decreased transparency; anoxic hypolimnion; macrophyte problems; warm-water fisheries only; supportive of all swimmable/aesthetic uses but "threatened"
60–70	eutrophic; cyanobacteria dominance; scums possible; extensive macrophyte problems
70–80	hypereutrophic; heavy algal blooms possible throughout summer; dense macrophyte beds
> 80	algal scums; summer fish kills; few macrophytes due to algal shading

[1] Hypolimnion is defined as lake or reservoir bottom waters, located below the thermocline.

TSI values are based on the relationship between nutrients (as measured by total phosphorus), algal biomass (chlorophyll-*a*), and water clarity (Secchi disk depths). Three relationships are used to estimate the TSI:

TSI SD = 60 – 14.41 ln (Secchi depth in m) Equation 1
TSI Chl-*a* = 30.6 + 9.81 ln (chlorophyll-*a* in μg/L) Equation 2
TSI TP = 4.15 + 14.42 ln (TP in μg/L) Equation 3

Although these values may be averaged, differences between them can indicate other important water quality conditions. For example, the reservoir may be light- or nitrogen-limited instead of phosphorus-limited, and Secchi depth may be affected by transport of silt or clays rather than by algae.

While there are limited Secchi disk, chlorophyll-a, and TP data to characterize 1975 conditions in Lake Henshaw, the three samples that are available indicate that TSI values for Lake Henshaw corresponded to a eutrophic lake for Secchi disk (TSI 52 and 77) and chlorophyll-a (TSI 49–68), although TP values were relatively low and tended to predict a mesotrophic to mildly eutrophic lake (TSI 40–51; Table 2-6). Chlorophyll-a concentrations have increased substantially between historical sampling and more recent monitoring in 2022, with all but one sample collected from multiple sites throughout the lake (including samples at depth) during late February and March 2022, over 50 μg/L and the highest concentration at 153 μg/L (Appendix B). The more recent chlorophyll-a concentrations correspond to a TSI of 70–80, which indicates hypereutrophic conditions. The TSI for TP during the first half of 2021 was similar to 1975 values, whereas TP-based TP values increased to levels indicating eutrophic conditions in the second half of 2021 and 2022 (Figure 2-23).

To examine potential phosphorus and nitrogen limitations in Lake Henshaw, the mass ratio of N:P was estimated from the sum of total nitrogen (nitrate + nitrite + ammonium + organic nitrogen) divided by total phosphorus, for available data. Results were compared to empirically-derived ratios for freshwater algae, where a mass N:P ratio greater than 17 suggests phosphorus limitation, a ratio less than 10 suggests nitrogen limitation, and values between 10 and 17 suggest that either nutrient may be limiting (Forsberg and Ryding 1980, Hellström 1996). Mass ratios of total inorganic nitrogen (TIN = nitrate + nitrite [negligible] + ammonia) and orthophosphate (OP) also were calculated as an indication of the relative amounts of the more bioavailable fractions of nitrogen and phosphorus.

The existing data indicate that on a total nutrient basis, while nitrogen was strongly limiting total algal biomass in 1975 (Table 2-6), phosphorus is now more likely to be limiting. On a TIN:OP basis, nitrogen was strongly limiting in 1975 and is still more likely to be limiting under current conditions. While there are occasional periods of co-limitation apparent from the available data (i.e., mid-winter and mid-summer; Figure 2-22), given that TIN and OP are both relatively high during the primary growth season in Lake Henshaw, it is more likely that bioavailable nutrients are so plentiful that light becomes limiting to algae and cyanobacterial growth in the winter months, when solar insolation[8] is low, as well as in the summer months when solar isolation is high but thick surface accumulations of cyanobacteria can scatter and block light below the top few inches (in) of the water column.

[8] Solar insolation is the total amount of solar radiation energy received on a given surface area over a given time interval.

Overall, the Carlson TSI and N:P ratios indicate that chlorophyll-a concentrations have increased relative to historical conditions (although data are limited), as have nutrient concentrations, and the lake has shifted from a nitrogen-limited system to one that is phosphorus-limited on a total nutrient basis. On a bioavailable nutrient basis, nitrogen is relatively less available than phosphorus, although both are present at high levels throughout the year.

Table 2-6. Lake Henshaw nutrient, chlorophyll-a, and Secchi depth data collected during USEPA National Eutrophication Survey and used to calculate Carlson TSI associations with water quality.

Date	TP (mg/L)	Ortho-P (mg/L)	Nitrite+ nitrate (mg/L)	Ammonia (mg/L)	TIN [1] (mg/L)	TN (mg/L)	Chl-a (ug/L)	Secchi depth (m)	TN:TP	TIN:OP	TSI:SD [2]	TSI:Chl-a [3]	TSI:TP [4]
3/7/1975	0.12	0.063	0.02	0.044	0.064	0.62	29.3	0.3	5.2	1.0	77	64	40
6/23/1975	0.262	0.218	0.022	0.032	0.053	0.788	6.6	1.7	3.0	0.2	52	49	51
11/13/1975	0.134	0.075	0.08	0.022	0.102	0.605	44.4	-	4.5	1.4	-	68	42

[1] TIN = nitrate [NO_3^-] + nitrite [NO_2^-; negligible] + ammonia [NH_4^+])
[2] Calculated using Equation 1.
[3] Calculated using Equation 2.
[4] Calculated using Equation 3.

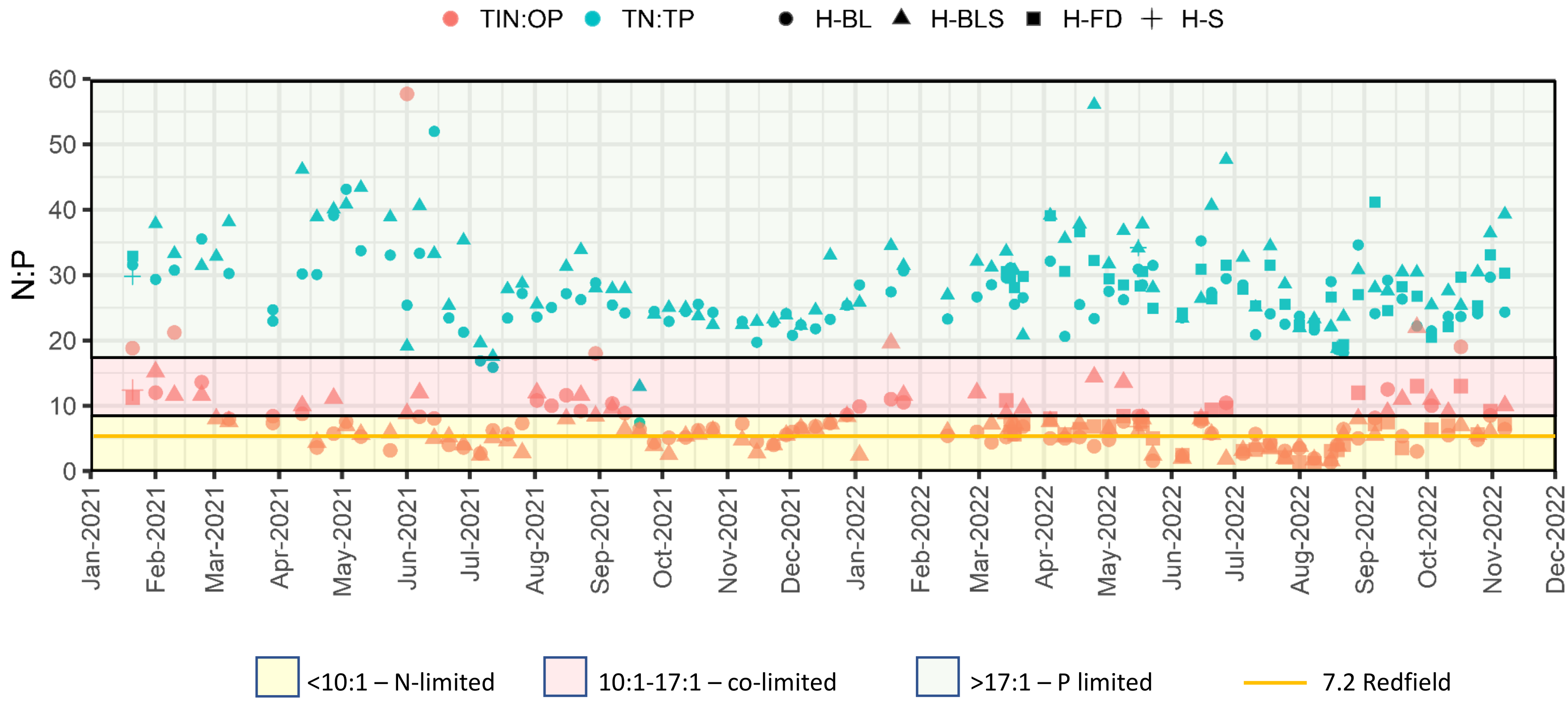

Figure 2-22. Nitrogen to phosphorus ratio (N:P) for total inorganic nitrogen to orthophosphate (TIN:OP where TIN = NH_4^++ NO_3^- [NO_2^- is negligible] and OP = PO_4^{3-}) (pink) and total nitrogen to total phosphorus (TN:TP) (blue) in Lake Henshaw, January 2021 through November 2022. Potential nutrient limitation regions for algae and cyanobacteria are shown by shading.

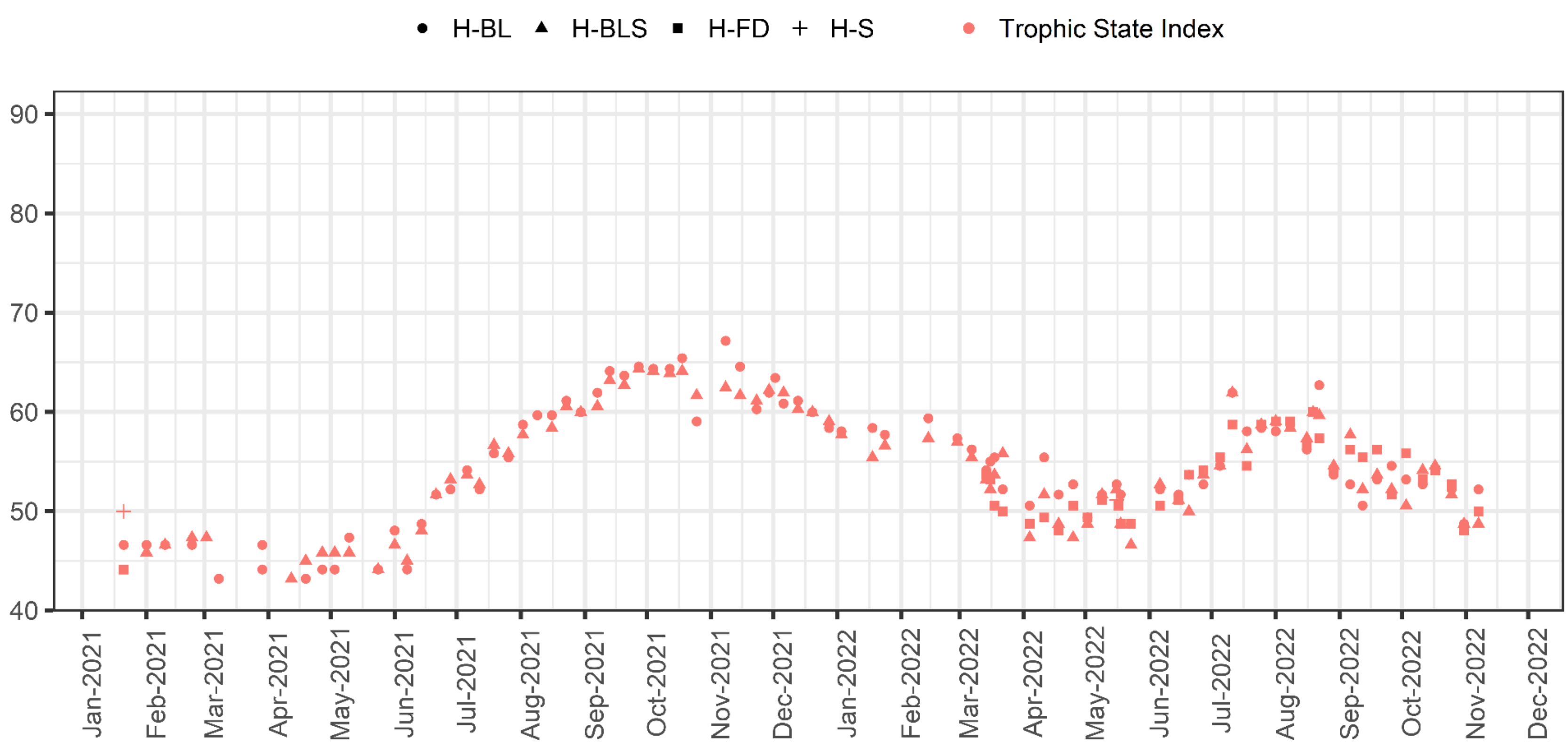

Figure 2-23. Carlson Trophic Status Index (TSI) based on TP concentrations for Lake Henshaw for the period January 2021 through November 2022.

2.2.2.5 Cyanotoxins

Since March 2020, microcystin samples have been collected from Lake Henshaw at the shoreline (H-S), fishing dock (H-FD), and buoy line (H-BL) sites and, when the lake is releasing water, at the lake outlet (H-R; Figure 2-12). Anatoxin-a samples have been collected in Lake Henshaw at two to three of these sites since August 2020 (the shoreline site was discontinued in March 2021 since anatoxin-a is not typically associated with shoreline surface accumulations).

Both microcystin and anatoxin-a have been detected in Lake Henshaw. In 2020, microcystin concentrations in the reservoir peaked above 1,000 µg/L in August and did not consistently decline below the CCHAB caution threshold of 0.8 µg/L until March 2021 (Figure 2-24). Peak microcystin concentrations occurred at the shoreline site (H-S) in 2020 and corresponded to the warmest water temperatures during that year (> 80 ºF). Microcystin concentrations at the fishing dock (H-FD) and buoy line bottom (H-BL) sites during July through November 2020 were generally one to two orders of magnitude lower than concentrations at the shoreline, although concentrations at these two open water sites also exceeded the CCHAB warning (6.0 µg/L) and danger (20.0 µg/L) thresholds for multiple weeks in 2020. In 2021, concentrations decreased during winter and spring, remaining below the CCHAB caution threshold (0.8 µg/L) even as water temperatures climbed over 80 ºF. The peak microcystin concentration in 2021 occurred in fall, again at the shoreline site (H-S), although at 11 µg/L it was two orders of magnitude lower than the peak of 2020. In general, 2021 microcystin concentrations were more similar across the three monitoring sites (H-S, H-FD, H-BL). Microcystin concentrations decreased to below the CCHAB caution threshold in December 2021 and remained at or below the threshold (with a few exceptions) through summer 2022. Microcystin concentrations were generally decreasing prior to a copper-based algaecide treatment in August 2022, and further decreased after treatment (Figure 2-24). Microcystin concentrations did not exhibit a peak within 10–13 days following treatment with the copper-based algaecide, in contrast to patterns observed in March and May when concentrations peaked at multiple sites approximately one to two weeks after the algaecide treatment before returning to pre-treatment levels (Figure 2-24 and Stillwater Sciences *in prep*). For the 2020–2022 dataset, with the exception of a small number of instances when there is a surface accumulation of cyanobacteria along the western shore of Lake Henshaw, the buoy line bottom site (H-FD) is a good predictor of microcystin concentrations at the fishing dock site (H-FD) (Figure 2-25).

Anatoxin-a was detected occasionally from October 2020 through June 2021 and consistently from November 2020 through January 2021 and again in July and early August 2022; thus this toxin also exceeded the CCHAB caution threshold (i.e., detection, where the analytical method must detect ≤ 1µg/L anatoxin-a; Table 1-1) for multiple weeks during recent monitoring (Figure 2-26). Peak anatoxin-a concentrations did not coincide with peak water temperatures in 2020 (where data are available), and they lagged peak water temperatures in 2021 by roughly four weeks (Figure 2-26). A similar lag was observed in summer 2022. Anatoxin-a concentrations ranged 1.09–7.15 µg/L prior to a copper-based algaecide treatment in August 2022 Within one to four days of treatment anatoxin-a concentrations were reduced below the reporting limit (Figure 2-26 and Stillwater Sciences *in prep*). For anatoxin-a in the 2020–2022 dataset, the fishing dock location tends to be slightly higher than the buoy line bottom (Figure 2-25).

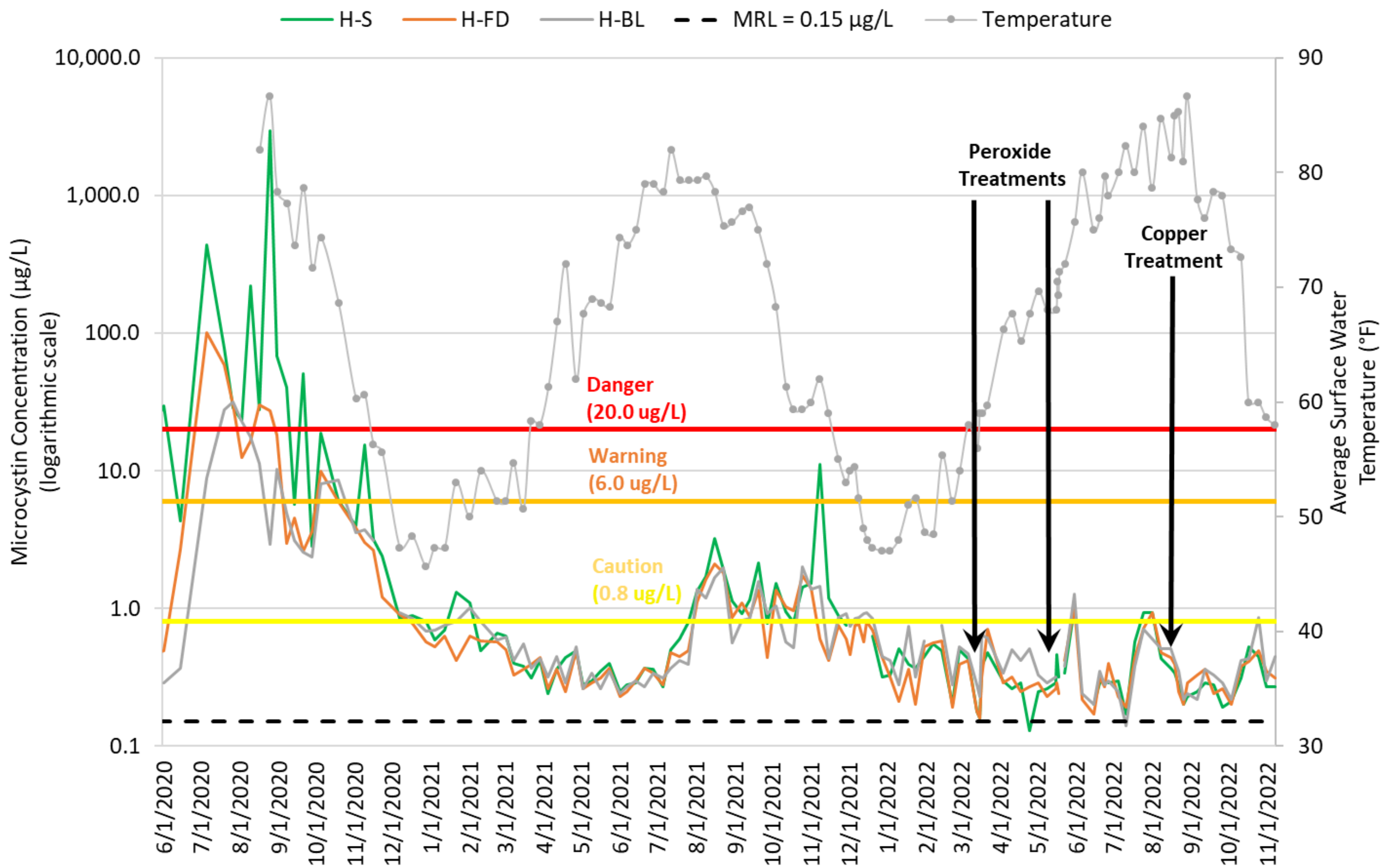

Figure 2-24. Microcystin concentrations in Lake Henshaw (left y-axis) for the period June 2020 through November 2022. Microcystin was also sampled in March 2020; however, results are not indicated here. Note that surface water temperature is shown on the right y-axis using a grey line with solid circles to distinguish this line from microcystin concentrations at the buoy line bottom site (H-BL; solid grey line). CCHAB threshold concentrations for inland recreational waters are shown by horizontal red, orange, and yellow lines (see also Table 1-1).

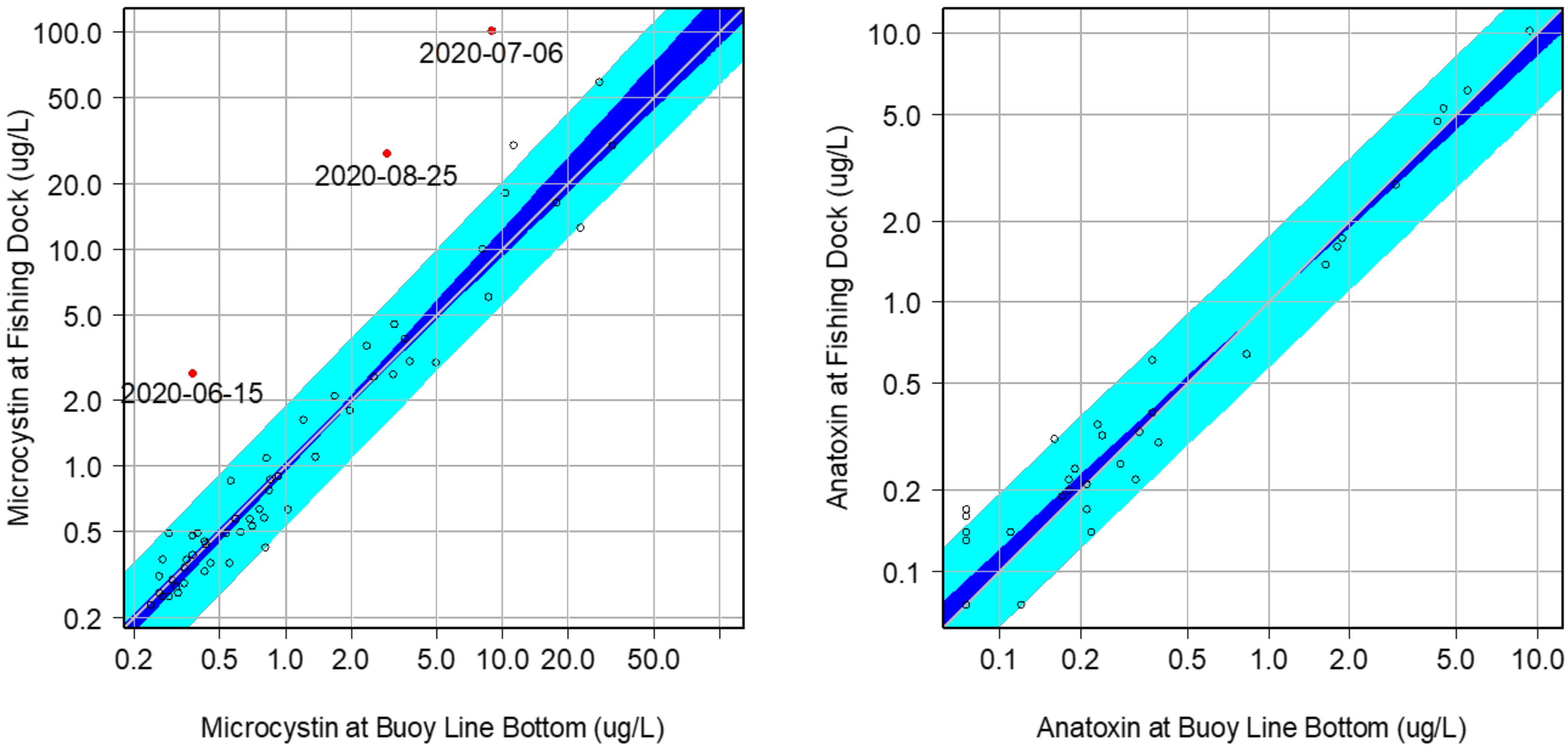

Figure 2-25. Data comparison for microcystin (left) and anatoxin-a (right) concentrations between buoy line bottom (H-BL) samples (x-axis) and fishing dock (H-FD) samples (y-axis). Diagonal grey line shown is 1:1 correspondence. Dark blue shading represents a 95% confidence band that shows that the fitted relationship is not significantly different from y=x (i.e., slope of 1). Light blue shading represents a 95% prediction band that shows how far away from the line a point would have to be before it is meaningfully different from y=x. Red dots represent outliers when surface accumulations at the shoreline site (H-S) exhibited substantially higher microcystin concentrations than were measured at the buoy line bottom site (H-BL).

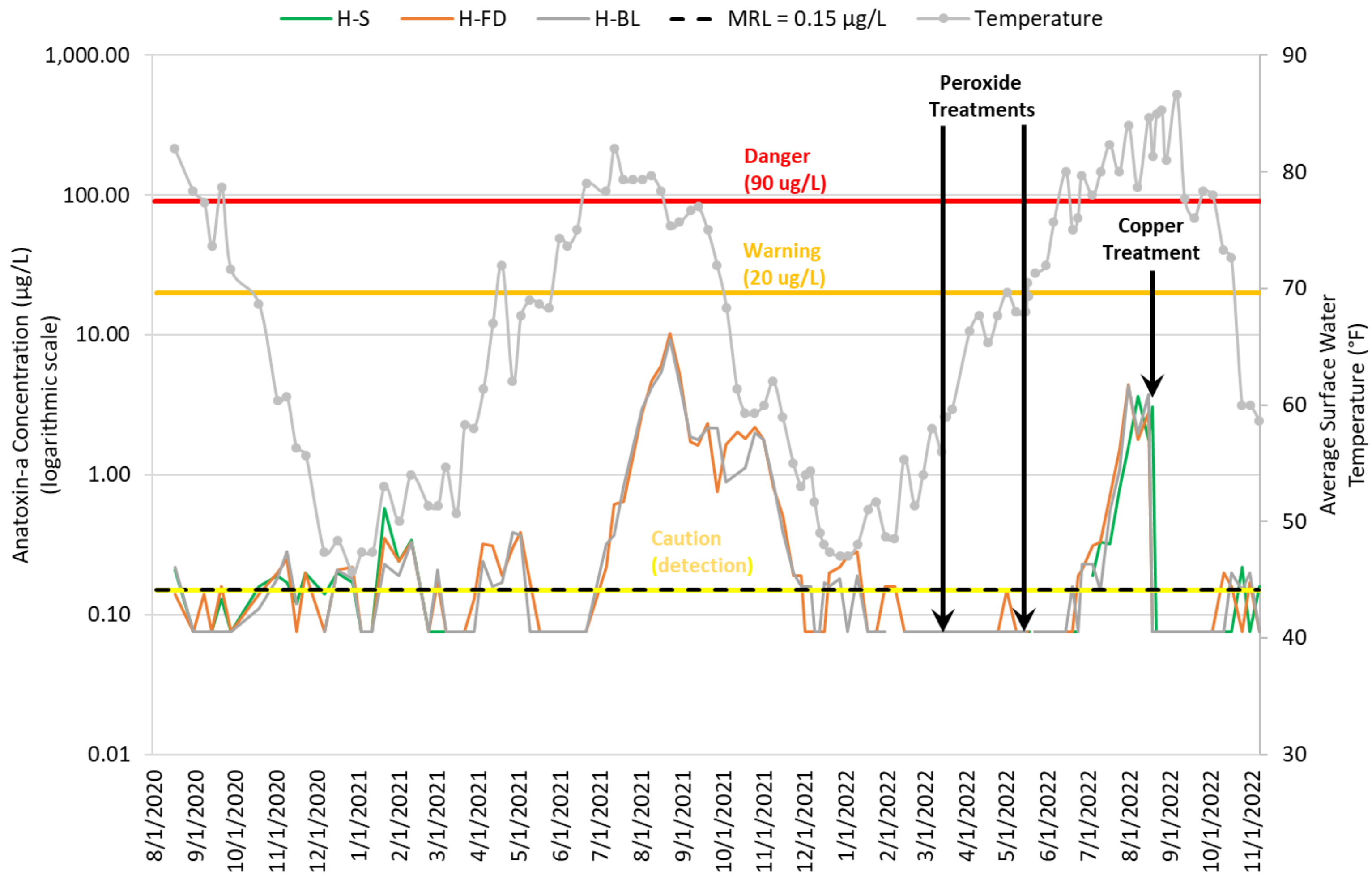

Figure 2-26. Anatoxin-a concentrations in Lake Henshaw, September 2020 through May 2022. Note that surface water temperature is shown on the right y-axis using a grey line with solid circles to distinguish this line from microcystin concentrations at the buoy line bottom site (H-BL; solid grey line). CCHAB threshold concentrations for inland recreational waters are shown by horizontal red, orange, and yellow lines (see also Table 1-1). Values reported below the method reporting limit (MRL) are reported as 0.5 x MRL.

2.2.3 Algae community composition and dynamics

2.2.3.1 Planktonic (suspended) algae

The dominant phytoplankton genera identified in the USEPA National Eutrophication Study included one heterokont algae (*Mallamonas*) and two cyanobacteria (*Microcystis* and *Oscillatoria*) (USEPA 1975). A robust quantitative comparison between data collected in the 1975 plankton surveys and more recent data collected by the District is difficult to conduct because most recent samples characterize relative abundance of cyanobacteria (i.e., dominant, sub-dominant, present) whereas the USEPA study provides total cell counts per unit volume (i.e., mL). The District collected cyanobacterial cell density (i.e., cells/mL) samples between late February and mid-March of 2022 (Appendix B), which allow for a more direct comparison with historical data. Based on the USEPA total cell counts, it appears that in 1975 there were little to no cyanobacteria species present during spring (March), the cyanobacteria *Microcystis* was co-dominant with a type of green algae (*Ankistrodesmus*) in summer (June), and the cyanobacteria *Oscillatoria* was dominant along with a mix of other green algae, diatoms, and cryptophyte algae in late fall/early winter (November) (Table 2-7). Recent cyanobacteria cell counts in Lake Henshaw suggest cell densities of *Microcystin* and *Planktothrix* (formerly called *Oscillatoria*) observed in 2022 were one-to-three orders of magnitude greater than those observed in 1975 (see Appendix B compared with values presented below in Table 2-7). Furthermore, the total density of cyanobacterial cells in 2022 was several orders of magnitude greater than the total density of all genera combined in 1975. Comparing chlorophyll-*a* densities recently measured in Lake Henshaw (between approximately 60–120 µg/L) (Appendix B) to historical chlorophyll-*a* densities (between approximately 6–45 µg/L) (USEPA 1975) further supports the conclusion that primary productivity in the lake has increased in the intervening period.

Table 2-7. Lake Henshaw phytoplankton collected during USEPA National Eutrophication Survey[1].

Sampling Date	Group	Dominant Genera	Algal Units per mL
3/7/1975	Heterokont algae (*Ochrophyta*)	*Mallomonas*	7,701
	Cryptophyte algae (*Chryptophyta*)	*Chroomonas*[2]	553
	Green algae (*Chlorophyta*)	*Ankistrodesmus*	277
	Cryptophyte algae (*Chryptophyta*)	*Cryptomonas*	184
	Diatoms (*Bacillariophyta*)	*Melosira*	184
	Other genera	-	323
	Total	-	9,222
6/23/1975	Blue-green algae (*Cyanobacteria*)	*Microcystis*	987
	Green algae (*Chlorophyta*)	*Ankistrodesmus*	816
	Dinoflagellates (*Dinoflagellata*)	*Glenodinium*	258
	Cryptophyte algae (*Chryptophyta*)	*Chroomonas*[2]	258
	Green algae (*Chlorophyta*)	*Kirchneriella*	215
	Other genera	-	728
	Total	-	3,262

Sampling Date	Group	Dominant Genera	Algal Units per mL
11/13/1975	Blue-green algae (*Cyanobacteria*)	*Oscillatoria*	681
	Cryptophyte algae (*Chryptophyta*)	*Chroomonas*[2]	367
	Green algae (*Chlorophyta*)	*Ankistrodesmus.*	314
	Green algae (*Chlorophyta*)	*Pediastrum*	209
	Diatoms (*Bacillariophyta*)	*Melosira.*	209
	Other genera	-	577
	Total	-	2,357

[1] Samples were collected as integrated depth samples at two sites (i.e., one site near the dam and one site in the shallow southern portion of the lake) and composited for each sampling date.
[2] USEPA (1978) includes "?" in association with this genus.

More recently, cyanobacteria species have been identified at three sites in Lake Henshaw and at one site at the lake outlet when the lake is releasing water (Figure 2-6). Cyanobacteria grab samples are collected weekly from surface waters at the H-S and H-FD sites and from bottom waters at the H-BL site (Figure 2-6) and typically identified to genus as dominant, sub-dominant, or present in the sample. A subset of grab samples collected in late February, March, May, and June 2022 were identified to genus and included cyanobacteria cell counts (Appendix B).

The primary cyanobacterial genera observed in Lake Henshaw since February 2020 include: *Microcystis*, *Planktothrix*[9], *Snowella*, *Aphanizomenon*, *Woronichinia*, and *Dolichospermum.* All genera listed, excluding *Snowella*, are known to produce microcystin toxins (e.g., Fastner et al. 1999). *Aphanizomenon*, *Woronichinia*, and *Dolichospermum* produce anatoxins.

Planktothrix grows as a single trichome (i.e., filament), rather than a clumped mass of trichomes (like *Aphanizomenon*) or a mass of cells encased in a mucus (like *Microcystis*), and thus *Planktothrix* cannot regulate its buoyancy well. *Planktothrix* does not tend to accumulate in surface scums. *Planktothrix* grows well in open, well-mixed waterbodies such as Lake Henshaw in the winter and spring months (see Appendix B) where buoyancy regulation is not a great advantage over competing genera.

Aphanizomenon and *Dolichospermum* are able to fix nitrogen from the atmosphere when supply of this nutrient is low in lake water. These two genera can form akinetes, which are resilient, thick-walled, non-motile cells that allow cyanobacteria to survive harsh environmental conditions and periods of extended dormancy. Their ability to form akinetes means that they are likely to be fairly resilient members of Lake Henshaw's cyanobacterial community.

Relative abundance of cyanobacteria genera has varied seasonally during 2020-2022. Though minor differences in cyanobacterial community composition exist between sampling locations, overall patterns are generally consistent across sites, with *Microcystis* dominating in the summer of 2020 and *Dolichospermum* in early fall 2020, followed by a switch to *Snowella* in late fall 2020 and continuing into spring 2021. *Planktothrix* became dominant throughout late spring and summer 2021, whereas *Dolichospermum* dominated in late summer and fall 2021 (Figure 2-27 through Figure 2-31). *Planktothrix* regained dominance in winter of 2021/2022 with sub-dominance oscillating between *Snowella, Aphanizomenon,* and *Cuspidothrix. Geitlerinema* became dominant in late summer 2022.

[9] Some species of *Planktothrix* were previously grouped within the genus *Oscillatoria*, but based on more recent scientific identification work, *Planktothrix* has been defined as its own genus.

Aggregated weekly estimates of cyanobacterial cell concentrations produced using chlorophyll-a and phycocyanin remote sensing data (from EPA's CyAN app [Cyanobacteria Assessment Network Application (CyAN app) | US EPA]) provide insight into bloom density over the course of the 2020–2022 sampling period (USEPA 2021). Because satellite imagery can only effectively capture conditions in surface waters, the CyAN app may underestimate cell concentrations for blooms dominated by filamentous species that also occupy lower portions of the water column or do not form large aggregations at the surface. Nonetheless, cyanobacterial concentrations appeared to peak in summer and fall 2020, where the summer remotely-sensed cyanobacteria cell count peak corresponded with *Microcystis sp.* relative dominance and the highest microcystin toxin concentrations measured in Lake Henshaw to date (Figure 2-27 through Figure 2-31). The fall 2020 remotely-sensed cyanobacteria cell count peak corresponded to a *Planktothrix sp.* relative dominance, but a resurgence of *Microcystis sp.* and a return to high microcystin toxin concentrations. Cell counts declined in winter 2020 and early spring 2021, as *Microcystis sp.* became sparse and microcystin concentrations decreased to below the CCHAB caution threshold of 0.8 µg/L. During this period, *Snowella sp.* and *Planktothrix sp.* were co-dominant, depending on the week, and although microcystin was relatively low, anatoxin-a was periodically detected, albeit at low concentrations (Figure 2-27 through Figure 2-31). Remotely-sensed cyanobacteria cell counts indicate a spring bloom in April-June 2021, which was dominated by *Planktothrix sp.*, a large summer/fall bloom in July-October dominated by *Dolichospermum sp.*, and a continuing large fall/winter bloom in November-February that was again dominated by *Planktothrix sp.* Remotely-sensed cyanobacteria cell counts decreased in late spring and early summer 2022 before climbing in June and July. Peak anatoxin-a concentrations corresponded to the large summer/fall bloom dominated by *Dolichospermum sp.* in 2021 and *Geitlerinema* in 2022, and microcystin concentrations, while detected, were one to two orders of magnitude lower as compared with 2020 (Figure 2-27 through Figure 2-31).

Overall, the 2020 and 2021 data indicate that 1) the highest cyanotoxin concentrations tend to coincide with the highest cyanobacteria cell counts (remotely sensed), and 2) the primary microcystin producer may be *Microcystis sp.* and the primary anatoxin-a producer may be *Dolichospermum sp.*, although genetic studies would be necessary to confirm the primary producers of cyanotoxins in Lake Henshaw. There may be a benthic component to production as well (see also Section 2.2.3.2). Data from summer 2022 suggest *Geitlerinema* could also be involved in anatoxin-a production, as elevated cyanobacteria cell counts of this genus coincided with a spike in anatoxin-a concentrations measured in the lake. Further discussion of cyanotoxin concentrations and the cyanobacteria community in relation to algaecide treatment in May 2022 are presented in Appendix E and in August are presented in Stillwater Sciences (*in prep*).

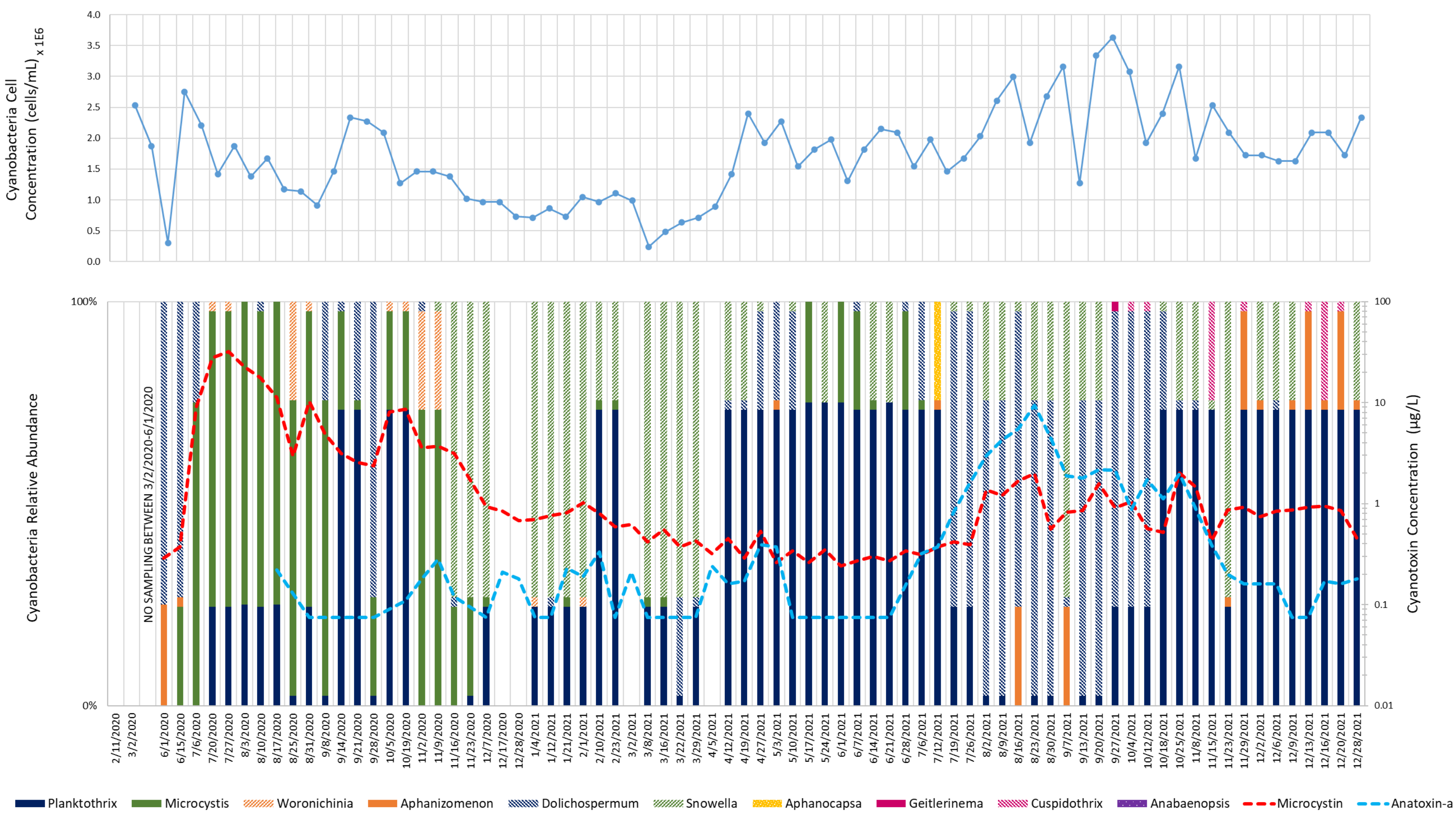

Figure 2-27. Lake Henshaw remotely sensed lake-wide aggregate cell concentrations (top) and estimated planktonic (suspended) cyanobacteria relative abundance and measured cyanotoxin concentrations at H-BL (bottom) in 2020–2021.

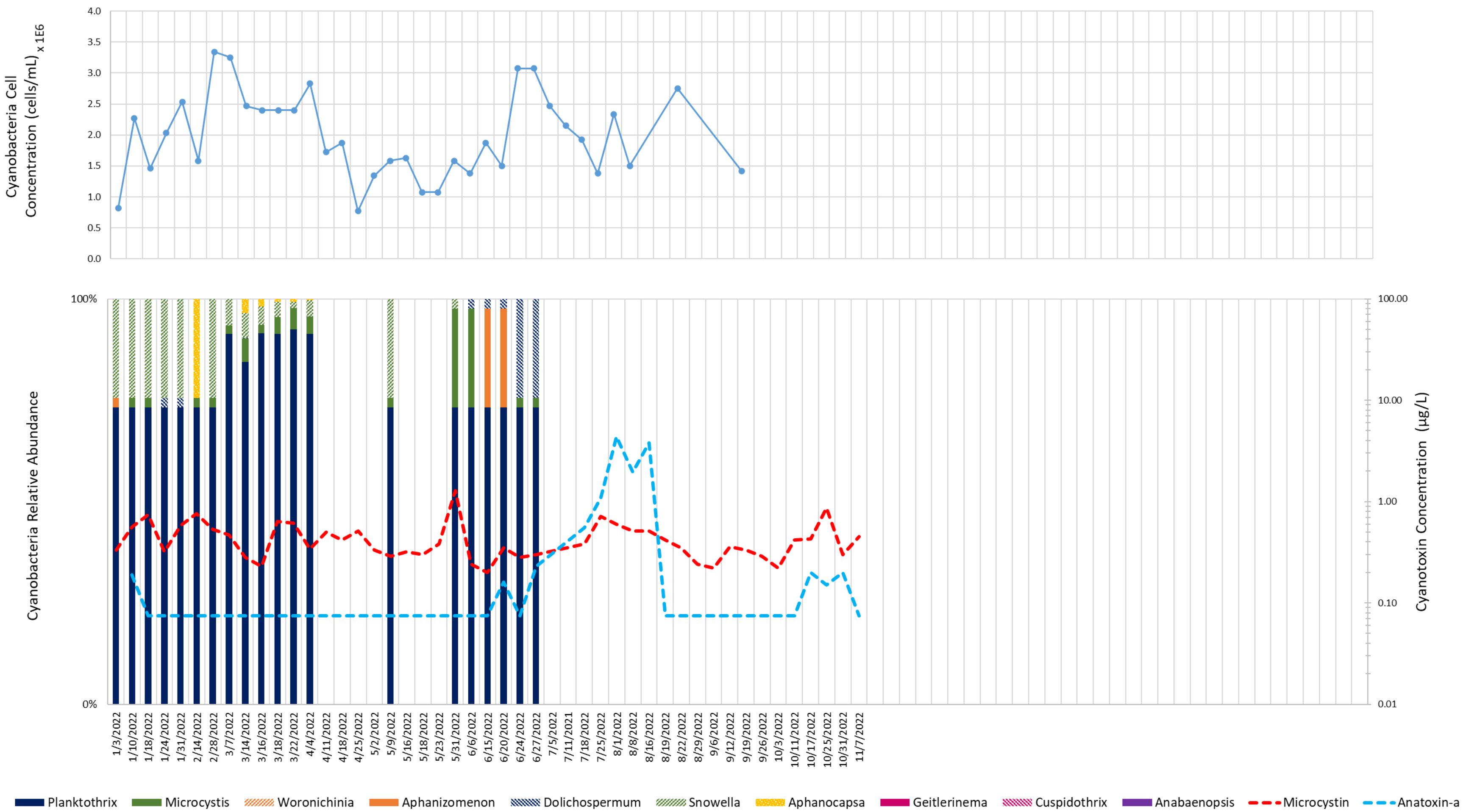

Figure 2-28. Lake Henshaw remotely sensed lake-wide aggregate cell concentrations (top) and estimated planktonic (suspended) cyanobacteria relative abundance and measured cyanotoxin concentrations at H-BL (bottom) in 2020–2022.

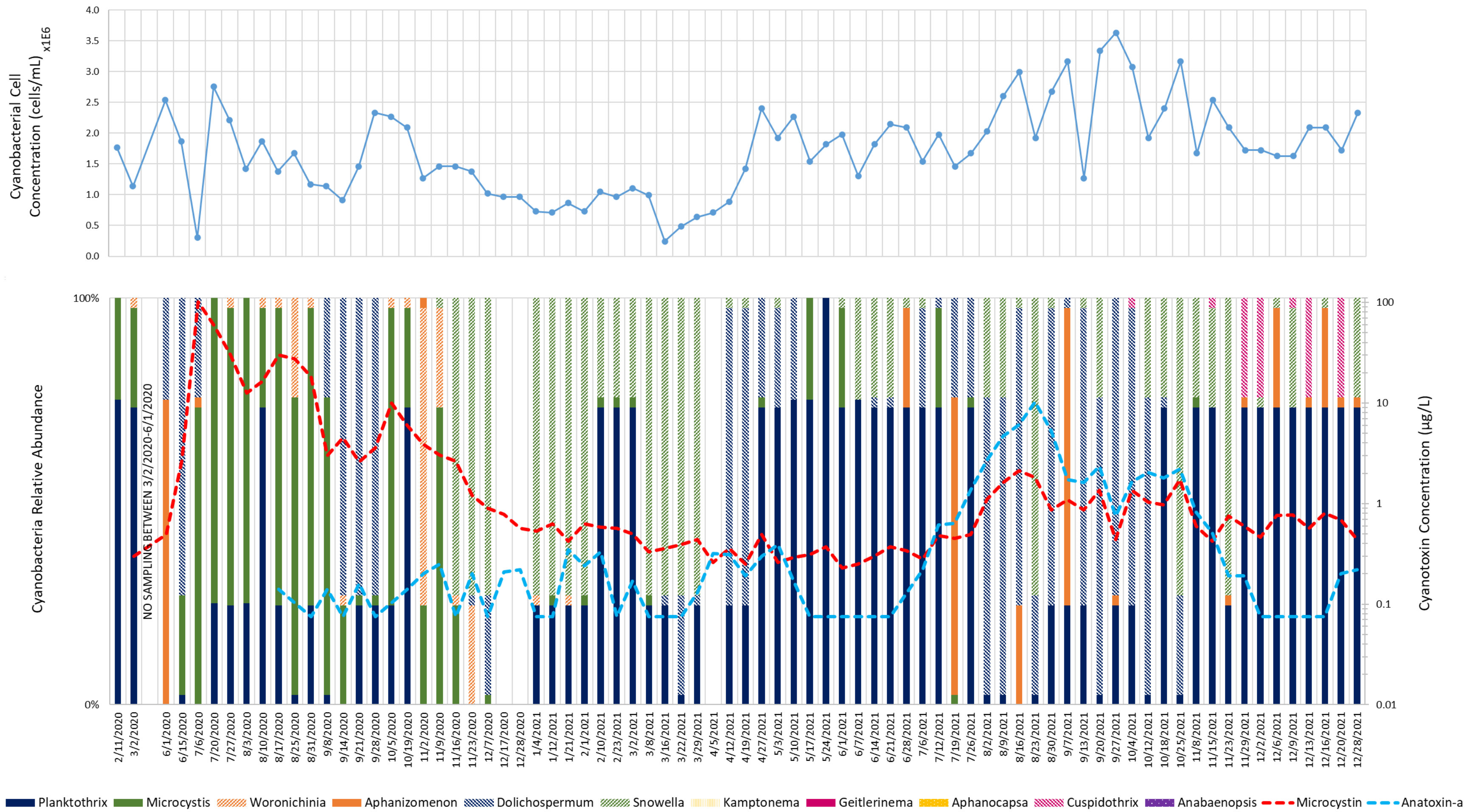

Figure 2-29. Lake Henshaw remotely sensed lake-wide aggregate cell concentrations (top) and estimated planktonic (suspended) cyanobacteria relative abundance and measured cyanotoxin concentrations at the H-FD (surface) 2020–2021.

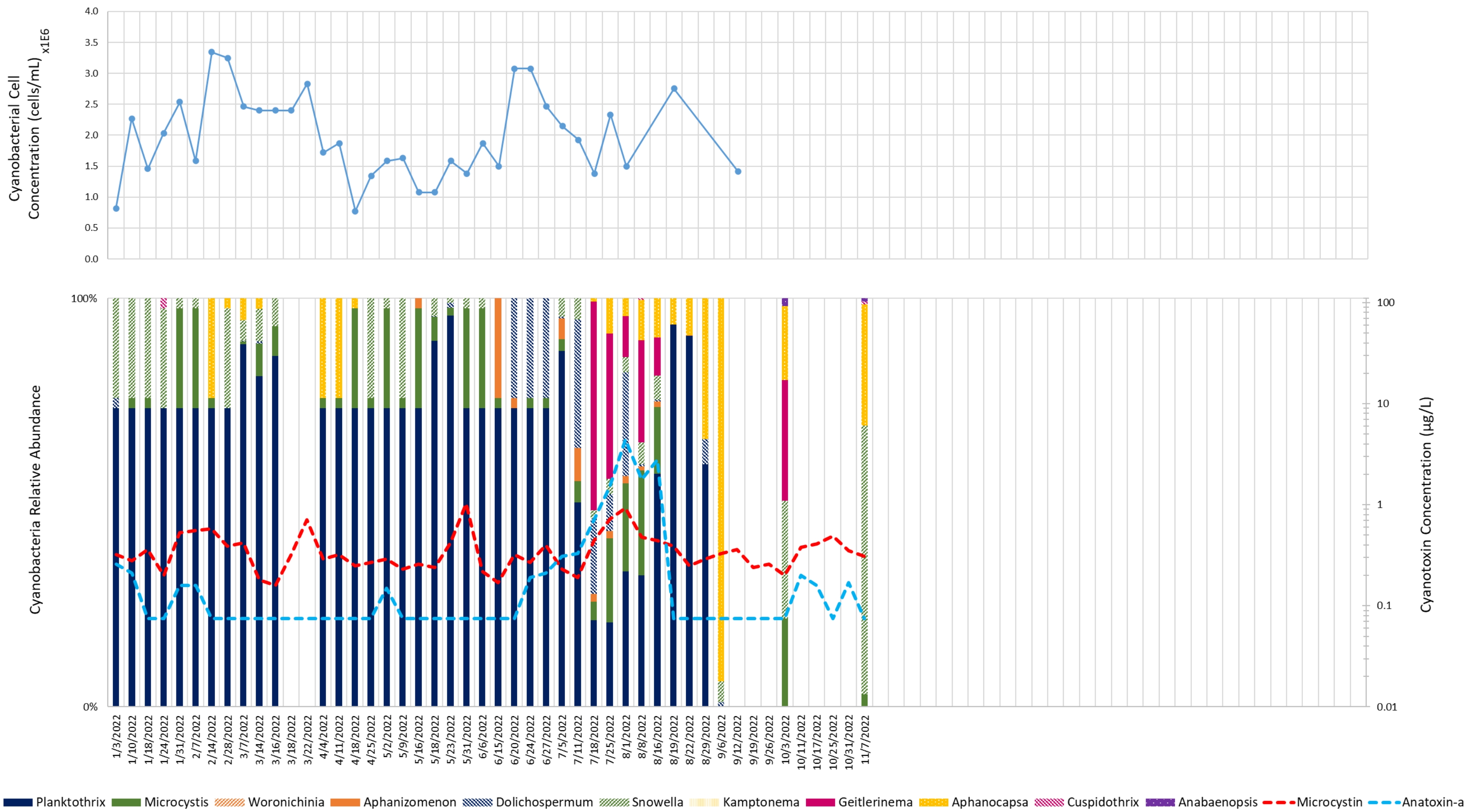

Figure 2-30. Lake Henshaw remotely sensed lake-wide aggregate cell concentrations (top) and estimated planktonic (suspended) cyanobacteria relative abundance and measured cyanotoxin concentrations at the H-FD (surface) 2020–2022.

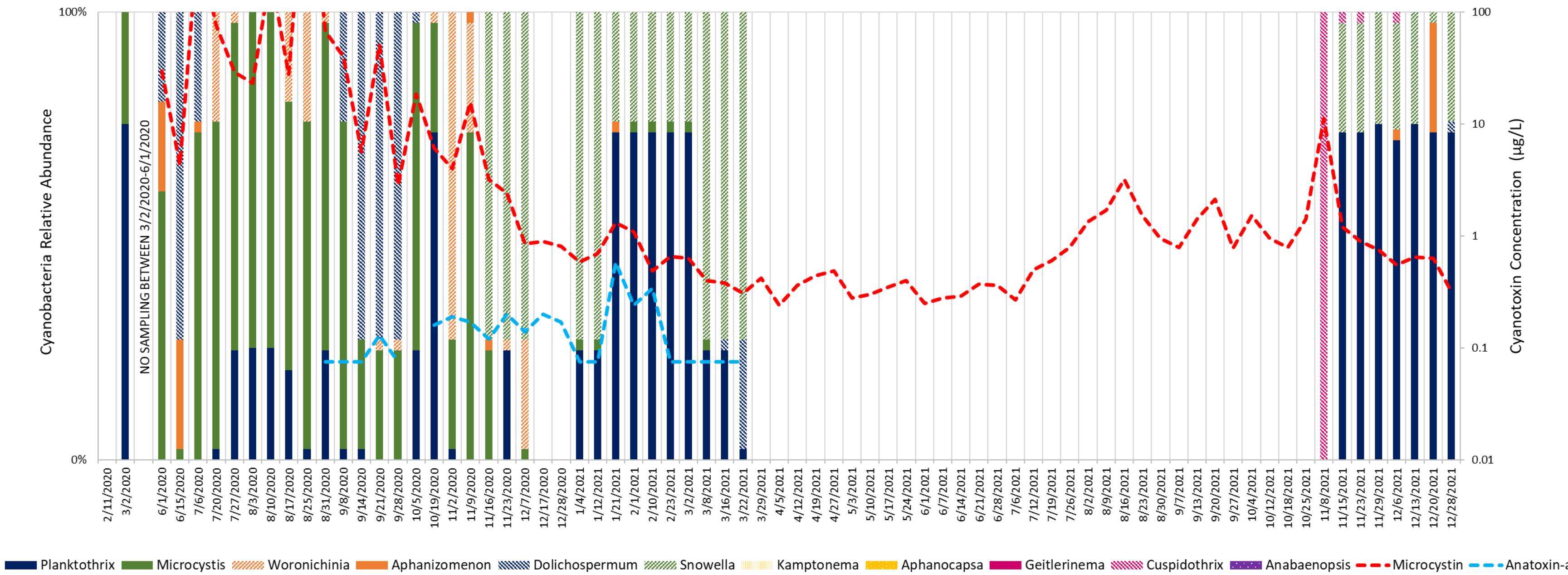

Figure 2-31. Lake Henshaw estimated planktonic (suspended) cyanobacteria relative abundance and measured cyanotoxin concentrations at H-S in 2020–2021. Cyanobacterial species and anatoxin-a concentrations not sampled between 3/22/2021 and 11/8/2021.

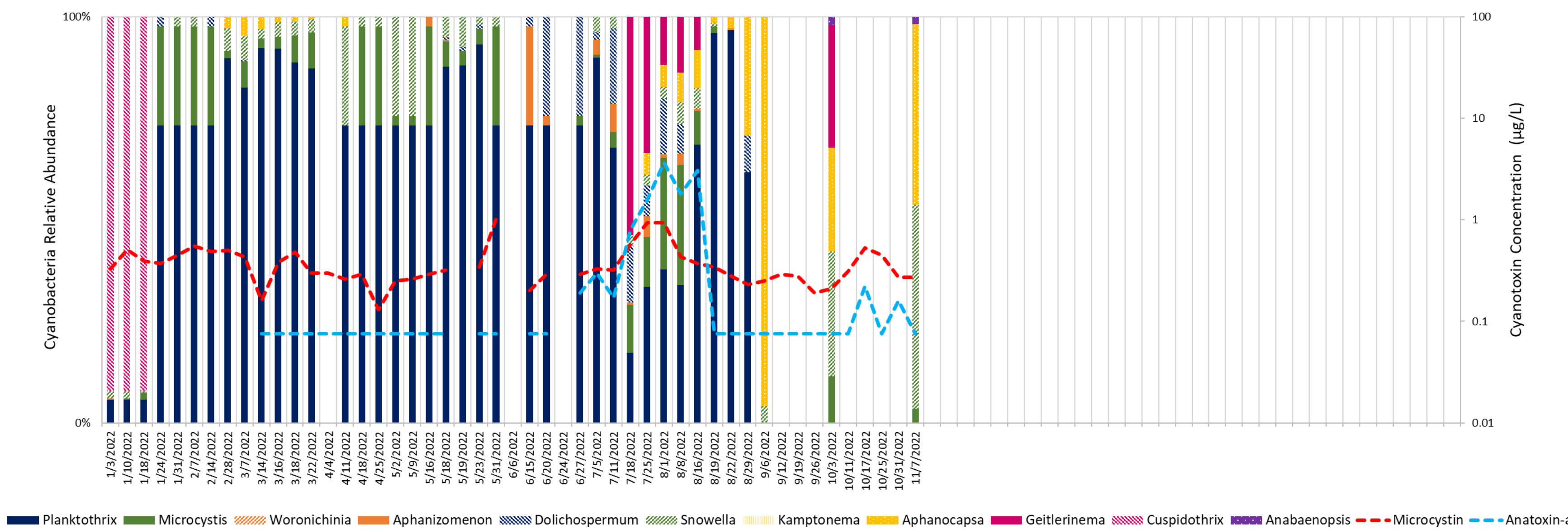

Figure 2-32. Lake Henshaw estimated planktonic (suspended) cyanobacteria relative abundance and measured cyanotoxin concentrations at H-S in 2020–2022.

2.2.3.2 Benthic (attached) algae

In lakes and reservoirs, benthic algae are attached species associated with sediments and other bottom substrates (e.g., rocks, rooted aquatic macrophytes) within the photic zone, such as shallow areas near the shoreline. Benthic algae can be an important component of overall primary productivity in lakes and reservoirs. Although there are no available benthic algae data for Lake Henshaw, they may be an important part of the algal community with respect to production of cyanotoxins (see Section 2.2.2.5).

2.3 San Luis Rey River and Escondido Canal

2.3.1 Hydrology

During April through October in most years, water in the San Luis Rey River between Henshaw Dam and the Escondido Canal Diversion Dam is predominantly supplied by releases from the dam (Figure 2-33). During winter months in years with high runoff, peak releases from the dam tend to lag the primary runoff period by two to three months since flows from Prisoner, Cedar, and several unnamed creeks enter the river between Henshaw Dam and the Escondido Canal Diversion Dam (Figure 1-1), and runoff from these creeks dominates flow in the San Luis Rey River during and immediately following storm events (Figure 2-33). Dam releases in April through October tend to be in the range of 30 to 50 cubic ft per second (cfs), although during drier periods (e.g., 2016–2020) they can be lower. Releases in the 30 to 50 cfs range can provide recreational flows at the La Jolla Band's concession on their reservation lands, which include a Creekside campground (Figure 2-34).

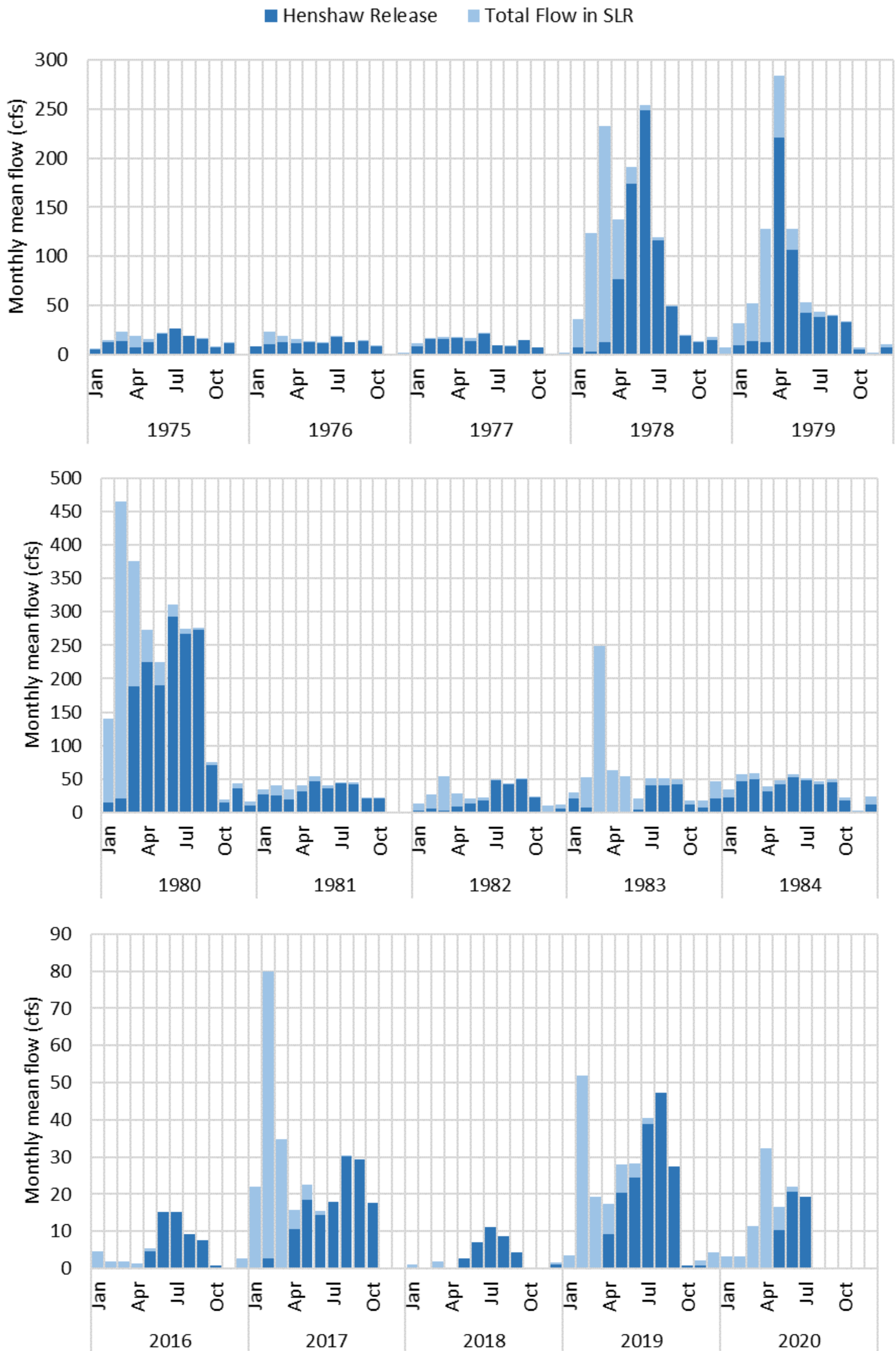

Figure 2-33. Monthly mean Lake Henshaw release flows as a component of total flows in the San Luis Rey River (SLR) between Henshaw Dam and the Escondido Canal Diversion Dam for selected years during the period 1953-2020. Note the change in y-axis for each plot.

Figure 2-34. San Luis Rey River adjacent to the La Jolla Band campground, looking downriver, March 4, 2021. Releases from Lake Henshaw were not occurring on this date.

At the Escondido Canal Diversion Dam, San Luis Rey River water is diverted into the Escondido Canal (Figure 2-36), which is approximately 14 miles long, approximately 8 ft wide, 6 ft in height, and the canal top is open the majority of its length (Figure 2-36). Escondido Canal is rated at a capacity of 70 cfs, but typically does not carry more than 50 cfs. During some periods, a portion of flow in the Escondido Canal is diverted back into the San Luis Rey river at a small measurement flume just downstream of the Diversion Dam to meet the annual entitlement for the Rincon Indian Reservation. The remaining water in the canal is delivered to Escondido Creek just upstream (approximately 500 yards) of Lake Wohlford. A rough estimate of hydraulic residence time (HRT) for water traveling at 30 to 50 cfs between Henshaw Dam and the Escondido Canal Diversion Dam is approximately 8 hours, and from the Diversion Dam to Lake Wohlford is approximately 12 hours.

Figure 2-35. Escondido Canal Diversion Dam looking upstream from the right bank, March 4, 2021.

Figure 2-36. Escondido Canal just downstream of the diversion dam, looking upstream, March 4, 2021.

2.3.2 Water quality

Water quality samples in the San Luis Rey River downstream of Henshaw Dam and the Escondido Canal have been collected by the District, Escondido, and the La Jolla Band periodically since 2020 (Table 2-8 and Figure 2-37).

Table 2-8. San Luis Rey River and Escondido Canal monitoring sites.

Site ID	Location	Latitude	Longitude
H-BL	Buoy line at dam in bottom waters	33.23963°N	116.76174°W
H-R	Dam release channel approximately 10 ft upstream of flow measurement weir (point of release to San Luis Rey River)	33.23923°N	116.76594°W
H-RR	Rey River Ranch	33.23963°N	116.76174°W
SWSLRCC8	La Jolla Band Campground Lower Site	33.23923°N	116.76594°W
DD/SWSLRK12	Diversion Dam/La Jolla Band Diversion Dam Site	33.27224°N	116.82998°W
PG	Paradise Grates	33.27230°N	116.84936°W

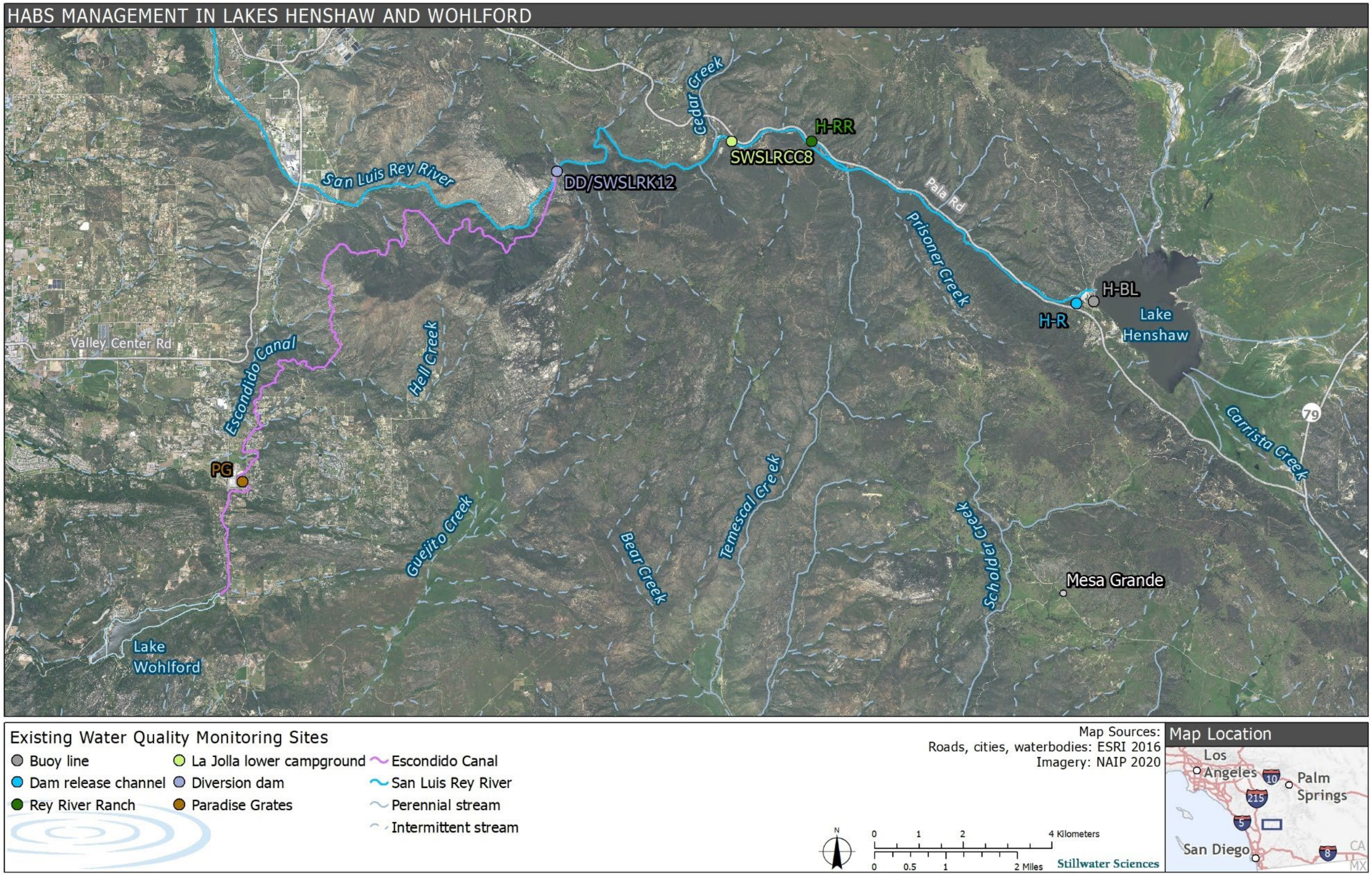

Figure 2-37. San Luis Rey River and Escondido Canal monitoring sites.

2.3.2.1 Nutrients

Nitrogen

TN concentrations in the San Luis Rey River and the Escondido Canal ranged 1.00 to 5.36 mg/L (ten dates in March 2021 through May 2022) during periods when Henshaw Dam releases were occurring. For the period of record, TN concentrations did not exhibit a consistent upstream to downstream pattern during summer or winter months (Figure 2-38), suggesting that TN transformations are likely to be minimal during the short transit (i.e., hours to days) from Lake Henshaw to Lake Wohlford.

Ammonia concentrations in the San Luis Rey River and the Escondido Canal during the aforementioned period ranged 0.004 to 0.37 mg/L. On June 1, 2021, ammonia concentrations decreased steadily from near the Lake Henshaw release (0.37 mg/L) through the San Luis Rey River (0.26 mg/l) and Paradise Grates (0.21 mg/L; Figure 2-38), suggesting uptake of ammonia, transformation of ammonia to nitrate through microbial nitrification, and/or dilution during transit. However, this pattern was not discernable on any other date during the period of record. Nitrate concentrations in the San Luis Rey River and the Escondido Canal during the aforementioned period ranged 0.05 to 0.33 mg/L and during summer months, when uptake of nitrate by algae and aquatic macrophytes is expected to be relatively high and warm temperatures can support microbial denitrification in river bottom substrate when DO is low, nitrate concentrations did not exhibit a consistent upstream to downstream pattern. During winter, when nitrate uptake is typically suppressed by low temperatures, there was also no pattern as water traveled from Henshaw Dam to Lake Wohlford (Figure 2-38).

Overall, the available TN, ammonia, and nitrate data suggest that nitrogen transformations are likely to be minimal during the short transit (i.e., hours to days) from Lake Henshaw to Lake Wohlford.

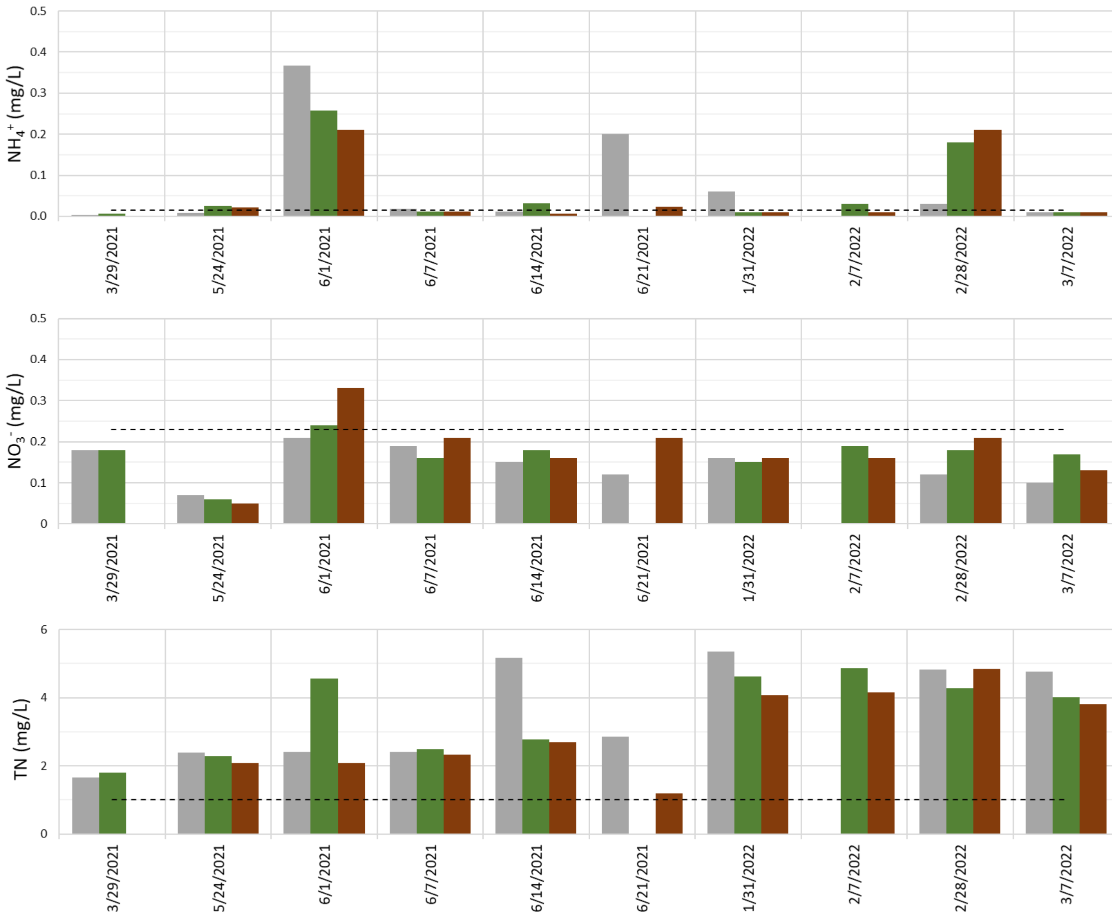

Figure 2-38. Total nitrogen (TN), nitrate (NO_3^-), and ammonia (NH_4^+) in a longitudinal transect from Lake Henshaw near the dam release (i.e., buoy line bottom [H-BL]), the Rey River Ranch (H-RR) and the downstream end of the Escondido Canal (i.e., Paradise Grates [H-PG]) when releases from Henshaw Dam are occurring.

Phosphorus

TP concentrations in the San Luis Rey River and the Escondido Canal ranged 0.04 to 0.4 (10 dates in March 2021 through May 2022) during periods when Henshaw Dam releases were occurring. For the period of record, TP concentrations did not exhibit a consistent upstream to downstream pattern during summer or winter months (Figure 2-39), suggesting that TP transformations are likely to be minimal during the short transit (i.e., hours to days) from Lake Henshaw to Lake Wohlford. Orthophosphate concentrations ranged 0.01 to 0.10 mg/L. Orthophosphate concentrations tended to be slightly higher at the Paradise Grates (H-PG; Figure 2-39) than at other sites, which could be due to algae cell rupture (lysis) and release of orthophosphate in the more turbulent waters of the river and canal relative to conditions in Lake Henshaw. However, overall the available TP and orthophosphate data suggest that phosphorus transformations are likely to be minimal during the short transit (i.e., hours to days) from Lake Henshaw to Lake Wohlford.

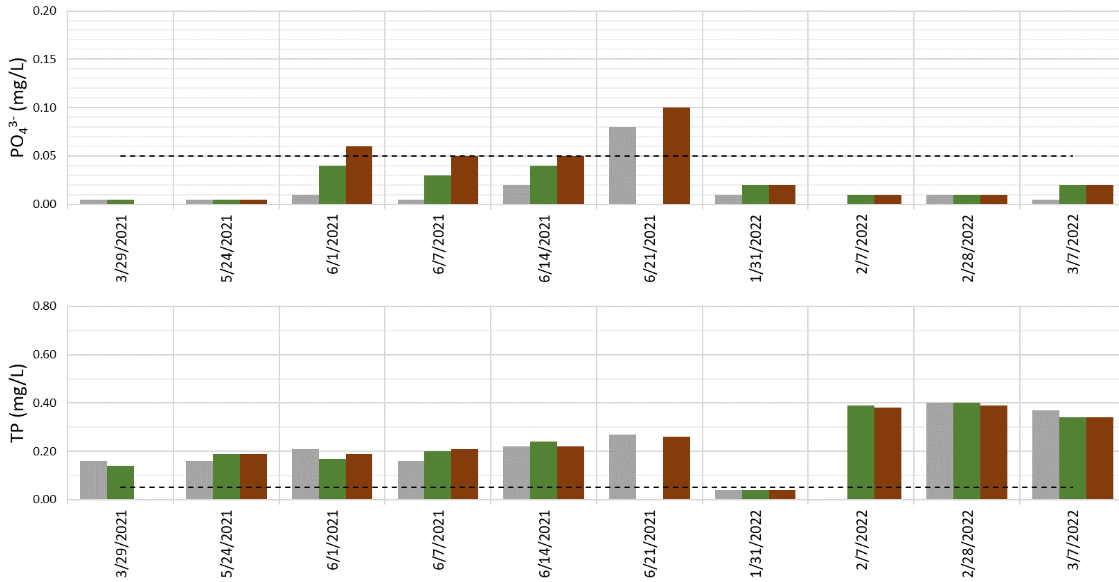

Figure 2-39. Total phosphorus (TP) and orthophosphate (PO_4^{3-}) in a longitudinal transect from Lake Henshaw near the dam release (i.e., buoy line bottom [H-BL]), the Rey River Ranch (H-RR) and the downstream end of the Escondido Canal (i.e., Paradise Grates [H-PG]) when releases from Henshaw Dam are occurring.

2.3.2.2 Cyanotoxins

Microcystin concentrations in the San Luis Rey River and the Escondido Canal ranged 0.2 to 0.5 µg/L (nine dates in May 2021 through May 2022) during periods when Henshaw Dam releases were occurring. While the available data suggest that concentrations tend to decrease slightly with distance downstream, the variability in concentrations across dates and sites results in no statistically significant trend ($F[4,35] = 1.77$, $p = 0.16$ for analysis of variance [ANOVA]) with distance downstream (e.g., Figure 2-40) for all nine dates sampled. The lack of upstream to downstream trend suggests that substantial amounts of microcystin are not being produced during transit between Lake Henshaw and Lake Wohlford, nor is a substantial amount of degradation of this toxin occurring along the way.

Anatoxin-a was not consistently detected in samples in the San Luis Rey River and the Escondido Canal when Henshaw Dam releases were occurring, since detection in Lake Henshaw typically precludes releases of water containing this cyanotoxin. Anatoxin-a was detected at the Diversion Dam (0.19 µg/L) on February 7, 2022, which suggests uneven mixing of water as it travels downstream, or potentially small amounts of production in transit.

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Figure 2-40. Microcystin concentrations in a longitudinal transect from Lake Henshaw near the dam release (i.e., buoy line bottom [H-BL]), to the downstream end of the Escondido Canal (i.e., Paradise Grates [PG]) when releases from Henshaw Dam were occurring for the continuous period of May 24 through June 21, 2021 (five of the nine dates for which there are data). Note that not all sites were sampled on each date.

2.3.3 Algae community composition and dynamics

2.3.3.1 Planktonic (suspended) algae

When water is being released from Lake Henshaw, planktonic cyanobacteria species have been identified at one site at the lake outlet (buoy line bottom; H-BL), one site at the release (H-R), one site in the San Luis Rey River (Rey River Ranch; H-RR), and one site along the Escondido Canal (Paradise Grates; H-PG). Cyanobacteria grab samples are collected and identified to genus as dominant, sub-dominant, or present in the sample.

The available data indicate that as water travels from Lake Henshaw into the San Luis Rey River and through the Escondido Canal, the primary planktonic (suspended) cyanobacterial genus observed in the lake during these release periods (*Planktothrix sp.*) remains dominant in the river and canal (Figure 2-41). This is consistent with the growth form of *Planktothrix* (i.e., single trichome or filament), which would remain suspended during transport and is unlikely to accumulate in scums or settle out of the water column even when water velocity decreases, such as in backwater eddies or along riverbanks (Figure 2-42). *Microcystis* and *Snowella* cells are often encased in a mucus and buoyant colonies can form accumulations in backwaters and shorelines; these genera remain subdominant at sites in the river and the Escondido Canal for all release dates sampled.

Consistent with nutrient (Section 2.3.2.1) and cyanotoxin (Section 2.3.2.2) data, the available algae community composition data suggest that the short transit time (i.e., hours to days) from Lake Henshaw to Lake Wohlford is not altering the dominant or subdominant planktonic species in the water being released from Henshaw Dam.

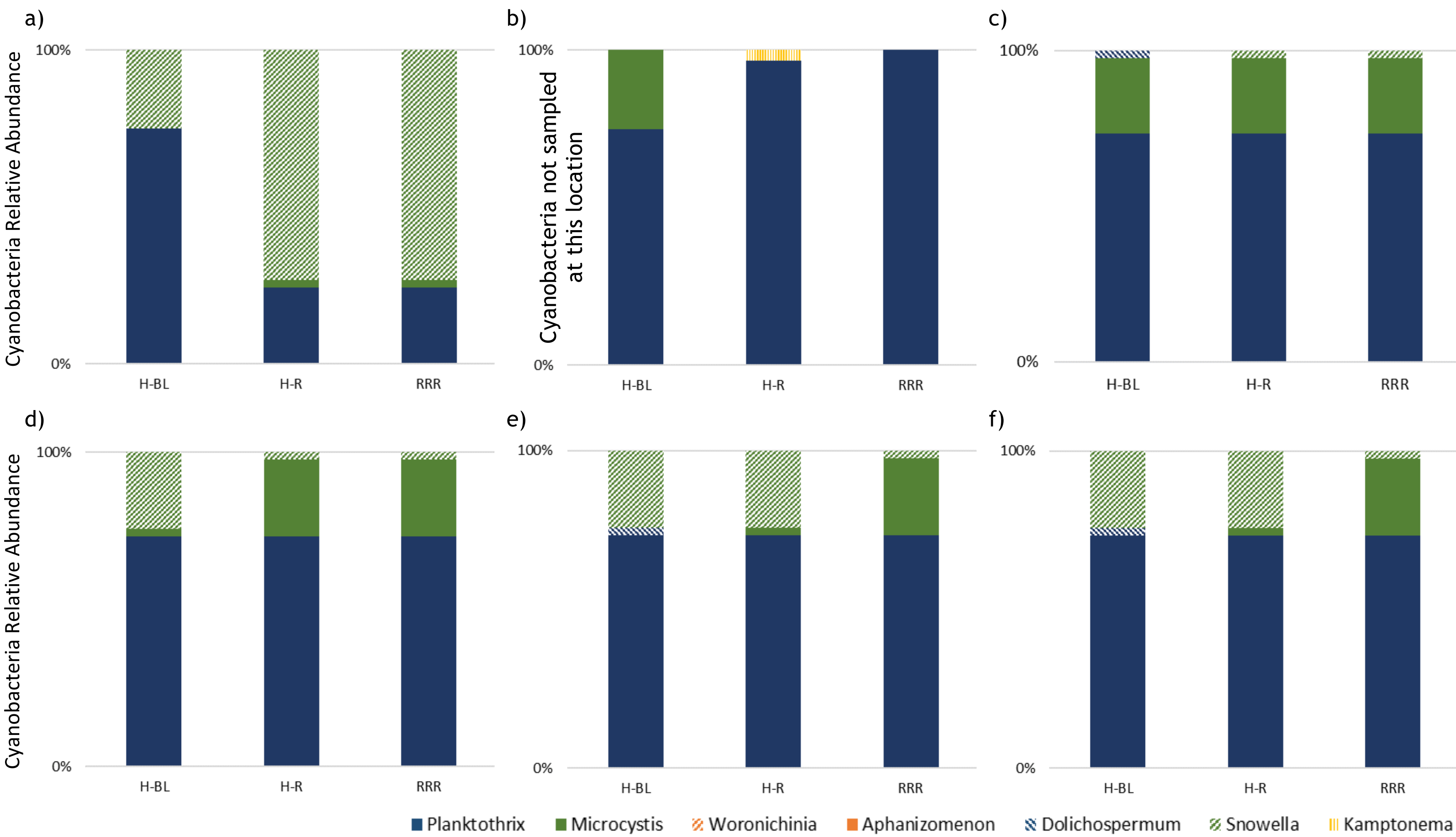
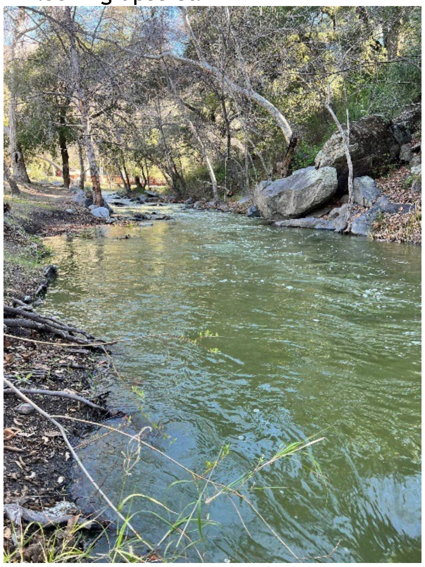
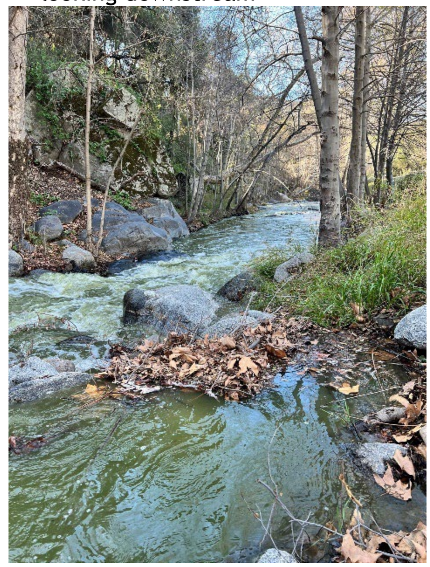


Figure 2-41. Planktonic (suspended) algae and cyanotoxin concentrations in Lake Henshaw and Escondido Canal on (a) March 29, 2021; (b) May 24, 2021; (c) June 1, 2021; (d) June 14, 2021; (e) January 31, 2022; (f) February 7, 2022.

a) La Jolla Lower Campground looking upstream



b) La Jolla Lower Campground looking downstream



c) Escondido Canal Diversion Dam looking upstream

d) Escondido Canal Diversion Dam looking downstream

Figure 2-42. San Luis Rey River during a release of water from Lake Henshaw that was dominated by *Planktothrix*, February 28, 2022. Photo credit: La Jolla Band.

2.3.3.2 Benthic (attached) algae

In rivers and streams, benthic algae are attached species associated with sediments and other bottom substrates (e.g., sand, cobble, boulders, aquatic macrophytes) within the photic zone, including shallow areas near the shoreline. Benthic algae can be an important component of overall primary productivity in flowing waters. There are no quantitative benthic algae data for the San Luis Rey River or the Escondido Canal, although no obvious mats of benthic algae have been reported or documented in the river or tributary creeks (e.g., Figure 2-34). The District and Escondido have reported stands of filamentous algae (e.g., *Chara*) growing in the Escondido Canal, occasionally at nuisance levels that need to be physically removed and/or treated with algaecide (SCS Engineers 2014). Given the lack of evidence of readily apparent benthic algae colonization of the San Luis Rey River (between Henshaw Dam and the Escondido Diversion Dam) and the District's and Escondido's ongoing management activities to control such growth in the Escondido Canal, as well as the lack of evidence that substantial amounts of cyanotoxins are being produced in the river or canal (Section 2.3.2.2), it is unlikely that benthic algae are an important source of cyanotoxins in the river/canal itself.

2.4 Lake Wohlford

2.4.1 Hydrology

Lake Wohlford is a small reservoir relative to Lake Henshaw, with a maximum storage capacity of approximately 6,500 AF and a maximum surface area of approximately 224 acres, originally constructed in 1895. California Division of Safety of Dam (DSOD) requirements beginning in 2007 reduced the maximum storage capacity to approximately 2,800 AF and the maximum surface area to approximately 145 acres (Figure 2-43). Although the maximum depth at full capacity is 80 ft, DSOD requirements also have reduced the maximum depth to 60 ft. Typical operating maximum depths are currently 50–55 ft, with little variation in water level within the past two decades. Relative to Lake Henshaw, the wind fetch at Lake Wohlford is relatively low at approximately 0.25−1 mile.

The City of Escondido is currently developing a project to replace Wohlford Dam, with an anticipated new water surface elevation of elevation of 1,490 ft, or approximately 10 ft higher than the elevation of the original dam (MWH 2016). The new dam will be located approximately 200 ft downstream of the existing dam.

Inflow to Lake Wohlford arrives primarily through the Escondido Canal (Figure 1-1), which delivers water from Lake Henshaw into the Escondido Creek just upstream (approximately 500 yards) of Lake Wohlford when releases from Henshaw Dam are occurring. Inflows are typically in the range of 30 cfs to 50 cfs (Figure 2-44), consistent with releases from Henshaw Dam (see also Section 2.3.1). Runoff from the 8 square-mile surrounding area also contributes to Escondido Creek inflows during winter months, although the magnitude of these flows are uncharacterized.

Average release flows from Lake Wohlford are approximately 23 cfs, with rare instances of releases over the dam spillway and into the downstream Escondido Creek that occur only during heavy and/or sustained precipitation (SCS Engineers 2014). Assuming the aforementioned average outflow, the mean hydraulic retention time of Lake Wohlford at the original maximum capacity of 6,500 AF would be just under 5 months and at the current maximum capacity of 2,800 AF would be roughly 2 months.

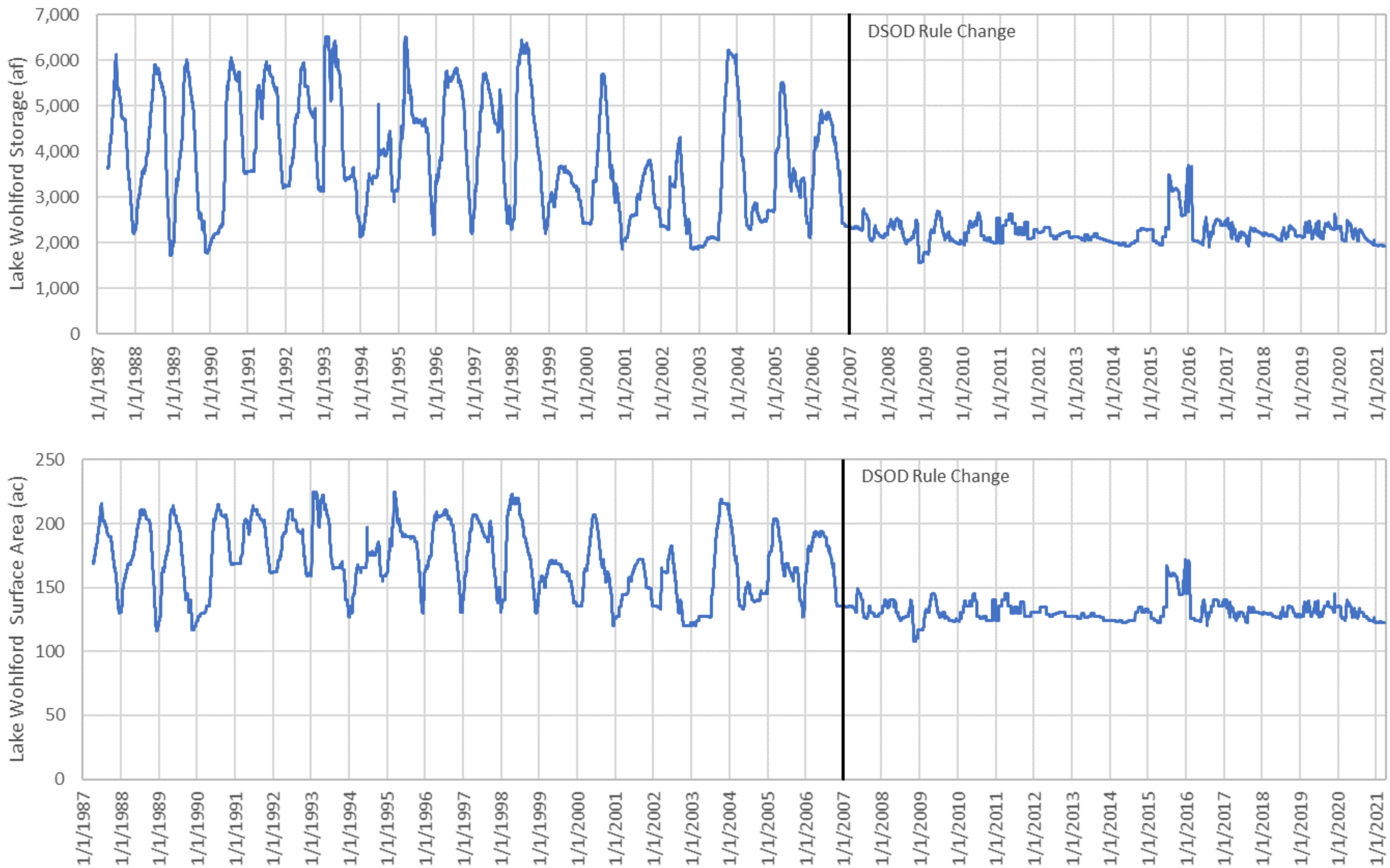


Figure 2-43. Lake Wohlford storage and surface area for the period 1987–2020. California Division of Safety of Dams (DSOD) rule change is shown by the black vertical line.

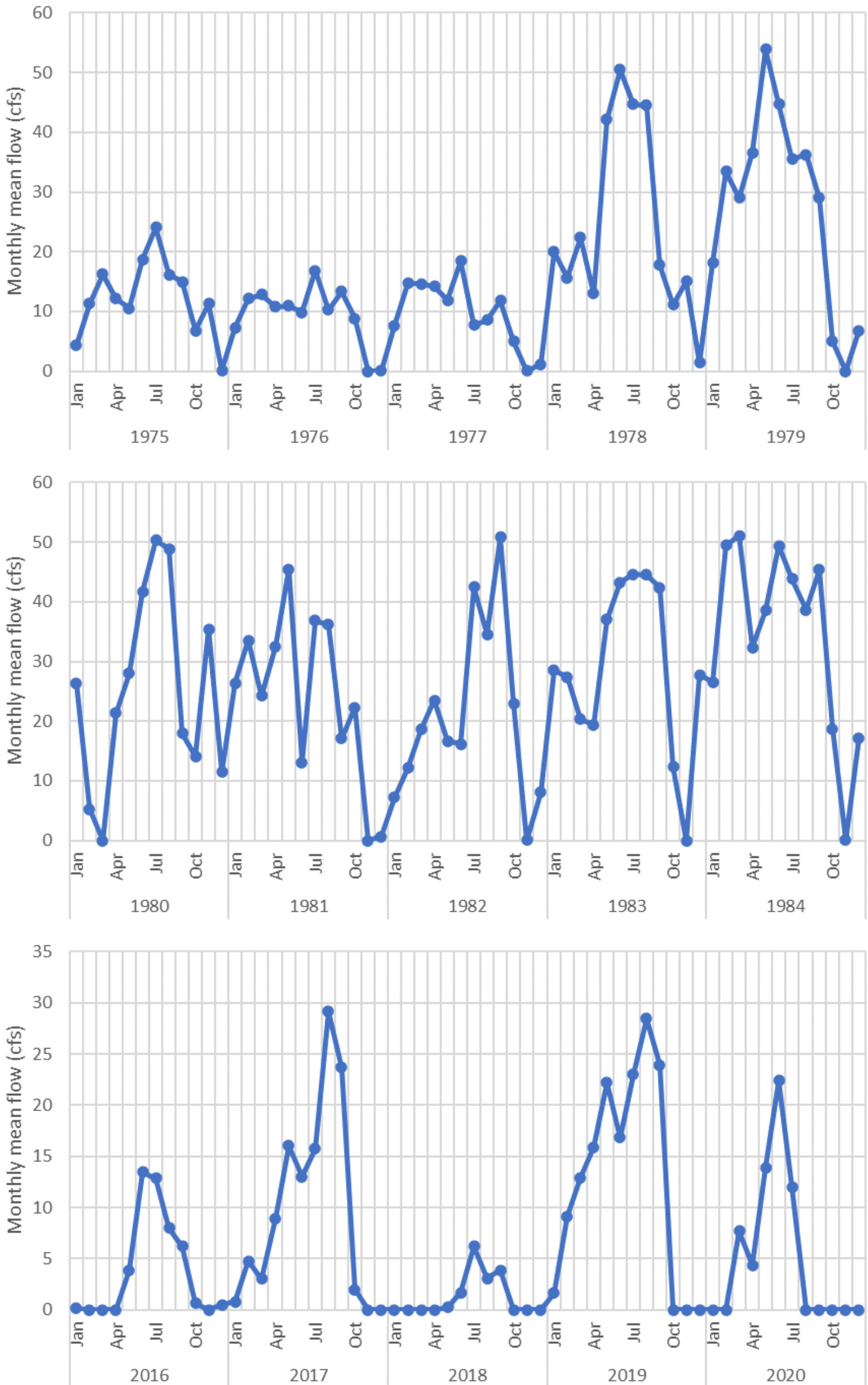


Figure 2-44. Monthly mean inflows from the Escondido Canal to Lake Wohlford for selected years during the period 1953–2020. Note the change in y-axis for the bottom plot.

2.4.2 Water quality

Lake water quality samples have been collected by Escondido approximately quarterly since 1997 for nutrients (nitrate, nitrite), and other constituents and parameters (alkalinity, *Escherichia coliform* and fecal coliform bacteria, conductivity, dissolved organic carbon, hardness, manganese, pH, total dissolved solids, total organic carbon, turbidity, water temperature). Quarterly samples have been collected near the outlet tower in the deepest portion of the reservoir. pH data are discussed in Section 2.4.2.2, and nutrients are presented in Section 2.4.2.3.

Escondido has also collected vertical profiles for *in situ* water temperature and DO in the deepest portion of the reservoir near the outlet tower. Profiles have been collected roughly bi-monthly since March 2016. Escondido operates an artificial aeration system (156 cubic ft per minute [cfm]) in Lake Wohlford, which generates coarse air bubbles (i.e., millimeters [mm] to a centimeter [cm] in diameter) that leave a single pipe located in the deepest portion of the reservoir near the dam. The artificial aeration system operates 19 hours a day (i.e., compressor is on between 11 pm and 4 pm), 7 days a week. The current condition and efficiency of the aeration system is unknown, although systems of this type generally supply little oxygen directly to the water since compressed air contains only 21% oxygen and the coarse air bubbles rise rapidly through the water column and exit to the atmosphere before oxygen can dissolve into the surrounding water. Available water temperature and DO data are presented in Section 2.4.2.1.

Beginning in May 2021, Escondido began collecting cyanotoxin data at three sites within Lake Wohlford (Figure 2-45; Table 2-9). Cyanotoxin sampling is primarily focused on microcystin but anatoxin-a samples are also collected on occasion. Additional details are provided in Section 2.4.2.4.

Table 2-9. Lake Wohlford cyanotoxin monitoring sites.

Location	Latitude	Longitude
Canal inlet	33.17510°N	116.99037°W
Near boat dock	33.17317°N	116.99893°W
Tower near dam	33.16694°N	117.00413°W

Lastly, algal sampling occurred weekly in Lake Wohlford between 2004 and 2011, and in 2016, with collected taxa generally identified to genus. Algae community composition for the period of record is discussed in Section 2.4.3.

Figure 2-45. Existing cyanotoxin monitoring sites in Lake Wohlford.

2.4.2.1 Seasonal stratification

Vertical profiles of water temperature in Lake Wohlford in the deepest portion of the water column near the dam indicate that occasional brief periods of thermal stratification can occur in the lake in the late winter/early spring (e.g., February–April 2017), summer (May 2018, June 2019), and fall (September 2019). However, the majority of the time the water column is isothermal (i.e., same temperature in surface and bottom waters) (Figure 2-46a,b). The regular lack of thermal stratification in the deepest Lake Wohlford waters is likely due to the artificial aeration system that, while not directly transferring much oxygen to the lake, mixes the water column near the dam for 19 hours a day, 7 days a week, such that temperatures are similar or the same in surface and bottom waters.

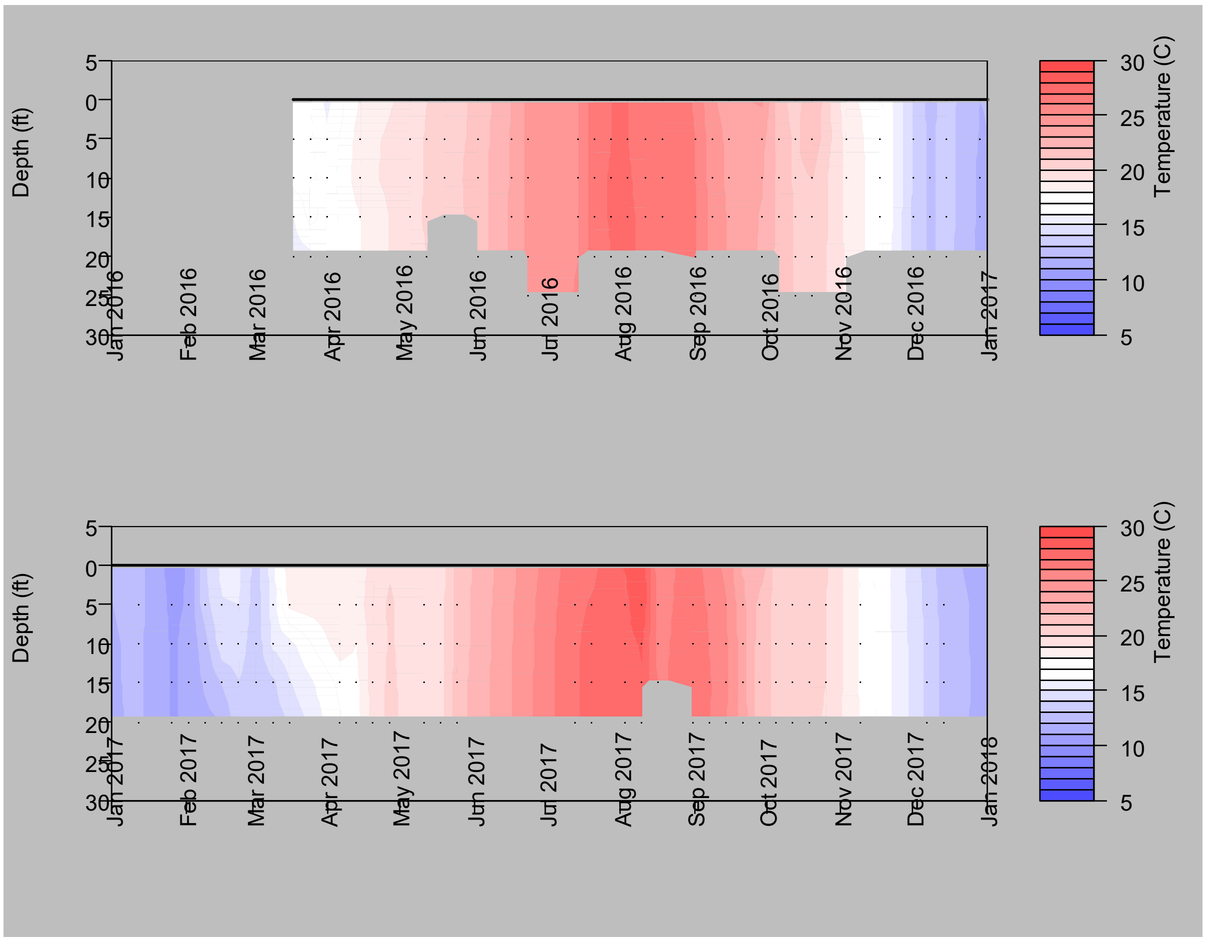


Figure 2-46a. Vertical profiles of water temperature in Lake Wohlford near the dam for the period 2016–2017.

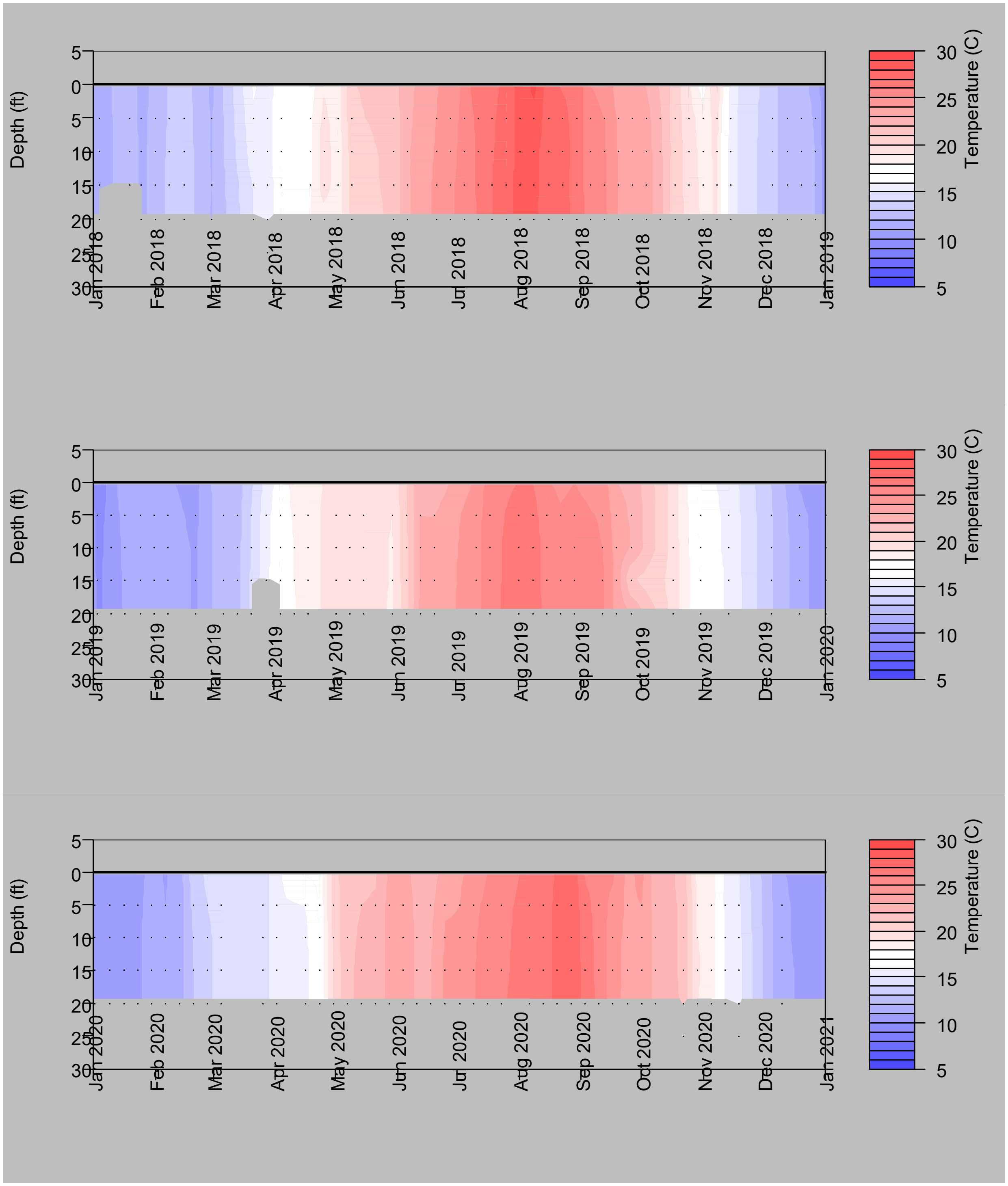


Figure 2-46b. Vertical profiles of water temperature in Lake Wohlford near the dam for the period 2018–2020.

Despite the lack of thermal stratification in Lake Wohlford, DO concentrations do exhibit chemical stratification in Lake Wohlford throughout the year, although the degree of stratification has generally been greatest in the spring (e.g., March–April 2016, March–April 2017, May 2018) when DO concentrations in surface waters ranged 10–12 mg/L and in bottom waters were at or near 0 mg/L (Figure 2-47a,b). Multiple brief periods of low DO (< 4 mg/L) in most or all of the water column have occurred in June 2016, September 2017, October 2018, June and August 2019, and September 2020 (Figure 2-47a,b).

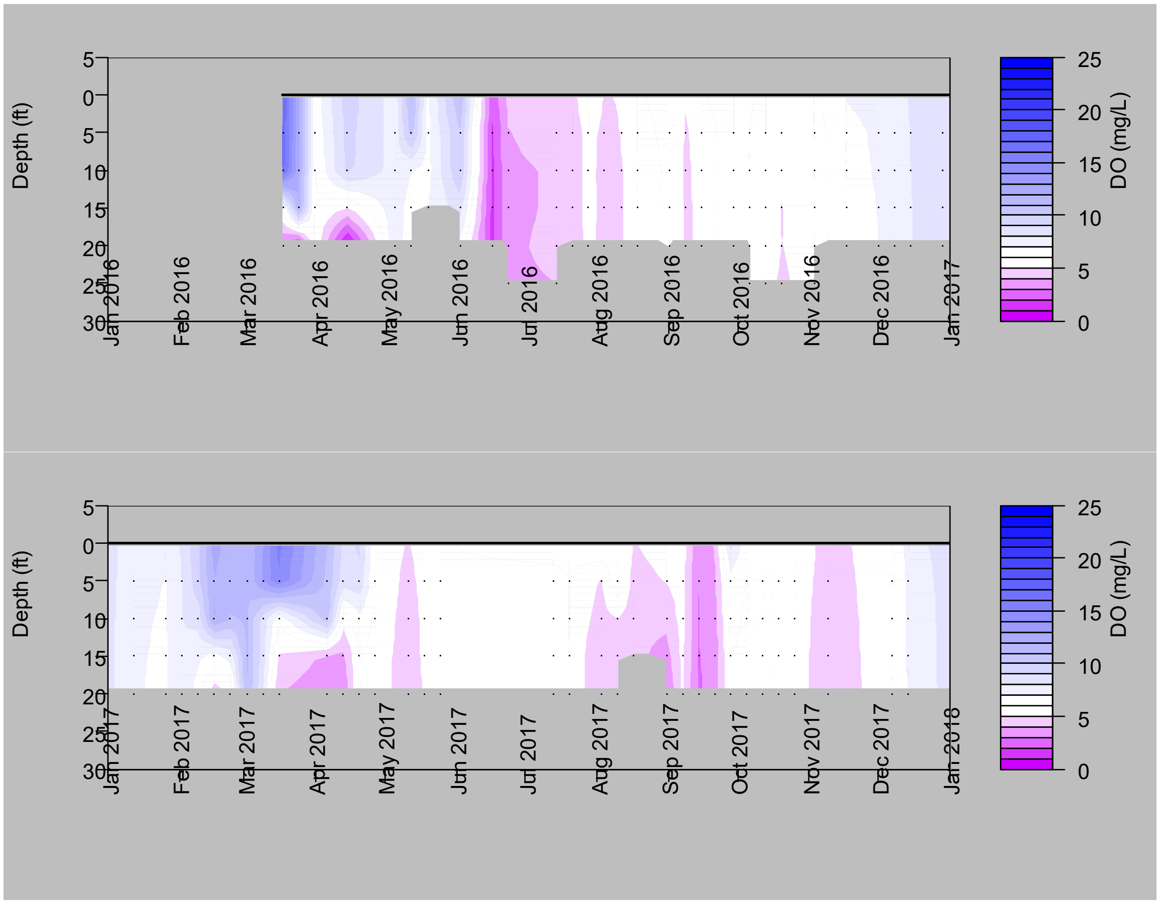


Figure 2-47a. Vertical profiles of dissolved oxygen (DO) in Lake Wohlford near the dam for the period 2016–2017.

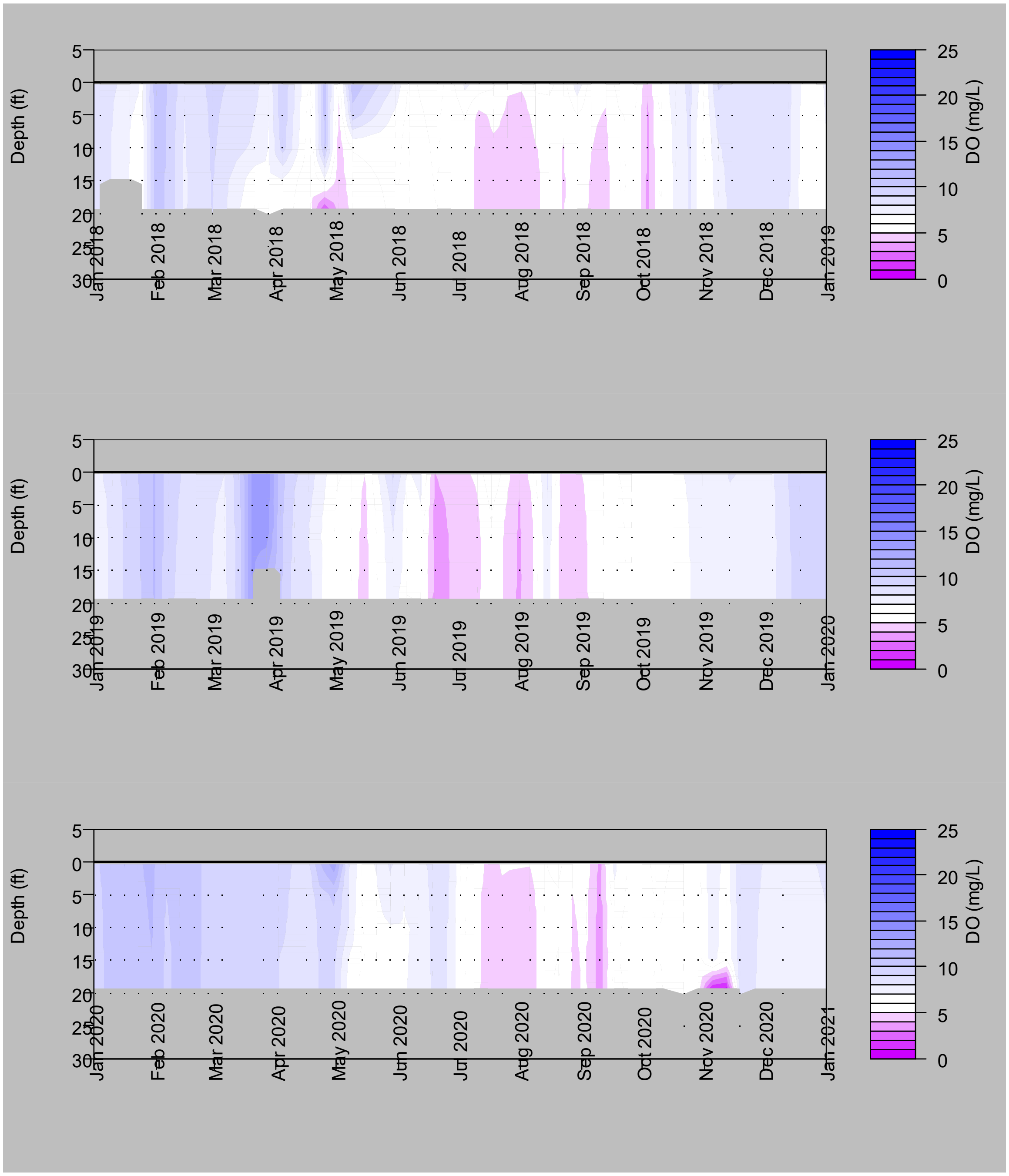


Figure 2-47b. Vertical profiles of dissolved oxygen (DO) in Lake Wohlford near the dam for the period 2018–2020.

The Lake Wohlford vertical profile DO data indicate that the artificial aeration system, despite running for 19 hours a day, seven days a week, is not efficiently transferring oxygen to the water column. This is not surprising since the air compressor technique used for Lake Wohlford, or any other standard artificial aeration-mixing system in lakes and reservoirs, supplies little oxygen directly to the water. Systems such as the one in Lake Wohlford generate coarse air bubbles (i.e., millimeters [mm] to a centimeter [cm] in diameter). The oxygen dissolution reaction is controlled by surface area between the gas and water and the partial pressure driving the gas from the bubble into the liquid. Injecting a plume of gas bubbles (i.e., 78% nitrogen and only 21% oxygen) into the water column creates a relatively small surface area for gas dissolution and does not provide efficient oxygen transfer into the water. Further, the coarse air bubbles that leave the aeration manifold or pipe rise rapidly through the water column and exit to the atmosphere in less than a minute. It is possible that over half of the oxygen in the compressed air commonly used in aeration-mixing systems is released back to the atmosphere at the water surface. That said, the bubbles do act as tiny lift pumps for entraining surrounding bottom water and lifting it to the surface. There, the relatively low DO water from the bottom of the lake can mix with high DO water that is generated by photosynthesizing algae at the lake surface before the water sinks back down towards the sediments. Thus, the primary means to bring oxygen to bottom waters using aeration-mixing systems is the creation of a vertical mixing cell in the water column that relies upon high DO in surface waters. However, the oxygen demand of algae that are producing high DO in surface waters during the day can often overwhelm the aeration system at night and even in deeper, shaded waters during the day; given the profile data from Lake Wohlford in 2016–2020, high oxygen demand appears to be overwhelming the existing aeration system.

2.4.2.2 pH

Quarterly pH data in Lake Wohlford are available for surface waters starting in 2007 and range 7.4–8.8 standard units (s.u.) (Figure 2-48). Most values (i.e., 30 of 33 or 90% of quarterly measurements) are below the Basin Plan instantaneous maximum water quality objective of 8.5 s.u. (RWQCB 2016). There is no apparent seasonal pattern in surface water pH (Figure 2-48).

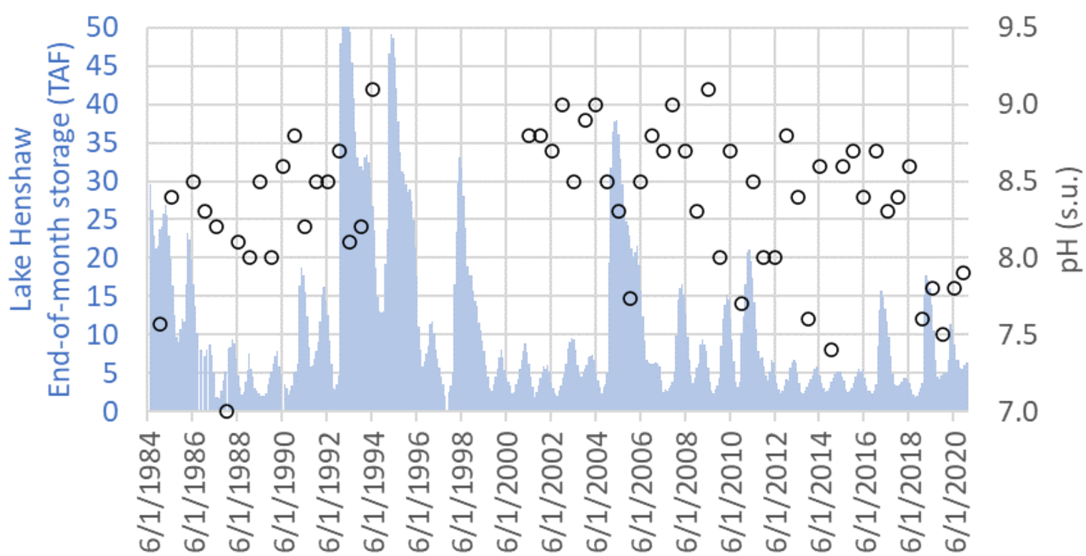


Figure 2-48. Monthly storage (acre feet [AF]; blue shading) and surface water pH (s.u.; open circles) collected in Lake Wohlford during the period 2003–2020 (pH data begin in 2007).

2.4.2.3 Nutrients

Nitrogen

Although recent data for the San Luis Rey River and the Escondido Canal indicate that TN, ammonia, and nitrate are transported downstream into Lake Wohlford when Henshaw Dam is releasing (Section 2.3.2.1), nitrate is the only nitrogen species for which historical and current data exist in Lake Wohlford. Prior to the decrease in water levels and reservoir storage in 2007, nitrate (NO_3^-) concentrations in Lake Wohlford surface waters ranged from less than 0.05 mg NO_3-N/L to 5.3 mg NO_3-N/L (average 0.9 mg NO_3-N/L; n=32; Figure 2-49). While there are no nitrate data in the upstream Lake Henshaw during 1997–2000, during 2001–2006 nitrate concentrations were generally below detection (< 0.04 to < 0.44 mg/L; Figure 2-16), suggesting that the source of higher nitrate to Lake Wohlford prior to 2007 was not Lake Henshaw. After water levels were lowered in Lake Wohlford, nitrate concentrations were generally lower, ranging from less than 0.05 mg NO_3-N/L to 1.1 mg NO_3-N/L (average 0.1 mg NO_3-N/L; n=32; Figure 2-49). The lower nitrate concentrations in recent years may be due to a decrease in nitrate entering the lake from another (unknown) source and/or periodically low DO concentrations in the water column (see Section 2.4.2.1) that can support microbial denitrification of nitrate to nitrogen gas (N_2).

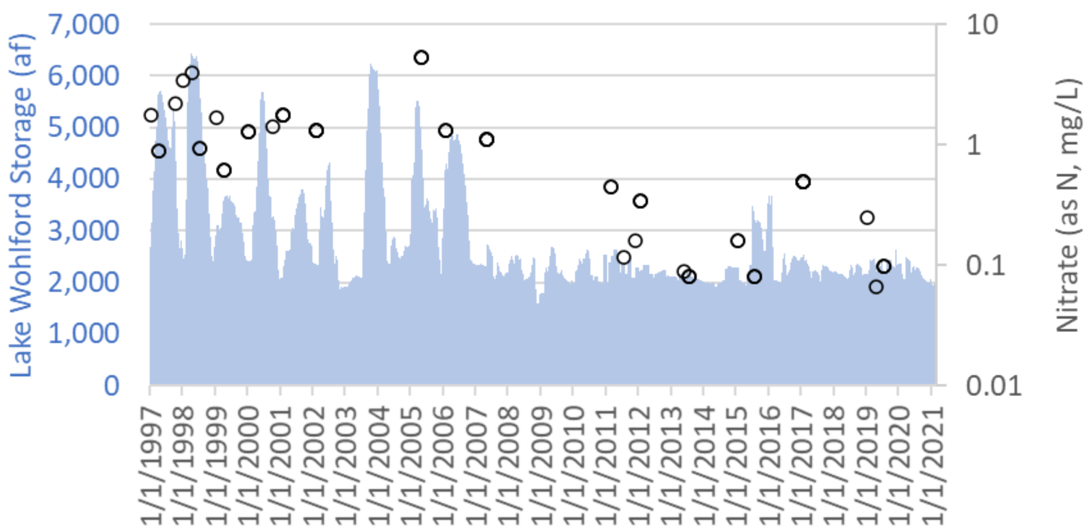


Figure 2-49. Monthly storage (acre ft [af]; blue shading) and nitrate (NO_3^- [mg/L]; open circles) collected from Lake Wohlford surface waters during the period 1997–2020.

Results from recent laboratory chamber studies of Lake Wohlford sediments under oxic (DO > 2 mg/L), hypoxic (DO < 2 mg/L), and anoxic (DO = 0 mg/L) conditions support the notion that denitrification is an important N-removal mechanism in Lake Wohlford. During the chamber experiments, nitrate release rates from some of the Lake Wohlford chambers were slightly negative (approximately -2 to -5 mg NO_3-N/m^2/d) under oxic and hypoxic conditions, which indicates that low levels of microbial denitrification were occurring in low-oxygen sediment microsites even when the water column was oxygenated (Beutel 2021). Under anoxic conditions, all chambers had highly negative nitrate fluxes (i.e., nitrate loss of approximately -10 to -30 mg NO_3-N/m^2/d), which is indicative of high levels of denitrification. During oxic and hypoxic periods, other chambers exhibited low to moderate nitrate releases from sediments (less than approximately 10 mg-N/m^2/d), where the likely mechanism was microbial nitrification[10] (Beutel 2021). Combined, the chamber studies indicate that both nitrification and denitrification are occurring in Lake Wohlford, potentially as coupled reactions in response to DO levels that vary throughout the day, season, and location in the lake. High algal productivity during the day combined with wind mixing may help to oxygenate the sediment-water interface in more shallow areas of the lake, whereas the deeper area near the dam remains suboxic or anoxic in bottom waters, despite the existing aeration-mixing system, and supports denitrification. The overall effect may be to keep concentrations of nitrate generally low in both surface and bottom waters.

Additionally, in sediment chambers from all three sites in Lake Wohlford, ammonia release from sediments was negligible under oxic and hypoxic conditions and elevated during anoxic conditions (Beutel 2021). Anoxic release rates were highest in the sediments from the deepest areas near the dam and in the mid-portion of the reservoir where sediment organic matter and nitrogen content were highest, and they were lowest in sediment collected from the shallow, eastern portion of the reservoir where sediment organic matter and nitrogen content were relatively lower. Based on the anoxic ammonia release rates, Lake Wohlford sediment has the potential to release 40–80 mg NH_4^+-N/m^2/d via mineralization of organic matter, or approximately 30–60 NH_4^+-N/m^2/d normalized to 20 ºC (Beutel 2021). While these ammonia release rates are lower than those observed in the Lake Henshaw chamber sediments, they are relatively high and are typical of hypereutrophic lakes (Beutel 2021).

Phosphorus

There are no historical or current data characterizing phosphorus species in Lake Wohlford, although recent data for the San Luis Rey River and the Escondido Canal indicate that TP and orthophosphate are transported downstream into Lake Wohlford when Henshaw Dam is releasing, with concentrations in the range of 0.04 to 0.4 mg/L and 0.01 to 0.1 mg/L, respectively (Section 2.3.2.1).

Recent laboratory chamber studies of Lake Wohlford sediments indicate that, similar to ammonia, orthophosphate release was negligible under oxic and hypoxic conditions and elevated during anoxic conditions (Beutel 2021). Also similar to ammonia, anoxic orthophosphate release from Lake Wohlford sediments was highest in the chambers collected from the deepest area near the dam where organic matter is relatively high and lowest in the chambers collected from the shallow area near the inlet where organic matter is lower. The chamber results suggest that the source of orthophosphate release was mainly reductive dissolution of Fe oxides in the sediments under anoxic conditions. The orthophosphate release rates (approximately 50–70 PO_4-P/m^2/d) were higher than orthophosphate releases for deeper eutrophic lakes (5–20 mg PO_4-P/m^2/d; Nurnberg 1994 as cited in Beutel 2021) and anoxic orthophosphate releases in shallow and broad Lake Elsinore, located in southern California (approximately 20 mg PO_4-P/m^2/d; Beutel 2000b). As orthophosphate release is mediated by biotic (e.g., manganese- and iron-reducing bacteria) and abiotic (i.e., sorption to metal oxides, precipitation of FeS) processes, the effects of water temperature on the Lake Wohlford orthophosphorus release rates are more difficult to determine and Beutel (2021) does not normalize them to 20 ºC.

Iron

Total iron concentrations in Lake Wohlford range 93–270 µg/L (n=14 for the period 2011–2015; MWH 2016), which is roughly 2–5 times lower than reported concentrations in upstream Lake Henshaw (Figure 2-20). Correspondingly, in recent laboratory chamber studies of Lake Wohlford

[10] Biological oxidation of ammonia (NH_4^+) to nitrite (NO_2^-) followed by the oxidation of nitrite to nitrate (NO_3^-) performed by small groups of autotrophic bacteria and archaea under oxic conditions.

sediments, iron release rates were generally low under oxic (DO > 2 mg/L), hypoxic (DO < 2 mg/L), and anoxic (DO = 0 mg/L) conditions (Beutel 2021). Iron release rates measured in the Lake Wohlford sediment chambers (<0.1 to 2 mg Fe/m^2/d) were one to two orders of magnitude lower than release rates measured in two eutrophic northern California reservoirs (i.e., 20−40 mg/m^2/d in Lafayette Reservoir [Beutel 2000a] and 5−20 mg/m^2/d in San Pablo Reservoir [Stillwater Sciences and Brown and Caldwell 2016]), as well as those measured in sediment from the eutrophic Lake Hodges Reservoir (San Diego, CA) at 20−80 mg/m^2/d (Beutel 2000b).

Although Lake Wohlford sediment release rates for iron, as measured in the sediment chambers, were relatively low, water column concentrations are in the aforementioned range are unlikely to be limiting algal growth in the lake.

Manganese

As noted for Lake Henshaw (Section 2.2.2.3), while manganese is considered to be an algal micronutrient, it is almost always sufficiently available for growth in freshwater systems. Manganese concentrations in Lake Henshaw inputs to Lake Wohlford ranged from < 10 μg/L to 30 μg/L with a median of 96 μg/L (n=70) for the period 1984–2020. Based on quarterly samples, the median concentration in Lake Wohlford for the period 1997–2020 was similar, at 85 μg/L (n= 63), although the maximum value in Wohlford was relatively higher at 400 μg/L. Without this singularly high value in August 2012, the maximum Lake Wohlford concentration was 150 μg/L, or approximately five times higher than the highest concentrations in the Lake Henshaw source water.

Reduced (soluble) manganese (Mn^{+2}) can occur in lake sediments under anoxic (i.e., DO = 0 mg/L) conditions, which could diffuse into the water column and increase total water column manganese concentrations. As the available data are limited to total manganese, it is not possible to determine whether water column stratification and subsequent anoxic conditions cause elevated concentrations of manganese in Lake Wohlford. Ninety percent (90%, or 57 of 63 measurements) were greater than the California secondary maximum contaminant limit (MCL) for manganese in drinking water (50 μg/L) for the period 1984–2020. Despite relatively higher concentrations in Lake Wohlford, EVWTP personnel report no operational issues associated with soluble manganese.

In recent laboratory chamber studies of Lake Wohlford sediments, manganese concentrations in the water column increased with time (i.e., flux from sediments at all sites was positive) under hypoxic (DO < 2 mg/L) and anoxic (DO = 0 mg/L) conditions (Beutel 2021), which is consistent with typical redox and mineral oxide dissolution patterns for this metal. Under oxic conditions, manganese concentrations decreased slightly with time (i.e., flux from sediments was negative) or did not change under oxic (DO > 2 mg/L) conditions. While there are limited data for comparison, manganese release rates measured in the Lake Henshaw Wohlford chambers (approximately 0.2 to 24 mg Mn/m^2/d) were 2 to 300 times lower than release rates measured in a eutrophic northern California reservoir (i.e., 50−60 mg/m^2/d in San Pablo Reservoir [Stillwater Sciences and Brown and Caldwell 2016]).

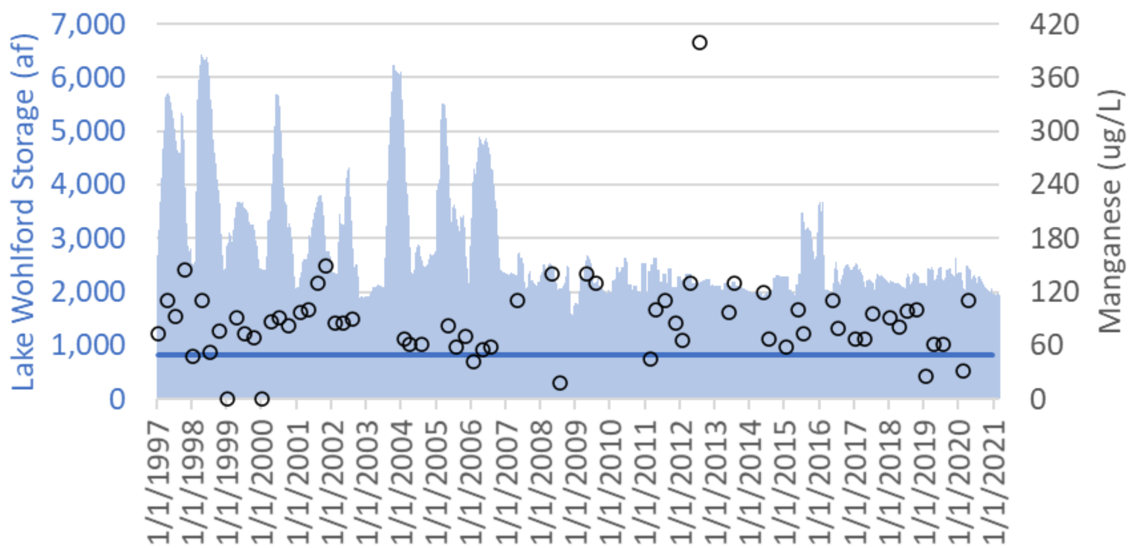


Figure 2-50. Monthly storage (acre ft [af]; blue shading) and manganese concentrations (ug/L; open circles) in Lake Wohlford during the period 1997-2021. Blue horizontal line is the manganese secondary MCL for drinking water (50 µg/L).

2.4.2.4 Cyanotoxins

Since May 2021, microcystin samples have been collected at three sites within Lake Wohlford (Figure 2-45; Table 2-9). Except for a brief period in early June 2021, microcystin concentrations have been below the analytical laboratory reporting limit (i.e., <0.3 µg/L; Figure 2-51). The elevated microcystin concentrations (0.5–0.7 µg/L) that occurred in June 2021 in Lake Wohlford coincided with releases from the upstream Lake Henshaw, although microcystin concentrations in Lake Henshaw at this time were lower at 0.3–0.4 µg/L, suggesting that internal production of cyanotoxins is occurring in Lake Wohlford. Based on the available data, there is no clear pattern of concentrations within Lake Wohlford (Figure 2-51). Note that the analytical laboratory reporting limit for microcystin in the Lake Wohlford samples is two times higher than that of the Lake Henshaw samples, although the laboratory has recently begun reporting data below the reporting limit, albeit with less certainty.

Since June 2021, when anatoxin-a concentrations began to be measured in Lake Wohlford, they have been consistently below the analytical laboratory reporting limit (i.e., <0.03 µg/L; **Figure 2-52**) at all three locations. Note that the laboratory reporting limit for anatoxin-a in the Lake Wohlford samples is five times lower than that of the Lake Henshaw samples.

Lake Wohlford is periodically treated with a chelated-copper based algaecide (Cutrine Plus). Escondido currently treats the lake when concentrations of microcystin exceed 0.3 µg/L. There was one treatment conducted in early July 2021, in response to elevated concentrations of microcystin in June 2021, although by the time of treatment microcystin concentrations had decreased in Lake Wohlford to below 0.3 µg/L (Figure 2-51).

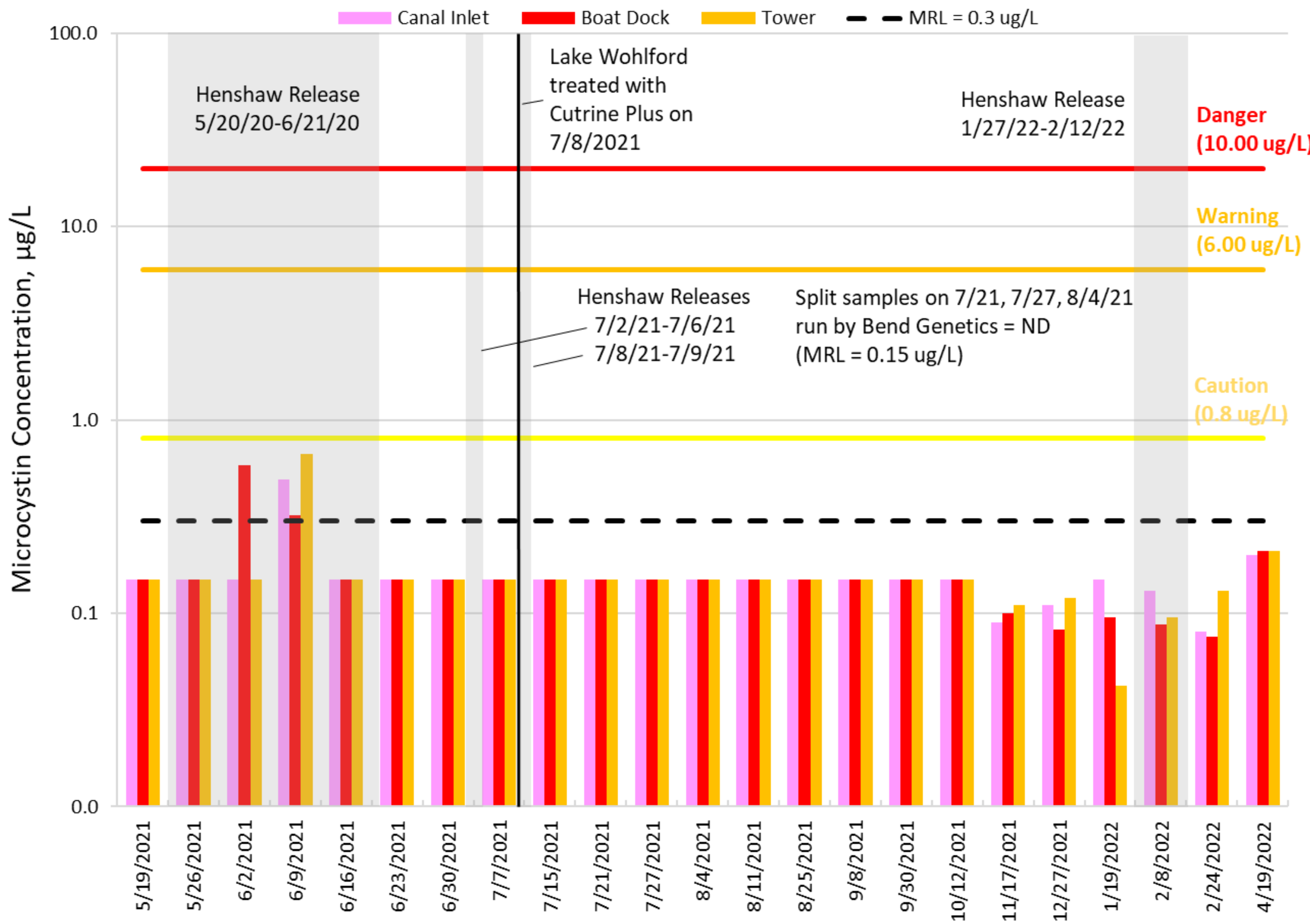


Figure 2-51. Microcystin concentrations in Lake Wohlford for the period May 2021 through May 2022.

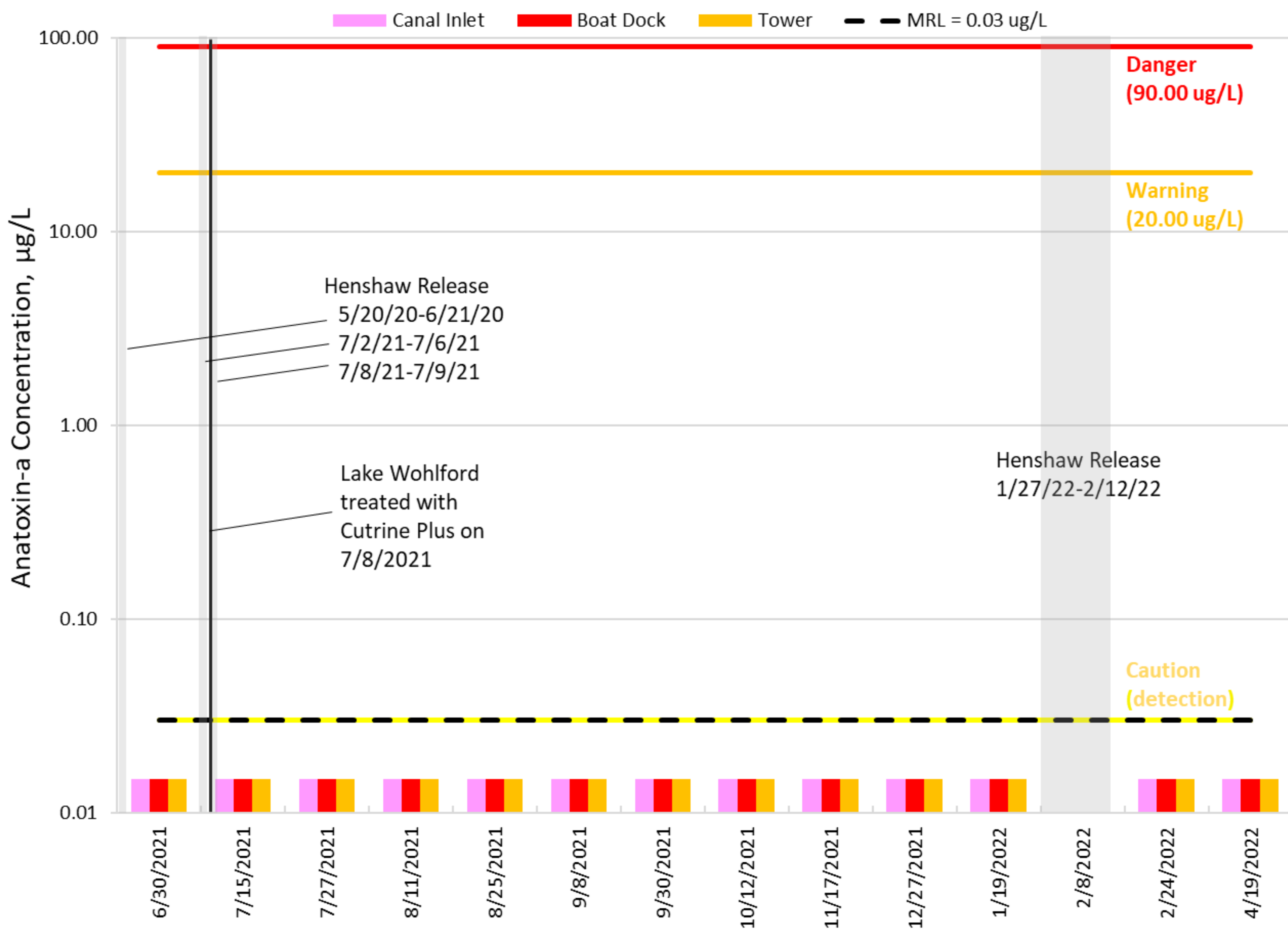


Figure 2-52. Anatoxin-a concentrations in Lake Wohlford for the period June 2021 through May 2022.

2.4.3 Algae community composition and dynamics

2.4.3.1 Planktonic (suspended) algae

For the period of record (2004–2011, part of 2016), green algae have tended to dominate the Lake Wohlford algal community, with diatoms as the second-most dominant genera and cyanobacteria as the third-most dominant. The relative abundance of each group varies seasonally in Lake Wohlford, with cyanobacteria increasing in abundance in late spring and summer months and receding in the winter, when green algae dominate the system. Diatom relative abundance tends to hold steady between and within years, and dinoflagellates have been generally rare.

The primary cyanobacterial genera observed in Lake Wohlford since February 2004 include: *Microcystis*, *Planktothrix*[11], *Aphanizomenon*, *Anacystis*[12], *Dolichospermum, Agmenellum, Coccochloris, Gomphosphaeria,* and *Lyngbya.* Across the sampling period, *Anacystis* was the most frequently detected genus. *Microcystis, Agmenellum,* and *Dolichospermum* were also regularly detected, whereas *Planktothrix* and *Aphanizomenon* were relatively rare.

Anacystis, *Dolichospermum, Planktothrix*, and *Aphanizomenon* are known to produce microcystin, while *Aphanizomenon* and *Dolichospermum* are known to produce anatoxin-a.

Aphanizomenon and *Dolichospermum* are able to fix nitrogen from the atmosphere when supply of this nutrient is low in lake water. These two genera can form akinetes, which are resilient, thick-walled, non-motile cells that allow cyanobacteria to survive harsh environmental conditions and periods of extended dormancy. Their ability to form akinetes means that they are likely to be fairly resilient as members of Lake Wohlford cyanobacterial community.

Recent algal species identification is not available for Lake Wohlford, and thus associations between cyanobacteria genera and cyanotoxin production is not currently possible. Visible evidence of blooms regularly occurs (Figure 2-54).

[11] Some species of *Planktothrix* were previously grouped within the genus *Oscillatoria*, but based on more recent scientific identification work, *Planktothrix* has been defined as its own genus.
[12] *Anacystis* is not presently a commonly identified genus. *Anacystis* identifications can overlap with identifications of *Synechococcus*, *Microcystis, Aphanothece*, and *Aphanocapsa*.

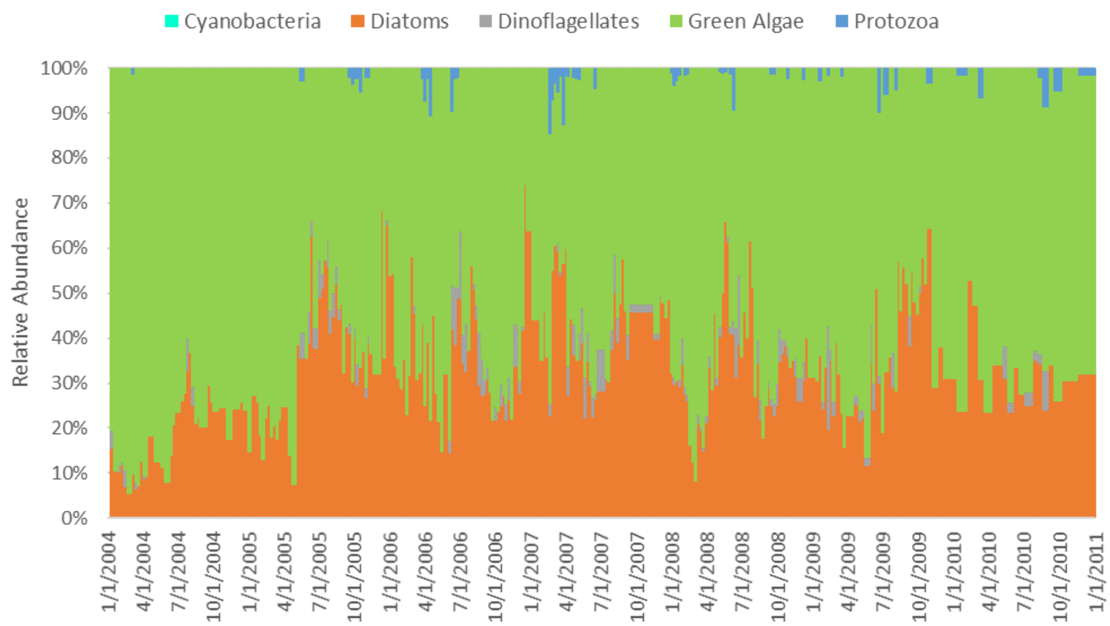


Figure 2-53. Planktonic (suspended) algae types in Lake Wohlford measured roughly weekly for the period 2004-2021.

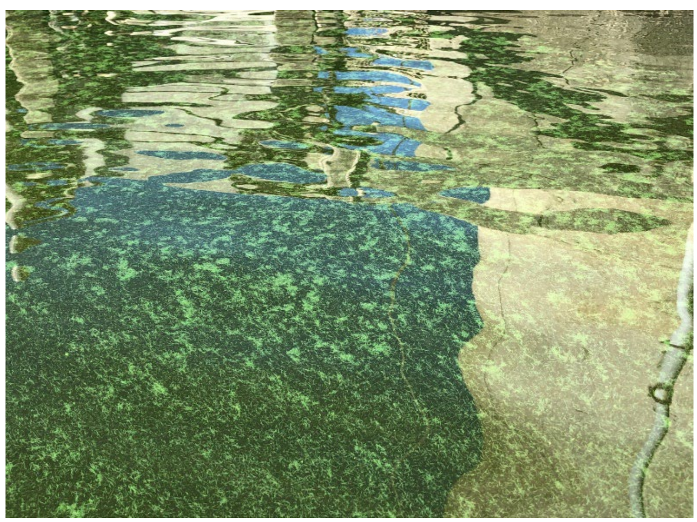


Figure 2-54. Planktonic (suspended) algae bloom in Lake Wohlford.

2.4.3.2 Benthic (attached) algae

Benthic algae are attached species associated with sediments and other bottom substrates (e.g., rocks, rooted aquatic macrophytes) within the photic zone of a waterbody, such as shallow areas near the shoreline. Benthic algae can be an important component of overall primary productivity in lakes and reservoirs. Although there are no available benthic algae data for Lake Wohlford, they may be contributor to production of cyanotoxins (see Section 2.4.2.4).

2.5 Existing Conditions Conceptual Model

Based on available data, several factors influence HABs production in Lake Henshaw and Lake Wohlford, forming the basis of an existing conditions conceptual model (Figure 2-55). The key factors are listed below by season (i.e., spring–summer–fall and winter).

2.5.1 Spring–summer–fall

2.5.1.1 Lake Henshaw

- Watershed runoff during spring, summer, and fall (typically March–November) is generally negligible, although runoff can occur in March and April. Pumped groundwater transfers from the Warner Ranch Wellfield in spring, summer, and fall contribute nitrate (NO_3^-), orthophosphate (PO_4^{3-}), and iron (Fe) to Lake Henshaw in concentrations sufficient to stimulate cyanobacterial blooms.
- Information regarding seasonal thermal stratification in Lake Henshaw is limited (Appendix B). Bottom waters and the sediment-water interface are assumed to regularly undergo periods of oxic (DO > 2 mg/L), hypoxic (DO < 2 mg/L), and anoxic (DO at or near 0 mg/L) conditions during spring, summer, and fall. Surface waters may become supersaturated with DO due to photosynthetic production by phytoplankton in the photic zone, although data are needed to confirm this. DO may be depleted during aerobic respiration in shaded deeper waters and at night. Sediments in the deepest part of the lake (near the dam) may undergo relatively longer and/or more frequent periods of hypoxia and anoxia than sediments in shallower areas.
- Beginning in spring and continuing through fall, periods of low DO occurring at the sediment-water interface likely facilitate biogeochemical processes that release bioavailable nutrients (orthophosphate [PO_4^{3-}], ammonia [NH_4^+], dissolved manganese [Mn^{2+}], and dissolved iron [Fe^{2+}]) from organic-rich bottom sediments. Based on recently conducted experimental sediment chamber studies, ammonia and orthophosphate release rates from Lake Henshaw sediments are high compared to those reported for Lake Wohlford and other hypereutrophic lakes. Internal nutrient loading, particularly in deeper areas of the lake, is likely to contribute a much larger proportion of bioavailable nutrients to the lake than does external loading (i.e., groundwater transfers or runoff) under existing conditions.
- High concentrations of bioavailable nutrients in the Lake Henshaw water column and warm water temperatures support large planktonic (suspended) cyanobacteria blooms throughout the shallow water column. The highest cyanotoxin concentrations in the lake tend to coincide with the highest cyanobacteria cell counts (remotely sensed). The primary microcystin producer may be *Microcystis sp.* and the primary anatoxin-a producer may be *Dolichospermum sp.*, although genetic studies are necessary to confirm the primary producers of cyanotoxins in Lake Henshaw. It is currently unknown whether benthic algae are contributing to cyanotoxin production in Lake Henshaw.
- Following the peak of the cyanobacteria bloom, algae cells settle to the bottom sediments, contributing organic matter, nutrients, and potentially persistent cyanotoxins to the sediments, and contributing to future internal loading (see above).

2.5.1.2 San Luis Rey River and Escondido Canal

- Water released from Lake Henshaw transports nutrients (i.e., orthophosphate [PO_4^{3-}], ammonia [NH_4^+], nitrate [NO_3^-], dissolved manganese [Mn^{2+}], and dissolved iron [Fe^{2+}])

downstream into the San Luis Rey River and Escondido Canal, but these nutrients do not appear to undergo substantial transformation during the short transit time (i.e., hours to days) to Lake Wohlford.

- Cyanobacteria and cyanotoxins, including microcystin and anatoxin-a, produced in Lake Henshaw are transported downstream when water is released from Lake Henshaw, but do not appear to be altered during the short transit time (i.e., hours to days) to Lake Wohlford.

2.5.1.3 Lake Wohlford

- Watershed runoff during spring, summer, and fall (typically March–November) is generally negligible. Water transfers from the Lake Henshaw releases, via the San Luis Rey River and Escondido Canal, contribute ammonia (NH_4^+) nitrate (NO_3^-), orthophosphate (PO_4^{3-}), and iron (Fe) to Lake Wohlford in concentrations sufficient to stimulate additional cyanobacterial bloom formation in the downstream waterbody. Water transfers also contribute cyanobacteria and cyanotoxins to Lake Wohlford when these constituents are present in the release water.
- Occasional brief periods of thermal stratification can occur in Lake Wohlford throughout the year, although the majority of the time the water column is isothermal (i.e., same temperature in surface and bottom waters). The regular lack of thermal stratification in the deepest Lake Wohlford waters is likely due to the artificial aeration system that, while not directly transferring much oxygen to the lake, mixes the water column near the dam for 19 hours a day, 7 days a week.
- DO concentrations do exhibit chemical stratification in Lake Wohlford throughout the year, although the degree of stratification has generally been greatest in the spring. Multiple brief periods of low DO (< 4 mg/L) in most or all of the water column can occur in summer and fall.
- Moderate concentrations of nitrate in the Lake Wohlford water column and warm water temperatures contribute to large planktonic (suspended) cyanobacteria blooms throughout the shallow water column. Genetic studies are necessary to confirm the primary producers of microcystin in Lake Wohlford. It is currently unknown whether benthic algae are contributing to cyanotoxin production in the lake.
- Following the peak of the cyanobacteria bloom, algae cells settle to the bottom sediments, contributing organic matter, nutrients, and potentially persistent cyanotoxins, to the sediments, and contributing to future internal loading (see above).

2.5.2 Winter

2.5.2.1 Lake Henshaw

- Watershed runoff is highest in winter (December–February) although it is variable by year and information regarding associated nutrient inputs is not currently available. Pumped groundwater transfers from the Warner Ranch Wellfield in winter continue to contribute nitrate (NO_3^-), orthophosphate (PO_4^{3-}), and iron (Fe) to Lake Henshaw in concentrations sufficient to stimulate cyanobacterial blooms once sunlight intensity increases in spring and water temperatures warm.
- Information regarding seasonal thermal stratification in Lake Henshaw is limited (Appendix B). Bottom waters and the sediment-water interface are assumed to primarily experience oxic (DO > 2 mg/L) conditions during winter. Surface waters can occasionally

become supersaturated with DO due to continued photosynthetic production by phytoplankton in the photic zone (Appendix B). DO may be depleted during aerobic respiration in shaded deeper waters and at night, even given lower water temperatures, although data are needed to confirm this.

- Infrequent periods of low DO occurring at the sediment-water interface may facilitate biogeochemical processes that release bioavailable nutrients (orthophosphate [PO_4^{3-}], ammonia [NH_4^+], dissolved manganese [Mn^{2+}], and dissolved iron [Fe^{2+}]) from organic-rich bottom sediments, although to a lesser extent than in spring, summer, and fall. Internal nutrient loading, particularly in deeper areas of the lake, is likely to contribute a larger proportion of bioavailable nutrients to the lake than does external loading from groundwater transfers under existing conditions, although in years with heavy runoff external loading from runoff may be the highest.
- Moderate concentrations of bioavailable nutrients in the water column support cyanobacterial productivity, though growth is modulated by relatively lower water temperatures and sunlight limitations. Algal blooms can form, but their intensity and frequency is reduced compared to those in spring, summer, and fall. Cyanobacteria may produce low to moderate concentrations of microcystin and anatoxin-a. Cyanotoxins can be transported downstream if present when water is released from Lake Henshaw.
- Following the peak of the cyanobacteria bloom, algae cells settle to the bottom sediments, contributing organic matter, nutrients, and potentially persistent cyanotoxins to the sediments and contributing to future internal loading (see above).

2.5.2.2 Lake Wohlford

- Watershed runoff (including runoff into the San Luis Rey River downstream of Lake Henshaw and above the Diversion Dam) is highest in winter (December–February) although it is variable by year. Information regarding associated nutrient inputs is not currently available. Water transfers from the Lake Henshaw releases, via the San Luis Rey River and Escondido Canal, in winter are not typical, but could contribute nitrate (NO_3^-), orthophosphate (PO_4^{3-}), and iron (Fe) to Lake Henshaw in concentrations sufficient to stimulate cyanobacterial blooms once sunlight intensity increases in spring and water temperatures warm.
- Occasional brief periods of thermal stratification can occur in Lake Wohlford even in winter, although the majority of the time the water column is isothermal (i.e., same temperature in surface and bottom waters). Artificial aeration-mixing, while not directly transferring much oxygen to the lake, mixes the water column near the dam for 19 hours a day, 7 days a week during winter months.
- Low DO concentrations are not typical during winter.
- Following the peak of the bloom, decomposition of senescent algae contributes organic matter and nutrients to the system, consuming dissolved oxygen in the water column in the process.

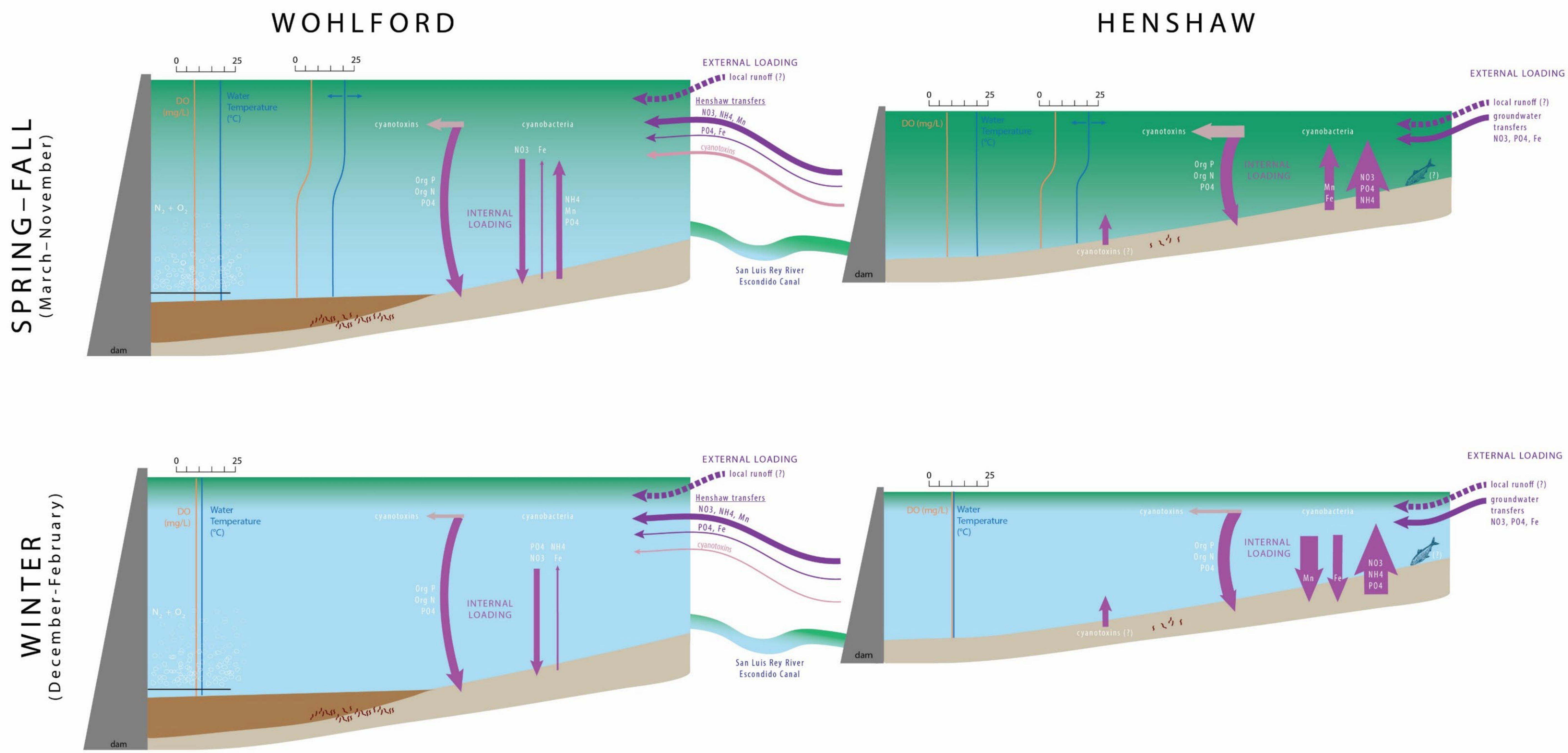


Figure 2-55. Conceptual model illustrating hypothesized seasonal water temperature and dissolved oxygen conditions and nutrient loading in lakes Henshaw and Wohlford.

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3 CONSIDERATION OF POTENTIAL SHORT-TERM MITIGATION METHODS

3.1 Screening of Potential Short-term Methods

During March and April of 2021, the Project Team[13] conducted a site visit and compiled and reviewed existing hydrology and water quality data for the Warner Ranch, lakes Henshaw and Wohlford, the San Luis Rey River, and the Escondido Canal. On April 22, 2021, the Project Team participated in a screening workshop to evaluate 18 potential in-lake management methods for their applicability in addressing HAB occurrences in Lake Henshaw in the short term, and to narrow the list of potential mitigation and treatment methods to one or two approaches that would be most suitable for implementation in 2021. Initial consideration of a broad list of potential management methods was undertaken to ensure that the project did not inadvertently overlook possible viable approaches for short term application.

Screening of short-term HAB mitigation and treatment alternatives discussed at the April 2021 workshop focused on implementation in Lake Henshaw because existing data suggest that while Lake Wohlford can produce cyanobacteria and cyanotoxins *in situ*, HABs in Lake Henshaw can produce high levels of cyanotoxins that are transported downstream through the San Luis Rey River, Escondido Canal, and into Lake Wohlford, including water that is intended to meet scheduled deliveries to the Indian Water Authority (see also Section 2.3). The 18 potential in-lake management methods considered at the screening workshop for implementation in Lake Henshaw are listed below.

Physical Methods

- Dredging
- Water level fluctuations
- Mixing and/or destratification
- Macrophyte harvesting
- Wetland filters on lake margins
- Algae harvesting/separation/skimming
- Selective withdrawal
- Dilution/flushing
- Sediment sealing fabrics
- Ultrasonic waves

Chemical Methods

- Algaecides
- Oxygenation/aeration using ultrafine or nanobubbles
- Shading/dyes
- Sediment sealing

Biological methods

- Pathogens/diseases of algae
- Grazers (on algae or macrophytes)
- Nutrient harvesting from fish/weeds
- Biomanipulation

Each of the in-lake/reservoir methods considered at the screening workshop is summarized in Table 3-1, including relevant existing information and additional information needs as understood at the time of the workshop (April 2021).

[13] The Project Team included staff from the District and Escondido involved in managing Lake Henshaw, the Escondido Canal, and Lake Wohlford, and members of the Stillwater Sciences' Team. The Project Team met with members of the La Jolla Band and the Indian Water Authority at the La Jolla Campground as part of the site visit. The Stillwater Sciences' Team included Stillwater Sciences, Brown and Caldwell, Alex Horne Associates, Robertson-Bryan Inc., University of California at Merced, Marine Biochemists, Water Quality Solutions, and Mr. Bill Taylor.

Table 3-1. Potential HAB mitigation and treatment methods considered at a screening level for short-term application in Lake Henshaw.

	Method (P/M) [1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Likely Permitting Timeframe	Potential to Implement in Short term (2021–2023)
Physical Controls						
1	Dredging (P)	• Removal of nutrients (primarily phosphorus) contained in lake sediments • Reduce nutrient internal loading to reservoir	• Requires extensive permitting • Costs would be prohibitively expensive	• Potential for phosphorus limitation of HABs • Spatial distribution of nutrients (primarily phosphorus) in lake sediments	2–3 years	No
2	Water level fluctuation (P)	• Macrophyte control • Oxidization of littoral sediments to improve redox conditions and reduce nutrient flux out of sediments	• Already occurs seasonally due to water supply • Dry winters when lake water level is low may increase HABs but potential to increase water level using pumped groundwater is limited relative to the much higher runoff volumes delivered during wet winters • Groundwater nutrient levels may be relatively high, such that adding groundwater during dry winters may be counterproductive	N/A	N/A	No
3	Mixing and/or destratification (P/M)	• Mix water column via macro bubbles (1-2 mm) or vigorous epilimnetic mixing (VEM) • Large colonial cyanobacteria can be outcompeted by single filament species and/or diatoms	• Bubbler or mixing arrays would need to be very large (e.g., 1,000 to 2,000 acres); not feasible in short term.	• Water column thermal and chemical stratification patterns to confirm stratification is an issue	N/A	No

Method (P/M) [1]		Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Likely Permitting Timeframe	Potential to Implement in Short term (2021–2023)
4	Macrophyte harvesting (P)	• Nutrient removal via plant harvest	• Macrophytes not a concern in this lake, cyanotoxins are likely caused by phytoplankton rather than epiphytic algae growing on macrophyte surfaces	N/A	N/A	No
5	Wetland filters (fringe) (P)	• Algae, nutrient removal via nutrient uptake and transformations in constructed wetlands	• Requires design, siting, permitting	N/A	2–3 years	No
6	Algae harvesting, separation, skimming (P, M)	• Physical removal of algae and nutrients	• Variable costs and methodologies; each may require various permits • \$500k ~ \$10M capital; method and flowrate dependent	• Additional understanding regarding applicability of various harvesting methods • Does not address dissolved cyanotoxins and subsequent treatment would be required	2–3 years	No
7	Selective withdrawal (M)	• Control which parcels of water are released from reservoir to minimize downstream nutrient and/or toxin transport	• Reservoir is shallow, outlet already minimizes toxin export by drawing from bottom waters	• Lack of information on seasonal stratification in lake	2–3 years	No
8	Dilution/ flushing/ bypass (P/M)	• Dilution/flushing would decrease residence time of water in lake to prevent or limit HABs by creating conditions unsuitable for cyanobacteria • Bypass would route cyanotoxin-free water around the lake to meet entitlements and other water supply needs	• High volume/low nutrient flow not available for dilution or flushing (groundwater nutrient levels may be relatively high, such that adding groundwater during dry winters may be counterproductive) • Bypass would reduce evaporative losses of pumped groundwater which could offset costs of construction and operation	• No recent data on nutrient inputs from local creeks	N/A	No

	Method (P/M) [1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Likely Permitting Timeframe	Potential to Implement in Short term (2021–2023)
9	Sediment sealing/ capping (fabrics) (P)	• Reduce phosphorus release from lake sediment	• Prohibitively expensive	• Potential for phosphorus limitation of HABs • Spatial distribution of nutrients (primarily phosphorus) in lake sediments	2–3 years	No
10	Ultrasonic waves (P)	• Render cyanobacteria unable to control their position in the water column	• Ultrasonic save buoy arrays would need to be very large (e.g., n=30 across 1,000 to 2,000 acres)	• Whether ultrasonic waves are sufficient as a single strategy or needs to be combined with other mitigation strategies	N/A	No
Chemical Controls						
11	Algaecides (M)	• Effective in destroying HAB-causing algae and potentially also toxins, if used properly	• Can be immediately effective, but effects typically last 2-3 weeks, follow up applications may be necessary • Low capital investment product cost varies by method, \$40K to \$250K	• Additional algal data needed for early warning to trigger algaecide treatment; spatial distribution of HAB needed to determine algaecide treatment locations	District obtained permit for Warner Ranch Wellfield and Lake Henshaw in June 2021; Escondido obtained permit for Lake Wohlford in 2013	**Yes**

	Method (P/M) [1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Likely Permitting Timeframe	Potential to Implement in Short term (2021–2023)
12	Oxygenation via Speece Cone/aeration using ultrafine or nano-bubbles (P)	• Speece Cone increases water column *and* sediment dissolved oxygen (DO) without destratification • Aeration/oxygenation using ultrafine bubbles (1–100 um diameter) or nanobubbles (< 200 nanometers [nm] diameter) increase water column DO without destratification • Nanobubbles are neutrally buoyant and stable in water column Ozone (O_3) input to bubblers can deactivate cyanotoxins	• Bubbler arrays would need to be very large (e.g., n=30 across 1,000 to 2,000 acres) • Placement of cone or aerators/bubblers in lake near dam in deepest portion of reservoir or on western shore, may not be able to oxygenate entire lake water column and/or sediments such that HABs could persist in shallow areas or along shorelines • Nanobubblers and use of O3 in ultrafine or nanobubblers are experimental; these approaches have not been tested at the scale of Lake Henshaw	• Lack data on spatial extent of blooms in lake; unknown whether anoxia in sediments and internal nutrient loading are drivers of HABs in this system	N/A	No
13	Shading/ dyes (P)	• Effective by shading algae and preventing or reducing growth	• Not feasible in large water bodies	N/A	N/A	No
14	Chemical sediment sealing (P)	• Particles (e.g., alum, Phoslock™) bind with phosphorus, algae, and/or detritus and are then removed from water column; settled particles form barrier on sediment surface to reduce phosphorus flux out of sediments	• Can be immediately effective but may require several applications	• Possible jar-testing necessary to determine application rates	6-12 months	No for 2021–2022, Maybe for 2023 if permitting started in 2022

	Method (P/M) [1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Likely Permitting Timeframe	Potential to Implement in Short term (2021–2023)
Biological Controls						
15	Pathogens/ diseases of algae (P)	• Alteration of food web and trophic linkages to reduce abundance of HAB-causing algae	• Requires detailed understanding of aquatic food web	• Not applicable in short term	N/A	No
16	Grazers (on algae or macro-phytes) (P)	• Consumption of macrophytes or HAB-causing algae	• Experimental • No macrophyte issue; algal grazing may be sufficiently improved by other mitigation methods		N/A	No
17	Nutrient harvesting from fish/weeds (P)	• Removal of nutrients from system via removal of organisms	• Experimental • Not recommended as single method; minimal removal of nutrients annually		N/A	No
18	Biomanipu-lation (P)	• Alteration of food web and trophic linkages to reduce abundance of HAB-causing algae	• Requires detailed understanding of aquatic food web		N/A	No

[1] "P" denotes a method designed to prevent HABs formation, "M" denotes a method designed to mitigate the effects of HABs formation.

3.2 Selected Mitigation Methods

Algaecides were selected by the Project Team as the most feasible short-term HABs control method for Lake Henshaw for the following reasons (see also Table 3-1):

- Algaecide application is a well-proven mitigation method for HABs. Approved algaecide chemicals act quickly (i.e., minutes to hours) and can prevent the formation of and interrupt an ongoing HAB and to stop cyanotoxin production. Some active ingredients can also destroy cyanotoxins in the water column (e.g., hydrogen peroxide).
- Little to no capital investment is required for algaecide application, since licensed applicators can be hired by the District to apply the chemicals and undertake monitoring needed to meet permit requirements.
- Costs are generally predictable and there are multiple algaecide products available on the market.
- In June 2021, the District obtained a Statewide Aquatic Weed Control Permit for application of copper sulfate, chelated copper, and sodium carbonate peroxyhydrate (peroxide) to control algae/cyanobacteria in Lake Henshaw.

The remainder of this section is focused on planning details related to algaecide application.

3.2.1 Algaecide application goals and objectives

The goal of applying algaecide to surface waters, including reservoirs, lakes, ponds, and streams, is to control the formation and growth of nuisance algae blooms (filamentous, planktonic, benthic, or cyanobacteria) by killing the organisms responsible for poor water quality.

In the context of HABs, the objectives of algaecide application to Lake Henshaw in the short term are the following:

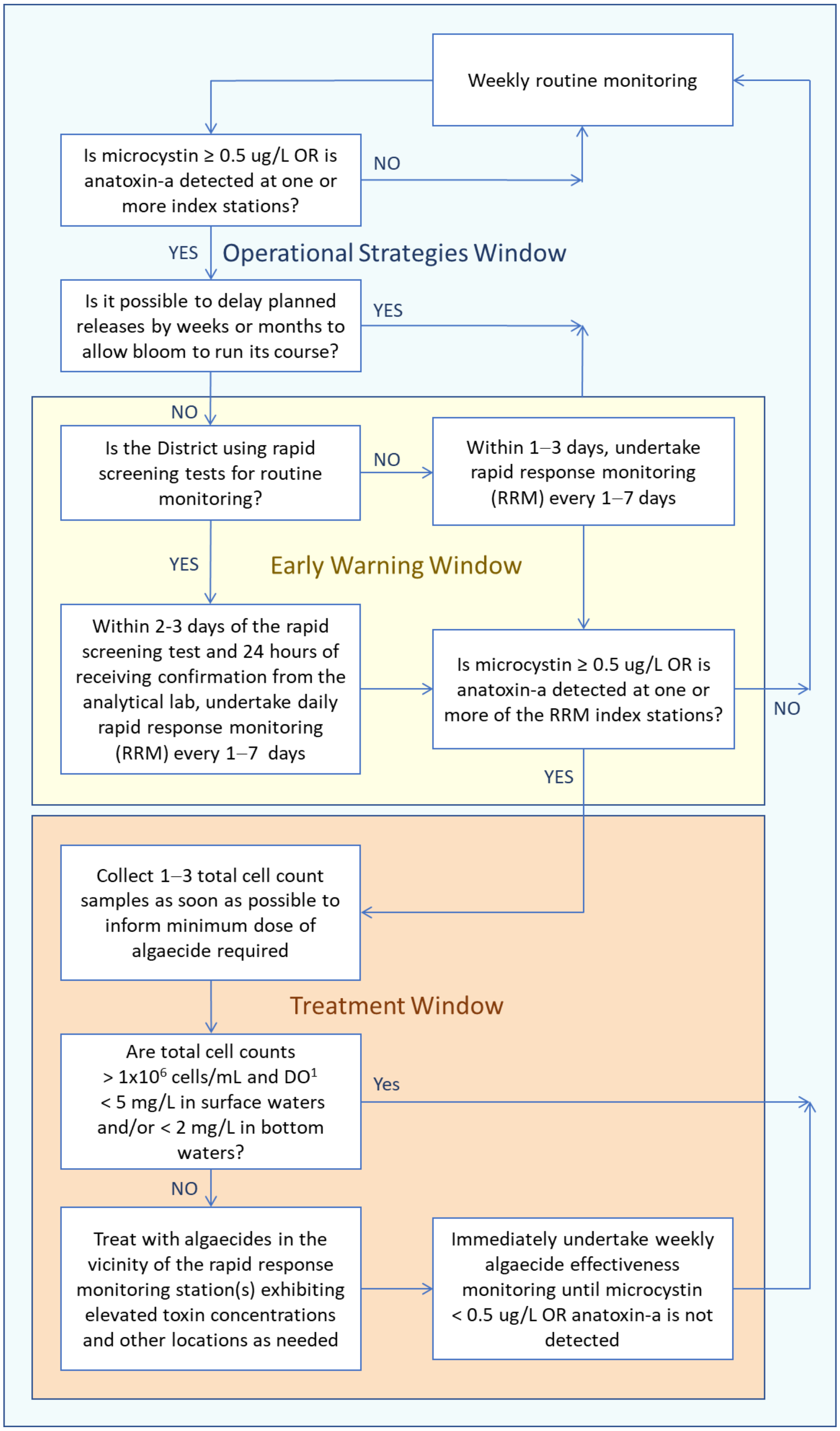
- To apply one or more approved chemicals to surface waters sufficiently early in a cyanobacteria bloom and at an appropriate application rate and location(s) to quickly and effectively reduce the bloom strength and thus avoid elevated concentrations of cyanotoxins.
- To be prepared with the requisite equipment and permits in place to deploy algaecide on short notice (i.e., 1–3 days) given that HABs and associated cyanotoxin events can develop rapidly.
- To undertake adequate monitoring before, during, and after an algaecide application event to identify where production is occurring in the lake to allow for targeted treatment, to provide an early warning that a HAB may be developing, to track the response of the HAB to algaecide treatment for adaptive learning and future management, and to meet the monitoring and reporting requirements of the application permit (e.g., eliminate chemical residuals in discharge waters).
- To obtain experience with the use of both copper- and peroxide-based algaecides in the lake over time.

3.2.2 Triggers for short-term algaecide application

In general, the operational triggers for short-term algaecide application in Lake Henshaw will be based on reservoir management windows of opportunity, including an *operational strategies window*, before a HAB occurs and when multiple options for reservoir operation are still available, to an *early warning window*, when monitoring data suggest that a HAB may be developing, and, as needed, to a *treatment window*, when algaecide application would occur prior to a HAB becoming out of control (Figure 3-1).

Currently, the District is obtaining experience with lake response to algaecide treatment. Thus, while triggers for moving into the *treatment window* may be assessed through the results of rapid response monitoring at multiple sites in Lake Henshaw, where meeting particular concentration thresholds for anatoxin-a or microcystin will trigger algaecide use, the District may also decide to treat Lake Henshaw based on the results of routine monitoring and/or other operational considerations.

Other water quality parameters/constituents (e.g., water temperature, dissolved oxygen, total and/or cyanobacteria cell counts, identification of potential toxin producing species) will be collected from Lake Henshaw along with cyanotoxin concentrations to inform the minimum dose of algaecide is anticipated to be effective. Details are provided as part of the Water Quality Monitoring Plan in Section 5.5.



[1] Dissolved oxygen (DO) monitoring results on the day of planned treatment or one day prior.

Figure 3-1. Cyanotoxin routine, rapid response, and algaecide effectiveness monitoring framework and operational triggers flowchart for Lake Henshaw. Operational strategies window (blue), early warning window (yellow), and treatment window (orange) are discussed in Section 5.1. Thresholds for particular water quality parameters and constituents are discussed in Section 5.5.2.3.

3.2.3 Application methods and algaecides expected to be used

Short-term mitigation for HABs in Lake Henshaw will involve algaecide treatment in the vicinity of the site(s) exhibiting elevated toxin concentrations. Treatment will occur within 1−3 days of the trigger.

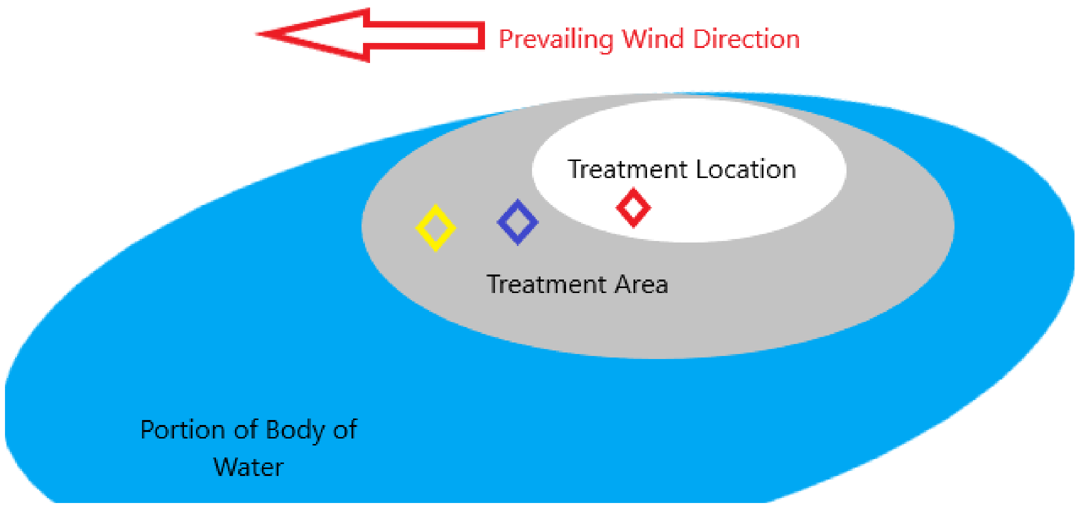


Figure 3-2. Lake Henshaw treatment event monitoring locations. Red diamond represents background (BG) sampling location; Blue diamond represents event monitoring (EM) sampling location; Yellow diamond represents post-event (PE) monitoring location. Note that the sampling locations are dependent on prevailing wind direction and/or current direction for subsurface applications. Source: Marine Biochemists (2021).

The District will apply various forms/formulations of algaecides (Table 3-2) with active ingredients including copper and hydrogen peroxide. Applications will be made by boat using surface and/or subsurface spray and/or injection methods, dependent upon product type and nuisance species, and at a rate consistent with the label requirements and California Department of Pesticide Regulation licensed Pest Control Adviser (PCA) recommendations. Label requirements for both copper and hydrogen peroxide include limitations on the extent of treatment locations in a lake to no more than one-half of the lake surface area for a single treatment event. All algaecide applications will be made in accordance with the product label.

If elevated toxin concentrations are recorded at several sites in Lake Henshaw, algaecide will be applied at multiple locations within the lake. Algaecide may be applied in alternating strips or rows across Lake Henshaw in order to treat no more than one-half of the lake surface during a single treatment event.

Table 3-2. Type of algaecide to be used in Lake Henshaw.

Algaecide Type	Application method	Primary Degradation Products
Chelated copper	Sprayer, injection boom, granular spreader	None
Copper sulfate	Sprayer or injection boom	None
Sodium carbonate peroxyhydrate (hydrogen peroxide)	Boom injector or spreader	Water, bicarbonate

3.2.4 Algaecide application logistics

The duration (i.e., hours/days) of algaecide application, number of boats necessary for application, and potential space needed for on-site storage prior to Lake Henshaw application are dependent on the specific algaecide used, and the rate at which the algaecide is applied. Off-site storage may also be required. Estimates presented in Table 3-3 assume that algaecides will be used on no more than one-half of the lake surface area per application and will be dispersed to a water column depth of approximately 5 ft. Treatment of 50% of the Lake Henshaw surface area will generally require 1–2 days for copper-based algaecides and 2–4 days for peroxide-based algaecides. The final use rate will be determined based on local conditions at the time of application (e.g., total and/or cyanobacteria cell counts). Lake Henshaw may be closed to recreational use while algaecide application occurs to allow application boats to operate without interference by recreational boating traffic.

Table 3-3. Potential use rates, transportation, storage, and application time for algaecides for short-term HABs control in Lake Henshaw.

Algaecide	Potential Use Rate[1]	Amount Needed for 500-acre Treatment[2]		No. Truck Loads	On-site Storage Required (ft^2) [3]	Estimated Time Required for Application
Copper-based algaecides						
SeClear	1.3 gal/AF	3,250 gal	11.8 totes[4]	0.8	400	1 day
	2.6 gal/AF	6,500 gal	23.6 totes[4]	1.7	800	2 days
Cutrine Plus	0.6 gal/AF	1,500 gal	5.5 totes[4]	0.4	200	< 1 day
	1.2 gal/AF	3,000 gal	10.9 totes[4]	0.8	400	1 day
Hydrogen peroxide-based algaecides						
Phycomycin	32 lbs/AF	80,000 lbs	–	2	800	2 days
	64 lbs/AF	160,000 lbs	–	4	1600	< 4 days
PAK27	32 lbs/AF	80,000 lbs	–	2	800	2 days
	64 lbs/AF	160,000 lbs	–	4	1600	< 4 days

1 Use rates provided by S. Schuler, Eutrophix, pers. Comm., May 2021.
2 Assumes that lake surface area is 1,000 acres such that only 50% of the lake surface area is treated during a single event.
3 Off-site storage may also be required.
4 Totes are 275 gallons each.

Notes:
gal/AF = gallons per acre-foot
lbs/AF = pounds per acre-foot
lbs = pounds
gal = gallon

3.2.5 Transportation and storage of algaecides

The algaecides listed in Table 3-3 require HAZMAT certification for transportation and specific conditions for storage. SeClear must be stored in a locked container or storage area protected from direct sunlight in a dry, cool, well-ventilated area, and is indicated as an environmentally hazardous liquid for transportation. Cutrine Plus must also be stored in a dry, cool, well-ventilated area protected from direct sunlight, but is not listed as a hazardous material for transportation purposes. Phycomycin must be stored in its original, unopened container and in a dry, cool location away from combustible material. PAK27 must be stored in a cool, dry, well-ventilated place away from direct sunlight and in its original container fitted with a safety valve or vent. Both Phycomycin and PAK27 are labeled as Class 5.1 (oxidizing agents) transport hazards.

The number of truckloads of algaecide to be delivered for a treatment event is dependent on the product selected and the application rate (Table 3-3). Liquid totes may be delivered for either copper-based product, while peroxide products are available in a solid, granular form. Both copper-based and peroxide-based algaecides may be stored on-site, dependent on the availability of sufficient areas for local storage that can meet the aforementioned storage conditions and will allow subsequent transportation to the Lake Henshaw boat launch area for application.

3.2.6 Monitoring associated with algaecide application

In addition to monitoring associated with operational algaecide treatment triggers, the Statewide Aquatic Weed Control Permit requires water quality monitoring prior to, during, and following algaecide application, as well as annual reporting. The permit-required monitoring is focused on demonstrating that no chemical residual is associated with algaecide application in waters outside of the application area and that no adverse water quality conditions result from algaecide application. Section 5.5.2 provides additional details related to algaecide application monitoring. Annual reporting is also required for the permit.

3.2.7 Responsibilities to implement short-term algaecide application and monitoring

Licensed applicators can be hired to apply algaecides and undertake monitoring needed to meet permit requirements. However, in order to build internal capacity for understanding algal bloom dynamics and HAB response to treatments, and to minimize the amount of time that unfolds between sampling collection and associated management decisions, District staff will be responsible for routine monitoring during the *operational strategies window*, as well as rapid response monitoring during the *early warning window*, and most monitoring following the *treatment window* (Figure 3-1, see also Section 5.5). Monitoring associated with the Warner Wellfield and Lake Henshaw APAP (Appendix C) will be undertaken by the applicator as part of treatment activities.

Because routine monitoring, and in particular rapid response monitoring, requires relatively quick turn-around for cyanotoxin concentration results to allow for decision making that supports rapid deployment of algaecide treatment, the District may elect to use commercially available rapid cyanotoxin screening tests for routine monitoring of one or more cyanotoxins. Currently, rapid tests are available for microcystin/nodularins and cylindrospermopsin, but not for anatoxin-a (Appendix D). If the District uses rapid screening tests for routine monitoring, then analytical laboratory samples may still be required to confirm the rapid screening test results. This timeline may require expedited processing by the analytical laboratory. Ultimately, rapid decision making is needed to ensure success of algaecide treatment as a short-term mitigation strategy. Additional details are provided in Section 5.5.

4 CONSIDERATION OF POTENTIAL LONG-TERM PREVENTION AND MITIGATION METHODS

4.1 Screening of Potential Long-term Methods

During October and November of 2021, the Project Team[14] participated in a two-part workshop to screen alternatives for preventing or minimizing HABs in Lake Henshaw and Lake Wohlford in the long term, where the latter is defined as occurring in the year 2024 or later. The Project Team evaluated the 18 potential in-lake management methods that were previously evaluated for short-term applicability in Lake Henshaw (Section 3). The screening of long-term alternatives also considered five potential out-of-lake management methods. The in-lake and out-of-lake methods considered at the long-term screening workshop are summarized in Table 4-1 for Lake Henshaw and in Table 4-2 for Lake Wohlford, including relevant existing information compiled over the course of the project.

Based on feedback from the screening workshop for long-term alternatives, the Project Team undertook further evaluation of a subset of selected management methods judged to have the greatest applicability and suitability for long-term water quality improvements in each lake, given the available information. The subset of selected methods is listed below for each lake along with the HABs Management and Mitigation Plan section where additional details are presented.

- Lake Henshaw Selected HABs Prevention Methods (Section 4.2)
  - Out-of-lake: source water nutrient control
  - In-lake: phosphorus inactivation/chemical sediment sealing
  - In-lake: oxygenation via Speece Cone or SDOX
- Lake Henshaw Selected HABs Mitigation Methods (Section 4.3)
  - Out-of-lake: bypass
  - In-lake: algaecide treatment
- Lake Wohlford Selected HABs Prevention Methods (Section 4.4)
  - In-lake: oxygenation via Speece Cone or SDOX
- Lake Wohlford Selected HABs Mitigation Methods (Section 4.5)
  - In-lake: selective withdrawal
  - In-lake: algaecide treatment

[14] The Project Team included staff from the District and Escondido involved in managing Lake Henshaw, the Escondido Canal, and Lake Wohlford, members of the Stillwater Sciences' Team, and Dr. David Caron, as a technical expert representing the Indian Water Authority. The Stillwater Sciences' Team included Stillwater Sciences, Brown and Caldwell, Alex Horne Associates, Robertson-Bryan Inc., University of California at Merced, Marine Biochemists, Water Quality Solutions, and Mr. Bill Taylor.

Table 4-1. Potential HABs prevention and mitigation methods considered for long-term use in Lake Henshaw.

	Method (P/M) [1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Preliminary Cost Estimate	Likely Permitting Timeframe	Recommended for Long Term? (post-2024)
In-lake Methods							
Physical Controls							
1	Dredging (P)	• Removal of nutrients (primarily phosphorus) contained in lake sediments • Reduce nutrient internal loading to reservoir	• On a TN:TP basis, P is more likely to be limiting HABs. On a TIN:OP basis, TIN is more likely to be limiting HABs but both N and P are so plentiful that light becomes limiting (Section 2.2.2.4) • TP content of sediment (n=4) 1,200–4,000 mg/kg dry weight (dw) is high relative to several other productive lakes/ reservoirs in California (600–1,300 mg/kg dw) (Beutel 2021) • Requires extensive permitting • Dredging entire reservoir is impractical and prohibitively expensive • Dredging TP hotspots near fishing dock and dam still expensive and may not be sufficient	• Additional data characterizing spatial distribution of nutrients (primarily TP) in lake sediments (currently n=4) • Onsite or offsite reuse locations	• Dredge area = 120 ac (near dam and fishing dock) Approx. $1M–$2M for onsite reuse; $4M–$11M for offsite reuse [2] • Dredge area = 1,000 ac (whole lake) Approx. $8M–$23M for onsite reuse; $32M–$95M for offsite reuse [2] • Permitting cost $200K–$300K	2–3 years	No
2	Water level fluctuation (P)	• Macrophyte control • Oxidization of littoral sediments to improve redox conditions and reduce nutrient flux out of sediments	• Already occurs seasonally due to water supply • Dry winters may increase HABs but high volumes of groundwater not available to match watershed input during wet winters • Groundwater nutrient levels may be relatively high, such that adding groundwater during dry winters may be counterproductive	• Additional data characterizing HAB intensity and lake storage during dry water years • Current data characterizing nutrients (PO_4^{3-}, TP, NH_4^+, NO_3^-, TN) in pumped groundwater inflows	N/A	N/A	No
3	Mixing and/or destratification (P/M)	• Mix water column via macro bubbles (1-2 mm) or vigorous epilimnetic mixing (VEM) • Large colonial cyanobacteria can be outcompeted by single filament species and/or diatoms	• Bubbler or mixing arrays would need to be very large (i.e., 1,000-2,000 acres) to be effective in shallow water column; smaller arrays would not disrupt HABs in shallow areas or along shorelines such that cyanotoxins would remain problematic • Fe:P ratio in lake sediments is ~30, suggesting that sediments have good PO_4^{3-} binding capacity when oxygenated (Beutel 2021) • Macro-bubbles or VEM would not be likely to oxygenate sediments where TP is elevated (Beutel 2021) and from which seasonal release of PO_4^{3-} and NH_4^+ is apparent (Section 2.2.2.3) • Mobley-style diffusers may snag recreational fishing anchors	• Confirmation of which cyanobacteria species in the lake are the dominant cyanotoxin producers (i.e., large colonial *Dolichospermum sp.* that would be disrupted by mixing, or single filament *Planktothrix sp.* that would not be disrupted by mixing [see also Section 2.4.3.1]) • Water column thermal and chemical stratification patterns to confirm to what degree seasonal stratification is occurring	$2M-$4M	N/A	No
4	Macrophyte harvesting (P)	• Nutrient removal via plant harvest	• Macrophytes not a concern in this lake, cyanotoxins are likely caused by phytoplankton rather than epiphytic algae growing on macrophyte surfaces	N/A	N/A	N/A	No
5	Wetland filters (fringe) (P)	Algae, nutrient removal via nutrient uptake and transformations in constructed wetlands	• Insufficient area downstream of dam (~ 4 ac) to treat suspended solids in lake release water at 5–45 cfs; possible washout if spillway activated • Shoreline options are possible, particularly along southern shoreline where terraced and shallow (1.25 ft deep) wetlands 35–140 acres could filter HABs/remove suspended solids [3] • Would require mobile pump in lake to effectively capture algal scums and transport for treatment in shoreline wetlands • High evapotranspiration loss anticipated in shoreline wetlands	N/A	$3M–$4.5M design and implementation; $140K–$200K annual operation and maintenance (O&M); $900K–$1.2M in ET losses [3]	2-3 years	No

	Method (P/M) [1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Preliminary Cost Estimate	Likely Permitting Timeframe	Recommended for Long Term? (post-2024)
6	Algae harvesting, separation, skimming (P, M)	Physical removal of algae and nutrients	• Algae screening in lake and screening and/or dissolved air flotation (DAF) at reservoir outlet • Existing in-lake technology limited to harvest at water surface • Screening alone does not address dissolved cyanotoxins • At reservoir outlet, screening and/or DAF could be combined with chemical (e.g., hydrogen peroxide, ozone) treatment to inactivate cyanotoxins but would require NPDES permit • Scale issues for facilities and biomass re-use/disposal for both in-lake and reservoir outlet	• Biomass re-use/disposal options need further investigation at scale	$2M–$10M capital costs and permitting; $500K–$1K per AF to operate for 6 months/year for 10 years	2-3 years	No
7	Selective withdrawal (M)	Control which parcels of water are released from reservoir to minimize downstream nutrient and/or toxin transport	• Reservoir is shallow, outlet already minimizes toxin export by drawing from bottom waters	• Water column thermal and chemical stratification patterns to confirm to what degree seasonal stratification is occurring	N/A	1-2 years	No
8	Dilution/ flushing/ bypass (P/M)	• Dilution/flushing would decrease residence time of water in lake to prevent or limit HABs by creating conditions unsuitable for cyanobacteria • Bypass would route cyanotoxin-free water around the lake to meet entitlements and other water supply needs	• High volume/low nutrient flow not available for dilution or flushing (groundwater nutrient levels may be relatively high, such that adding groundwater during dry winters may be counterproductive) • Bypass would reduce evaporative losses of pumped groundwater which could offset costs of construction and operation	• Dilution/flushing – recent data on nutrient inputs from local creeks and pumped groundwater from the Warner Ranch Wellfield • Bypass – response of the lake to substantially less to no input of groundwater each year	N/A for dilution/flushing; Initial estimate $10M–$15M for bypass; revised estimate in Section 4.3.1	3-4 years for bypass	No for dilution/ flushing; Yes for bypass
9	Sediment sealing/ capping (fabrics) (P)	Reduce phosphorus release from lake sediment	• Off-gassing of bottom sediments could float fabric off of the lake bottom • TP content of sediment (n=4) 1,200–4,000 mg/kg dry weight (dw) is high relative to several other productive lakes/ reservoirs in California (600–1,300 mg/kg dw) (Beutel 2021) • Capping entire reservoir is impractical and prohibitively expensive • Capping TP hotspots near fishing dock and dam still expensive and may not be sufficient	• Whether phosphorus availability is likely to limit HABs • Additional data characterizing spatial distribution of nutrients (primarily TP) in lake sediments (currently n=4)	>$5M	2–3 years	No
10	Ultrasonic waves (P)	Render cyanobacteria unable to control their position in the water column	• Ultrasonic save buoy arrays would need to be very large (e.g., n=30 across 1,000 to 2,000 acres)	• Whether ultrasonic waves are sufficient as a single strategy or needs to be combined with other mitigation strategies	N/A	N/A	N/A

	Method (P/M) [1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Preliminary Cost Estimate	Likely Permitting Timeframe	Recommended for Long Term? (post-2024)
Chemical Controls							
11	Algaecides (M)	• Effective in destroying HAB-causing algae and potentially also toxins, if used properly	• Can be immediately effective, with effects typically lasting 2–3 weeks, follow up applications may be necessary • Low capital investment, product cost varies by method	• Routine monitoring and rapid response monitoring approaches needed to provide early warning and trigger algaecide treatment before HAB is out of control (see also Section 3.2) • Spatial distribution and cell density of HAB needed to determine algaecide treatment locations and appropriate dosing	\$75K–\$400K/year for algaecide treatment (see also Section 3.2)	District obtained application permit for Warner Ranch Wellfield and Lake Henshaw in June 2021	Yes
12	Oxygenation via Speece Cone or SDOX/aeration using ultrafine or nanobubbles (P)	• Speece Cone or SDOX increases water column *and* sediment dissolved oxygen (DO) without destratification • Aeration/oxygenation using ultrafine bubbles (1-100 um diameter) or nanobubbles (< 200 nanometers [nm] diameter) increase water column DO without destratification • Nanobubbles are neutrally buoyant and stable in water column • Ozone (O_3) input to bubblers can deactivate cyanotoxins	• Speece Cone or SDOX most efficient way to oxygenate water column and sediments, and at least partially eliminate internal loading of N, P, Mn, Fe, Hg, S • Fe:P ratio in lake sediments is ~30, suggesting that sediments have good PO_4^{3-} binding capacity when oxygenated (Beutel 2021) • Apparent seasonal release of PO_4^{3-} and NH_4^+ from bottom sediments (Section 2.2.2.3) and measured sediment nutrient release rates from chamber study (Beutel 2021): – NH_4^+ = 150-264 mg-N/m2·d normalized to 20 oC (4–5x higher than temperature-normalized fluxes from several hypereutrophic lakes in California; – PO_4^{3-} = 90-120 mg-P/m2·d (4–20x higher than fluxes from other reported hypereutrophic lakes; – NH_4^+ and PO_4^{3-} release from sediments highest in anoxic (DO=0 mg/L) and hypoxic (DO< 2 mg/L) conditions and decreased with increasing water column DO • Placement of cone or aerators/bubblers in lake near dam in deepest portion of reservoir or on western shore, may not be able to oxygenate entire lake water column and/or sediments such that HABs could persist in shallow areas or along shorelines • Nanobubblers and use of O3 in ultrafine or nanobubblers are experimental; these approaches have not been tested at the scale of Lake Henshaw	• Data characterizing spatial extent of blooms in lake • Pilot study for Speece Cone and/or SDOX to determine extent of treatment influence in broad, shallow lake • Whether alum dosing in combination with oxygenation via Speece Cone is necessary	Speece Cone or SDOX oxygenation: \$2M–\$7M design and implementation; \$100K–\$200K annual O&M[5]	6 months to 2 years	Oxygenation via Speece Cone or SDOX—Maybe; Aeration using nanobubbles—No
13	Shading/dyes (P)	Effective by shading algae and preventing or reducing growth	• Not feasible in large water bodies	N/A	N/A	N/A	No
14	Phosphorus inactivation/ chemical sediment sealing (P)	Particles (e.g., alum, Phoslock™) bind with phosphorus, algae, and/or detritus and are then removed from water column; settled particles form barrier on sediment surface to reduce phosphorus flux out of sediments	• Can be immediately effective but may require several applications • External loading of PO_4^{3-} from pumped groundwater may be high enough that in-lake sediment sealing alone is not sufficient • Apparent seasonal release of PO_4^{3-} from bottom sediments (Section 2.2.2.3) and measured sediment phosphorus release rates from chamber study (Beutel 2021): – PO_4^{3-} = 90-120 mg-P/m^2·d (4–20x higher than fluxes from other reported hypereutrophic lakes; – PO_4^{3-} release from sediments highest in anoxic (DO=0 mg/L) and hypoxic (DO< 2 mg/L) conditions and decreased with increasing water column DO • Alum use requires NPDES Individual Permit • Phoslock use requires NPDES General Permit No. CAG999003 for lanthanum-modified clays	• While Fe:P ratio in lake sediments is ~30, suggesting that sediments have good PO4 binding capacity when oxygenated (Beutel 2021), the fraction of TP in sediments that is mobile (i.e., adsorbed to Fe, Al, and labile organics) versus refractory (i.e., adsorbed to minerals such as apatite and refractory organics) needs to be quantified to inform application rates	• Alum heavy dose (Al:P molar = 20; Al = 13 mg/L) \$160K–\$4M [6] • Phoslock heavy dose (2,000 lbs/ac) \$180K–\$4.5M [6] • Moderate 'Floc & Lock' Dose (Al:P molar = 10; Al = 6 mg/L; Phoslock 1,000 lbs/ac) \$175K-\$4.4M [6]	0.5–1 year	Yes—in combination with Out-of-lake Method No. 1 (see below)

	Method (P/M) [1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Preliminary Cost Estimate	Likely Permitting Timeframe	Recommended for Long Term? (post-2024)
Biological Controls							
15	Pathogens/ diseases of algae (P)	Reduce abundance of HAB-causing algae	• Experimental; not proven in large lakes/reservoirs	N/A	N/A	N/A	No
16	Grazers (on algae or macrophytes) (P)	Consumption of macrophytes or HAB-causing algae	• Enhanced grazing will occur if oxygenation and/or biomanipulation are used • Not a standalone method as a long-term method	N/A	N/A	N/A	No
17	Nutrient harvesting from fish/weeds (P)	Removal of nutrients from system via removal of organisms	• Experimental • Not recommended as single method; minimal removal of nutrients annually	N/A	N/A	N/A	No
18	Biomanipulation (P)	• Alteration of food web and trophic linkages to reduce abundance of HAB-causing algae	• Requires detailed understanding of aquatic food web, success may rely on abundance of leafy pondweeds, which is unlikely in Lake Henshaw	Detailed understanding of aquatic food web	N/A	N/A	No
Out-of-lake Methods							
1	Source water nutrient controls (P)	Reduce external nutrient loading	• Consider continuous chemical dosing on wellfield outflow, which may add 0.3 to 70 kg PO_4-P/month to the lake (Section 2.1.3.3)	• Recent data characterizing nutrient concentrations and flows from local creeks and pumped groundwater from the Warner Ranch Wellfield	\$50K–\$350K per year chemical costs; \$5K–\$25K application equipment	0.5–1.5 year	Yes
2	Stream rerouting (P)	Eliminate external nutrient loading by rerouting contributing stream(s) away from nutrient sources and/or around the receiving water	• Flows from local tributaries are seasonally ephemeral • Since local streams are the primary natural water source for the lake, it is not practical to bypass those flows	• Recent data characterizing nutrient concentrations and flows from local creeks and pumped groundwater from the Warner Ranch Wellfield	N/A	N/A	No
3	Erosion control BMPs (P)	Reduce external nutrient loading from manure and eroded sediments mobilized in stormwater runoff	• Grazing livestock are currently excluded from the lakeshore and most tributaries, which minimizes direct input of nutrients • In arid local climate, nitrogen from manure is rapidly taken up by vegetation and microbial community in soils, and is thus not available for transport to the lake in runoff	N/A	N/A	N/A	No additional measures needed—maintain current livestock exclusion fencing
4	Riparian filters (tributary streams) (P)	Reduce external nutrient loading by uptake and microbial cycling in riparian vegetation	• Tributary streams are largely seasonally ephemeral and would not support additional riparian extent	N/A	N/A	N/A	No
5	Treatment wetlands (tributary streams) (P)	Algae, nutrient removal via nutrient uptake and transformations in constructed wetlands	• Tributary streams are largely seasonally ephemeral and would not support adjacent treatment wetlands	N/A	N/A	N/A	No

[1] "P" denotes a method designed to prevent HABs formation, "M" denotes a method designed to mitigate the effects of HABs formation.
[2] Dredging cost estimates vary based on the acreage to be dredged, and whether sediment is to be reused on- or offsite. Preliminary cost estimates assume an average TP content in mg/kg dw based on Beutel (2021), dredge depth of 10–30 cm, dredged sediment density of 1.2 g/cm^3, and dredging unit cost of \$15-60/yd^3.
[3] Terraced treatment wetlands would be subject to lake inundation depending on elevation. On the southern shoreline, 35 ac of treatment wetland at 2,668–2,672 ft elevation would be inundated every 2 years; 142 acres of treatment wetland at > 2,672 ft elevation would be inundated every 3 years. Treatment wetland preliminary cost estimates assume 5 ft/yr evapotranspiration (ET) water loss.
[4] Preliminary algaecide application costs vary based on application frequency and the type and amount of chemical applied (see also Section 3.2).
[5] Preliminary oxygenation cost estimates assume 1.6 ton O_2/day (Section 4.2.3).
[6] Preliminary sediment sealing cost estimates assume treatment areas of ~ 30 ac (2,645 ft water surface elevation [WSE]), 120 ac (2,650 ft WSE), and 750 ac (2,660 ft WSE).

Table 4-2. Potential HABs prevention and mitigation methods considered for long-term use in Lake Wohlford.

	Method (P/M)[1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Preliminary Cost Estimate	Likely Permitting Timeframe	Recommended for Long Term? (post-2024)
In-lake Methods							
Physical Controls							
1	Dredging (P)	• Removal of nutrients (primarily phosphorus) contained in lake sediments • Reduce nutrient internal loading to reservoir	• Available data indicate 25–30 ft of sediment deposition near the dam. Deposition may have occurred in the 1950's or 1960's when alum was added to the lake or it may be due to shoreline erosion and creek bottom erosion after heavy rains. • TP content of sediment (n=4) 980–2,000 mg/kg dry weight (dw) is high relative to several other productive lakes/ reservoirs in California (600–1,300 mg/kg dw) (Beutel 2021). • Requires extensive permitting • Cost estimates consider the only top portion of the sediment for dredging, but almost all sediments would need to be removed to eliminate PO_4^{3-} release • Costs would be prohibitively expensive	• Whether phosphorus availability is likely to limit HABs • Spatial distribution of nutrients (primarily phosphorus) in lake sediments (currently n=3)	• Dredge area = 110 ac (near dam and fishing dock) Approx. \$800K–\$2.6M for onsite reuse; \$3.5M–\$10.5M for offsite reuse [2]	2–3 years	No
2	Water level fluctuation (P)	• Macrophyte control • Oxidization of littoral sediments to improve redox conditions and reduce nutrient flux out of sediments	• Already occurs seasonally due to water supply • Dry winters may increase HABs but there are data gaps • No apparent existing pattern between fluctuations and HAB blooms	• Additional data characterizing HAB intensity and lake storage during dry water years	N/A	N/A	No
3	Mixing and/or destratification/aeration (P/M)	• Mix water column via macro bubbles (1–2 mm) or vigorous epilimnetic mixing (VEM) • Large colonial cyanobacteria can be outcompeted by single filament species and/or diatoms	• Existing aeration-mixing system has a single end-of-pipe outlet in deeper waters near the dam. The existing thermal and DO profiles (Figure 2-46a,b, Figure 2-47a,b, Figure 4-12) indicate that the current aeration system is not able to prevent anoxic conditions in Lake Wohlford throughout the year. • Higher capacity bubbler or mixing arrays would need to be large (i.e., 80–160 acres) to be effective; the smaller system does not disrupt HABs in shallow areas or along shorelines due to continuing internal loading of nutrients. • Fe:P ratio in lake sediments is ~30–45, suggesting that sediments have good PO_4^{3-} binding capacity when oxygenated (Beutel 2021). • Escondido intends to continue operation of the current system until the new dam is in place, at which point an expansion of the current system or replacement of the current system with a more efficient oxygenation system would be considered.	• Confirmation of which cyanobacteria species in the lake are the dominant cyanotoxin producers (i.e., large colonial *Dolichospermum sp.* that would be disrupted by mixing, or single filament *Planktothrix sp.* that would not be disrupted by mixing [see also Section 2.4.3.1])	N/A	N/A	Continue existing system operation until new dam is in place
4	Macrophyte harvesting (P)	• Nutrient removal via plant harvest	• Macrophytes not a concern in this lake, cyanotoxins are likely caused by phytoplankton rather than epiphytic algae growing on macrophyte surfaces	N/A	N/A	N/A	No
5	Wetland filters (fringe) (P)	• Algae, nutrient removal via nutrient uptake and transformations in constructed wetlands	• Available area upstream of reservoir will be flooded with new dam; potential for acreage further upstream of larger reservoir footprint?	N/A	N/A	2–3 years	No
6	Algae harvesting, separation, skimming (P, M)	• Physical removal of algae and nutrients	• Variable costs and methodologies; each may require various permits • Existing in-lake technology limited to harvest at water surface • Screening alone does not address dissolved cyanotoxins • Scale issues for facilities and biomass re-use/disposal for in-lake locations	• Biomass re-use/disposal options need further investigation at scale	N/A	2–3 years	No

	Method (P/M)[1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Preliminary Cost Estimate	Likely Permitting Timeframe	Recommended for Long Term? (post-2024)
7	Selective withdrawal (M)	• Control which parcels of water are released from reservoir to minimize downstream nutrient and/or toxin transport	• Current outlet tower only has one usable gate; planning for the outlet tower on the new dam includes four gates	• Water column thermal and chemical stratification patterns in deeper, larger reservoir to confirm to what degree seasonal stratification is occurring	• $1.8M–$2.6M for capital costs	1–2 years	Yes, for new dam
8	Dilution/flushing (P/M)	• Dilution/flushing would decrease residence time of water in lake to prevent or limit HABs by creating conditions unsuitable for cyanobacteria	• High volume/low nutrient flow not available for dilution	N/A	N/A	N/A	No
9	Sediment sealing/ capping (fabrics) (P)	• Reduce phosphorus release from lake sediment	• Off-gassing of bottom sediments could float fabric off of the lake bottom • Available data indicate 25–30 ft of sediment deposition near the dam. Deposition may have occurred in the 1950's or 1960's when alum was added to the lake or it may be due to shoreline erosion and creek bottom erosion after heavy rains. • TP content of sediment (n=4) 980–2,000 mg/kg dry weight (dw) is high relative to several other productive lakes/ reservoirs in California (600–1,300 mg/kg dw) (Beutel 2021) • Capping entire reservoir is impractical and prohibitively expensive and P hotspots are unlikely	• Whether phosphorus availability is likely to limit HABs • Additional data characterizing spatial distribution of nutrients (primarily TP) in lake sediments (currently n=3)	>$5M	2–3 years	No
10	Ultrasonic waves (P)	• Render cyanobacteria unable to control their position in the water column	• Ultrasonic save buoy array would need to include a minimum of 6 units given an effect radius of 250 m per unit	• Whether ultrasonic waves are sufficient as a single strategy or needs to be combined with other mitigation strategies	N/A	N/A	No
Chemical Control							
11	Algaecides (M)	• Effective in destroying HAB-causing algae and potentially also toxins, if used properly	• Can be immediately effective, with effects typically lasting 2-3 weeks, follow up applications may be necessary • Low capital investment, product cost varies by method	• Routine monitoring and rapid response monitoring approaches needed to provide early warning and trigger algaecide treatment before HAB is out of control (see also Section) • Once new dam is in place, whether larger reservoir would require more frequent or more extensive treatment	• Algaecides: up to $45K (per year) for algaecide treatment[3]	• Escondido obtained permit for Lake Wohlford in 2013	Algaecides – Yes

	Method (P/M)[1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Preliminary Cost Estimate	Likely Permitting Timeframe	Recommended for Long Term? (post-2024)
12	Oxygenation via Speece Cone or SDOX/ ultrafine or nanobubbles (P)	• Speece Cone increases water column and sediment dissolved oxygen (DO) without destratification • Aeration/oxygenation using ultrafine bubbles (1–100 um diameter) or nanobubbles (< 200 nanometers [nm] diameter) increase water column DO without destratification • Nanobubbles are neutrally buoyant and stable in water column • Ozone (O_3) input to bubblers can deactivate cyanotoxins	• Speece Cone or SDOX most efficient way to oxygenate water column and sediments, and at least partially eliminate internal loading of N, P, Mn, Fe, Hg, S • Fe:P ratio in lake sediments is ~30-45, suggesting that sediments have good PO_4^{3-} binding capacity when oxygenated (Beutel 2021). • Apparent seasonal release of PO_4^{3-} and NH_4^+ from bottom sediments (Section 2.2.2.3) and measured sediment nutrient release rates from chamber study (Beutel 2021): – NH_4^+ = 30-60 mg-N/m2·d normalized to 20 oC (similar to temperature-normalized fluxes from several hypereutrophic lakes in California); – PO_4^{3-} = 50-70 mg-P/m2·d (5-10x higher than fluxes from other reported hypereutrophic lakes); – NH_4^+ and PO_4^{3-} release from sediments highest in anoxic (DO=0 mg/L) and hypoxic (DO< 2 mg/L) conditions and decreased with increasing water column DO • Nanobubblers and use of O_3 in ultrafine or nanobubblers are experimental; these approaches have not been tested at the scale of Lake Wohlford	• Pilot study for Speece Cone and/or SDOX to determine extent of treatment influence in deeper lake once dam is replaced • Whether alum dosing in combination with oxygenation via Speece Cone is necessary	• Speece Cone or SDOX oxygenation: \$6.3M–\$11M design and implementation; \$185K–\$360K annual O&M [4]	N/A	Oxygenation via Speece Cone or SDOX – Yes; Aeration using nanobubbles – No
13	Shading/dyes (P)	• Effective by shading algae and preventing or reducing growth	• Not feasible in a drinking water reservoir	N/A	N/A	N/A	No
14	Chemical sediment sealing (P)	• Particles (e.g., alum, Phoslock™) bind with phosphorus, algae, and/or detritus and are then removed from water column; settled particles form barrier on sediment surface to reduce phosphorus flux out of sediments	• Can be immediately effective but may require several applications • If external loading of PO_4^{3-} from Lake Henshaw discharges continues then in-lake sediment sealing alone would not be sufficient • Apparent seasonal release of PO_4^{3-} from bottom sediments (Section 2.2.2.3) and measured sediment phosphorus release rates from chamber study (Beutel 2021): – PO_4^{3-} = 50-70 mg-P/m2·d (5-10x higher than fluxes from other reported hypereutrophic lakes); – NH_4^+ and PO_4^{3-} release from sediments highest in anoxic (DO=0 mg/L) and hypoxic (DO< 2 mg/L) conditions and decreased with increasing water column DO • Alum use requires NPDES Individual Permit • Phoslock use requires NPDES General Permit No. CAG999003 for lanthanum-modified clays	• While Fe:P ratio in lake sediments is ~30-45, suggesting that sediments have good PO_4^{3-} binding capacity when oxygenated (Beutel 2021), the fraction of TP in sediments that is mobile (i.e., adsorbed to Fe, Al, and labile organics) versus refractory (i.e., adsorbed to minerals such as apatite and refractory organics) needs to be quantified to inform application rates	• Alum heavy dose (Al:P molar = 20; Al = 13 mg/L) \$120K-\$930K[5] • Phoslock heavy dose (2,000 lbs/ac) \$180K–\$1.4M[5] • Moderate 'Floc & Lock' Dose (Al:P molar = 10; Al = 6 mg/L; Phoslock 1,000 lbs/ac) \$10K–\$500K[5]	0.5–1 year	No since this method would only be used as part of an oxygenation system, not as a stand-alone method
Biological Control							
15	Pathogens/ diseases of algae (P)	• Reduce abundance of HAB-causing algae	• Experimental; not proven in large lakes/reservoirs	N/A	N/A	N/A	No
16	Grazers (on algae or macrophytes) (P)	• Consumption of macrophytes or HAB-causing algae	• Enhanced grazing will occur if oxygenation and/or biomanipulation are used • Not a standalone method as a long-term method	N/A	N/A	N/A	No
17	Nutrient harvesting from fish/weeds (P)	• Removal of nutrients from system via removal of organisms	• Experimental • Not recommended as single method; minimal removal of nutrients annually	N/A	N/A	N/A	No
18	Biomanipulation (P)	• Alteration of food web and trophic linkages to reduce abundance of HAB-causing algae	• Requires detailed understanding of aquatic food web, success may rely on abundance of leafy pondweeds, which is unlikely in Lake Wohlford	• Detailed understanding of aquatic food web	N/A	N/A	No

	Method (P/M)[1]	Goals and Capabilities	Relevant Existing Information	Additional Information Needs	Preliminary Cost Estimate	Likely Permitting Timeframe	Recommended for Long Term? (post-2024)
Out-of-lake Methods							
1	Source water nutrient controls (P)	• Reduce external nutrient loading	• Lake Henshaw is the primary source water to Lake Wohlford	N/A	N/A	N/A	Yes but only as it applies to Lake Henshaw as the primary source water
2	Stream rerouting (P)	• Eliminate external nutrient loading by rerouting contributing stream(s) away from nutrient sources and/or around the receiving water	• No major stream input, no data on seasonal flows or nutrient inputs for minor tributaries, not practical	• Information on seasonal flows or nutrient inputs from tributaries	N/A	N/A	No
3	Erosion control BMPs (P)	• Reduce external nutrient loading from sediments and nutrients mobilized in stormwater runoff from upstream agricultural and residential areas	• Sediment controls in agricultural areas upstream of reservoir may already be in place • Sediment and nutrient controls in limited residential areas adjacent to reservoir have already been addressed	• Information on seasonal flows, sediment controls, and nutrient inputs from upstream agricultural areas	N/A	N/A	No because already complete
4	Riparian filters (tributary streams) (P)	• Reduce external nutrient loading by uptake and microbial cycling in riparian vegetation	• Tributary streams: small catchment, riparian areas are limited but already exist	N/A	N/A	N/A	No
5	Treatment wetlands (tributary streams) (P)	• Algae, nutrient removal via nutrient uptake and transformations in constructed wetlands	• Once the new dam is in place, the upstream end of the reservoir will be inundated and unavailable as a potential treatment wetland location • The upstream agricultural areas are not currently available as a location for a treatment wetland	• Whether upstream agricultural lands would become available as a feasible location for an offline treatment wetland	N/A	N/A	No

1 "P" denotes a method designed to prevent HABs formation, "M" denotes a method designed to mitigate the effects of HABs formation.
2 Dredging cost estimates vary based on the acreage to be dredged, and whether sediment is to be reused on- or offsite. Preliminary cost estimates assume an average TP content in mg/kg dw based on Beutel (2021), dredge depth of 10–30 cm, dredged sediment density of 1.2 g/cm^3, and dredging unit cost of $15-60/yd^3.
3 Preliminary algaecide application costs vary based on application frequency and the type and amount of chemical applied. This estimate is based on the current size of Lake Wohlford and current operations.
4 Preliminary oxygenation cost estimates assume 1.5–3.5 ton O_2/day (Section 4.4.1).
5 Preliminary sediment sealing cost estimates assume treatment areas of ~ 30 ac (2,645-ft water surface elevation [WSE]), 120 ac (2,650-ft WSE), and 750 ac (2,660-ft WSE).

4.2 Lake Henshaw Selected Prevention Methods

4.2.1 Out-of-lake method 1 – Source Water Nutrient Control – Phosphorus Inactivation of Warner Ranch Wellfield Outflow

4.2.1.1 Goals and capabilities

The goal of source water nutrient control is to inactivate bioavailable phosphorus (i.e., orthophosphate [PO_4^{3-}]) in Warner Ranch Wellfield inflows to Lake Henshaw (Table 2-1 and Section 2.1.3.3) to reduce external loading of this nutrient to the lake. Currently, pumped groundwater transfers from the Warner Ranch Wellfield contribute orthophosphate to Lake Henshaw in concentrations sufficient to stimulate cyanobacterial blooms (Figure 2-5 and Section 2.1.3).

4.2.1.2 Implementation considerations

Source water nutrient control would occur through application of a chemical (e.g., alum, EutroSORB™) that binds rapidly and permanently with orthophosphate in flowing water coming from the wellfield. Application equipment would be installed at or near the terminus of the wellfield and at an outflow point associated with the 70's Wells to treat pumped groundwater as it leaves the ditch system and moves towards the lake (Figure 4-1, see also Figure 2-2). Application equipment location would need to be vehicle-accessible to allow regular chemical delivery and include a level site for installation of one or more storage totes. Chemical choice should include consideration of the chemical and the degree of floc that may be formed through phosphorus binding.

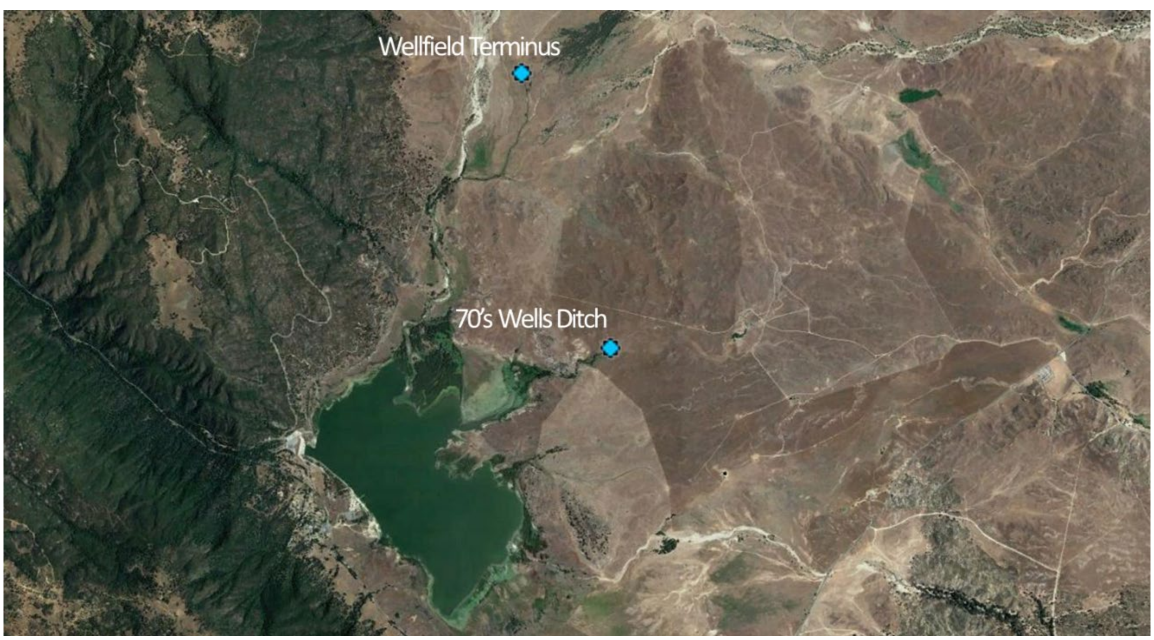


Figure 4-1. Preliminary location of source water nutrient control locations for phosphorus inactivation of Warner Ranch Wellfield groundwater inflows to Lake Henshaw.

The type of application equipment would depend upon the level of automation required. Options include a simple steady-state rate pump that is activated when the groundwater wells are pumping or a remote-controlled pump system using telemetry monitoring and rate integration based upon flow and phosphorus concentration. The lifespan of a steady-state system is 20+ years, whereas a more sophisticated automated system with rate integration would be approximately 10 years. Both systems would require minimal maintenance.

Mean monthly flows produced by the Warner Ranch Wellfield for the period of record indicate that peak flows rarely exceed 30 cfs and typically range 10–15 cfs when the wellfield is pumping (1953–present; Figure 2-3). Analysis of the most recent two decades indicates that mean monthly production is fairly steady at 12–14 cfs when the wellfield is pumping (Table 4-3). Since most of the wellfield production moves past the wellfield terminus, preliminary sizing for a chemical application system at this location is expected to be within this range.

Table 4-3. Warner Ranch Wellfield production (cfs) water year (WY) 2000–2020.

Month	Mean Flow (cfs)	Standard Deviation (cfs)
Jan	12.9	4.5
Feb	11.5	4.3
Mar	13.5	4.1
Apr	13.3	3.7
May	13.0	4.9
Jun	13.0	3.9
Jul	13.7	4.5
Aug	12.9	4.0
Sep	11.8	4.5
Oct	12.8	5.0
Nov	13.0	4.0
Dec	13.4	4.5

Dates when the wellfield is not pumping are not included in the summary statistics.

Historical orthophosphate concentrations for pumped groundwater from the Warner Ranch Wellfield average 0.05 mg/L, although the sample size is small (n=9; Table 2-1). More recent data collected by the District during winter 2022 indicate that total phosphorus in Warner Ranch Wellfield outflow was predominantly bioavailable orthophosphate, concentrations were fairly steady over several weeks at both locations, and average concentrations at the wellfield terminus were roughly twice those at the 70's Wells and roughly 40% higher than the historical average.

Table 4-4. Phosphorus concentrations in Warner Ranch Wellfield pumped groundwater during winter 2022.

Sample Date	Wellfield Terminus		70s' Wells	
	Orthophosphate (PO_4^{3-}) [1] (mg/L)	Total phosphorus (TP) [1] (mg/L)	Orthophosphate (PO_4^{3-}) [1] (mg/L)	Total phosphorus (TP) [1] (mg/L)
1/3/2022	0.08	0.08	0.03	0.03
1/18/2022	0.08	0.08	0.03	0.03
1/24/2022	0.03	0.04	0.08	0.09
1/31/2022	0.07	0.07	0.04	0.04
2/7/2022	0.07	0.07	0.04	0.04
2/14/2022	0.07	0.09	0.04	0.04
2/28/2022	0.08	0.08	0.03	0.03
3/7/2022	0.07	0.08	0.03	0.03
3/14/2022	0.06	0.06	0.03	0.02
4/4/2022	0.06	0.06	0.03	0.02
Mean	0.07	0.07	0.04	0.04
Standard deviation	0.01	0.01	0.02	0.02

[1] Method reporting limit (MRL) = 0.05 mg/L.

4.2.1.3 Anticipated implementation schedule

The design, permitting, and construction of a chemical dosing system for phosphorus inactivation of Warner Ranch Wellfield pumped groundwater is anticipated to require approximately one to two years, with the construction phase lasting 2–3 months depending on the level of automation required. The implementation schedule would need to be refined in a detailed design phase, with special consideration given to any permitting requirements and associated timeframes. The latter is currently estimated at 1−1.5 years to acquire an NPDES Individual Permit.

4.2.1.4 Compatibility

Source water nutrient control via phosphorus inactivation of Warner Ranch Wellfield outflow using chemical application is compatible with the District's water supply objectives, as it would reduce the external loading of bioavailable phosphorus that supports HABs in Lake Henshaw (Figure 2-5 and Section 2.1.3) and would thus improve the quality of water diverted to the District, Escondido, the Rincon Band, and the La Jolla Band. Bound phosphorus entering Lake Henshaw would be permanently buried in lake sediments. Use of an aquatic-approved chemical for phosphorus inactivation would not result in adverse impacts to lake biota, including warm water fish species that currently reside in Lake Henshaw.

To have maximum positive effect on Lake Henshaw water quality, source water nutrient control must be combined with one or more in-lake prevention methods (e.g., sediment sealing [Section 4.2.2], oxygenation [Section 4.2.3]) that controls release of orthophosphate, along with other redox-sensitive compounds, from the reservoir sediments during anoxic (low oxygen) conditions. Internal nutrient loading, particularly in deeper areas of the lake, is likely to contribute a much larger proportion of bioavailable nutrients to the lake than does external loading (i.e., groundwater transfers or runoff) under existing conditions (Figure 2-5 and Section 2.1.3), thus source water nutrient control should be coupled with one or more methods that addresses internal loading. Combined, source water nutrient control via phosphorus inactivation plus an in-lake prevention measure would improve visual and aesthetic characteristics in Lake Henshaw and in release waters, since reduced nutrient cycling would control cyanobacteria blooms and subsequently the prevalence of algal scum accumulation.

4.2.1.5 Estimated costs

While historical summary data suggest a range of phosphorous loading rates to Lake Henshaw at 0.3–70 kg PO_4-P/month, with a period average of 32 kg PO_4-P/month (Section 2.1.3.3), more recently collected data in winter 2022 suggest that loading is closer to 70 kg PO_4-P/month. Additional seasonal data are needed to determine whether this higher end of the historical range is consistent or whether there is meaningful seasonal variability in the loading amount. To remove 98–99% of 70 kg PO_4-P/month, chemical costs would range approximately $4K to $21K per month (based on average monthly wellfield production WY 2000–2020), or approximately $50K to $250K annually, depending on volume discounts and the chemical used. Targeting a 50% reduction in orthophosphate is expected to reduce anticipated chemical costs roughly proportionally to approximately $3K to $10K per month, or approximately $35K–$125K annually. The cost of application equipment would range from $3K–$4K for a simple steady-state rate pumping system to $25K for a remote-controlled pump system using telemetry monitoring and rate integration, with potential for solar power.

4.2.1.6 Permit requirements

Source water nutrient control via phosphorus inactivation of Warner Ranch Wellfield outflow using chemical application likely would require approval from the Regional Board through the NPDES permit process.

4.2.1.7 Additional information needs

Additional seasonal data characterizing phosphorus speciation (i.e., orthophosphate, total phosphorus) in Warner Ranch Wellfield outflows are needed to determine whether the higher end of the historical range of phosphorus loading is consistent over time or whether there is meaningful seasonal variability in the loading amount. The cost of chemical dosing is sensitive to the amount and form of phosphorus in the source water, so optimizing this information is important. Additional characterization of the source water would be needed to refine the chemical dosing using analysis of a relatively small number of additional grab samples (e.g., less than 10 gallons).

4.2.2 In-lake method 13 – Phosphorus Inactivation/Chemical Sediment Sealing

4.2.2.1 Goals and capabilities

The goal of phosphorus inactivation/chemical sediment sealing is to remove bioavailable phosphorus (i.e., orthophosphate [PO_4^{3-}]) from the water column and minimize or eliminate orthophosphate release from Lake Henshaw sediments during hypoxic (DO < 2 mg/L) or anoxic (DO = 0 mg/L) conditions. The magnitude of orthophosphate release measured in Lake Henshaw sediment chambers (approximately 90 to 120 mg PO_4-P/m^2/d) was the largest ever measured in similar studies of California reservoirs (Section 2.2.2.3). Minimizing or eliminating such a high flux would reduce internal loading of phosphorus and the potential for this nutrient to support

HABs (Figure 2-55). A fundamental assumption of phosphorus inactivation as a lake management technique is that phosphorus is the limiting nutrient and its control will, in turn, control nuisance algae growth. Existing information characterizing Lake Henshaw indicates the lake has shifted from a nitrogen-limited system to one that is phosphorus-limited on a total nutrient basis and on a bioavailable nutrient basis (i.e., orthophosphate, ammonia, nitrate) both nitrogen and phosphorus are present at high levels throughout the year (Section 2.2.2.4).

Phosphorus inactivation/chemical sediment sealing would occur through application of a chemical (e.g., alum, lanthanum) to the lake. Alum refers to an aluminum sulfate ($Al_2(SO_4)_3$) coagulant that is added to water to form a colloidal precipitate; other inorganic and organic coagulant alternatives are also available. Flocculation of colloidal precipitates results in particle settling, which physically removes particulate phosphorus associated with suspended sediments and algae, and chemically removes dissolved phosphorus that binds with hydroxide surfaces on floc particles. The latter settle into existing surficial sediments where the phosphorus remains bound over time. This process does not form a sediment cap and is not a biological barrier; benthic organisms live amongst the floc particles as they would other sediments. Phosphorus remains bound in the floc even during seasonal periods of low dissolved oxygen when it would otherwise be recycled back into the water column via internal loading and support algae growth. Alum has been and likely remains the most widely used technique to inactivate sediment phosphorus and reduce internal phosphorus loading in lakes (Welch and Gibbons 2005). The longevity of treatments varies, but typically about 10 years can be expected in lake systems with effectiveness waning over time as the alum floc layer sinks and new sediment with un-bound phosphorus settles and covers the alum layer.

In recent years, lanthanum (La)-amended bentonite clay has been used as an alternative to alum, where La is a metal and a rare earth element. The primary lanthanum-based product, Phoslock™, does not rely upon charge neutralization and flocculation of colloidal sized particulates as does alum; rather the La in Phoslock™ binds with the orthophosphate (PO_4^{3-}) to form an insoluble and biologically inert mineral called rhabdophane ($La(PO_4)\cdot H_2O$). While alum floc particles are "fluffy" and prone to migration along the lakebed by bottom currents, Phoslock™ particles settle out in a fine layer, usually fractions of a millimeter thick, and are not easily disturbed. In relatively shallow lakes with frequent sediment resuspension, Phoslock™ may be more effective at binding and keeping phosphorus in the sediments. Combined dosing of alum and Phoslock™ is also possible and may increase overall efficacy of phosphorus control in eutrophic lakes.

4.2.2.2 Implementation considerations

Alum and Phoslock™ are typically applied from a boat using a slurry of the selected chemical and water, where the slurry is sprayed or injected at the water surface and subsequently settles through the water column. Settling typically requires 24 hours for alum and approximately 2-4 hours for Phoslock™. A staging area for chemicals is needed close to the point where chemicals would be loaded onto boats for application. Dosing of either chemical can be accomplished as a one-time heavy dose (e.g., every 5-10 years), or applied as lighter doses in phases (e.g., 60% in first year, 40% in second year). Lighter doses undertaken across multiple years could reduce logistical constraints posed by higher volumes of heavy doses and could reduce the need for large staging areas. Application costs of multiple lighter doses would be higher, but the costs would be spread across multiple years.

Lake pH and alkalinity are a necessary consideration for alum dosing of a lake. At the pH of most lake and reservoir waters (pH 6 to 8), the insoluble, aluminum hydroxide precipitate $Al(OH)_3$ dominates. This is the form of aluminum that sorbs and inactivates phosphorus. As pH decreases toward 4, soluble intermediate forms of aluminum hydroxide (e.g., ($Al(OH)^{2+}$) become more prevalent, resulting in the release of phosphorus into the water column. At pH less than 4, free aluminum ion (Al^{3+}) dominates, which is toxic to aquatic life (Cooke et al. 2005). In weakly-buffered lakes with a low or moderate alkalinity (<30 to 50 mg/L as $CaCO_3$), alum addition significantly decreases pH and increases the amount of toxic, free aluminum ion. In well-buffered lakes with alkalinity greater than 75 mg/L as $CaCO_3$, including Lake Henshaw (100–200 mg/L as $CaCO_3$), the natural buffering capacity of the water takes up liberated hydrogen ions such that alum addition does not affect pH and hydroxide precipitate ($Al(OH)_3$) will form and inactivate phosphorus. In the case of systems with very high alkalinity, greater concentrations of alum or acidified alum could be used to lower pH into the range that supports the aluminum hydroxide precipitate ($Al(OH)_3$). However, this is not always a feasible solution. For example, experiments conducted for the high alkalinity (~500 mg $CaCO_3$/L) of Lake Elsinore, California, found that only high doses of acidified alum would reduce alkalinity and pH in the lake to appropriate levels. Due to the lack of assurances regarding how long the pH could be maintained and safety concerns associated with potentially high levels of dissolved and total aluminum, it was ultimately determined that Lake Elsinore was not a good candidate for alum treatment (CH2Mhill 2004, Anderson 2002).

Both alum and Phoslock™ applications can be targeted to sediment phosphorus "hotspots" such as deep areas of a reservoir or lake, shallow coves, or littoral shelves. Targeted applications can result in a substantial cost savings although they require spatial resolution on phosphorus concentrations in sediments. Results of recent sediment sampling in Lake Henshaw indicate that total phosphorus (TP) content of sediment varies by location (1,200–4,000 mg/kg dry weight [dw]) with the highest concentration at the fishing dock, and that TP content is high relative to several other productive lakes/ reservoirs in California (600–1,300 mg/kg dw) (Beutel 2021). However, the sample size for the recent sampling event was relatively small (n=4) and further data are needed to determine whether true hotspots are present in Lake Henshaw that could reduce the area of application from most of the sediments (approximately 750 acres), to moderately deep and deep areas (approximately 120 acres) to the deepest areas (approximately 30 acres; Figure 4-2).

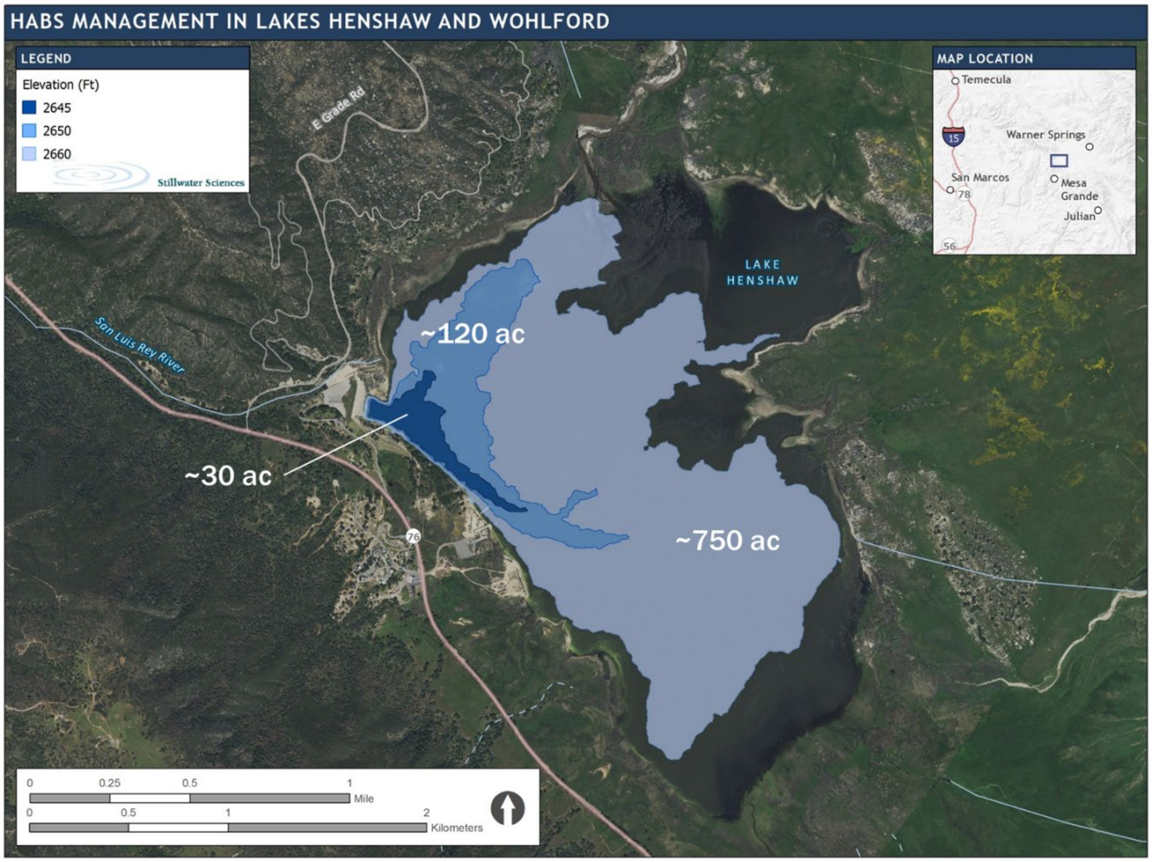


Figure 4-2. Extent of potential sediment areas for phosphorus inactivation/chemical sediment sealing in Lake Henshaw.

4.2.2.3 Anticipated implementation schedule

The permitting and implementation of an in-lake phosphorus inactivation/chemical sediment sealing event is anticipated to require approximately one to two years, depending on which chemical is used (or whether they are dosed in combination). The primary time constraint is acquisition of the required permits (see Section 4.2.2.6), although chemical supply is also a consideration.

4.2.2.4 Compatibility

Chemical sediment sealing for phosphorus inactivation is compatible with the District's water supply objectives as it would reduce internal loading of bioavailable phosphorus that supports HABs in Lake Henshaw (Figure 2-5 and Section 2.2.2.3) and would thus improve the quality of water diverted to the District, Escondido, the Rincon Band, and the La Jolla Band. Bound phosphorus entering Lake Henshaw would be permanently buried in lake sediments.

As discussed above, the main precaution associated with alum use is the presence of free aluminum at low pH (< 6.0), which can be toxic to aquatic life. To maintain the appropriate pH, alum treatments must be chemically buffered. This is common practice for environmental alum applications and would also be relevant for Lake Henshaw. Alum treatments have increased over the past four decades, such that the procedure is now considered to be routine and one of the most commonly used methods of lake treatment. Although La and La compounds (e.g., lanthanum chloride [$LaCl_3$], lanthanum(III) oxide [La_2O_3]) have been shown to bioaccumulate and/or to result in toxicity at elevated concentrations (> 0.5 milligrams per liter [mg/L]) for aquatic species (Herrmann et al. 2016), there are currently no regulatory thresholds for La. Based on laboratory bioassays, Phoslock™ is reported to have low aquatic toxicity potential, particularly at lower application doses (Herrmann et al. 2016, Lürling and Tolman 2010, Afsar and Groves 2009). Overall, use of alum or Phoslock™ at permitted dosing rates would not result in adverse impacts to lake biota, including warm water fish species that currently reside in Lake Henshaw.

Successful use of phosphorus inactivation/chemical sediment sealing requires that use of alum and/or Phoslock™ be combined with source water nutrient control (Section 4.2.1). While internal nutrient loading, particularly in deeper areas of the lake, is likely to contribute a much larger proportion of bioavailable nutrients to the lake than does external loading (i.e., groundwater transfers or runoff) under existing conditions (Figure 2-5 and Section 2.2.2.3), new un-bound phosphorus entering the lake after a chemical sediment sealing dosing event will settle and cover the original sediment layer. Over time, newly deposited un-bound external sources of phosphorus will then be available for internal recycling. Combined, chemical sediment sealing plus source water nutrient control via phosphorus inactivation would improve visual and aesthetic characteristics in Lake Henshaw and in release waters since reduced nutrient cycling would control cyanobacteria blooms, reducing the prevalence of algal scum accumulation.

4.2.2.5 Estimated costs

Estimated costs for alum, Phoslock™, and combined alum/Phoslock™ are presented in Table 4-5 through Table 4-7. The estimated costs are conservative as they are based on the highest values of TP in sediment measured in recent Lake Henshaw sediment sampling (Beutel 2021) and assume heavy dosing of each chemical would be undertaken if used individually and moderate dosing of each chemical would be undertaken if used in combination. Lighter doses could be undertaken more frequently in the lake, although application costs would increase with that approach.

Table 4-5. Estimated costs for a heavy dose of alum for phosphorus inactivation/chemical sediment sealing in Lake Henshaw.

Lake Treatment Area (ac)	TP in Sediment (mg/g dw) [1]	Al:P (molar)	Amount of Alum + Sodium Aluminate Application (gal)	Estimated Cost [2]
30	2.492	20	54,000	$162,000
117	2.492	20	213,000	$639,000
750	2.492	20	1,368,000	$4,104,000

[1] Sediments sampled near the fishing dock in Lake Henshaw (Beutel 2021). Assumes sediment bulk density of 0.05 g/cm^3 and depth of active sediment zone is 10 cm.
[2] Includes product, application and handling, and permit-required monitoring.

Table 4-6. Estimated costs for a heavy dose of Phoslock™ for phosphorus inactivation/chemical sediment sealing in Lake Henshaw.

Lake Treatment Area (ac)	TP in Sediment (mg/g dw) [1]	Phoslock™ (lbs/acre)	Estimated Cost [2]
30	2.492	2,000	$180,000
117	2.492	2,000	$703,000
750	2.492	2,000	$4,500,000

1 Sediments sampled near the fishing dock in Lake Henshaw (Beutel 2021). Assumes sediment bulk density of 0.05 g/cm^3 and depth of active sediment zone is 10 cm.
2 Includes product, application and handling, and permit-required monitoring.

Table 4-7. Estimated costs for a moderate combined dose of alum and Phoslock™ for phosphorus inactivation/chemical sediment sealing in Lake Henshaw.

Lake Treatment Area (ac)	TP in Sediment (mg/g dw) [1]	Al:P (molar)	Phoslock™ (lbs/acre)	Estimated Cost [2]
30	2.492	10	1,000	$176,000
117	2.492	10	1,000	$688,000
750	2.492	10	1,000	$4,416,000

1 Sediments sampled near the fishing dock in Lake Henshaw (Beutel 2021). Assumes sediment bulk density of 0.05 g/cm^3 and depth of active sediment zone is 10 cm.
2 Includes product, application and handling, and permit-required monitoring.

4.2.2.6 Permit requirements

Use of alum for phosphorus inactivation/chemical sediment sealing in Lake Henshaw would require approval from State Water Board through the Individual NPDES permit process. This process is currently anticipated to require 1–1.5 years. The San Diego Regional Water Quality Control Board has developed an NPDES General Permit (No. CAG999003) for lanthanum-modified clays, which is a streamlined pathway and may only require 3–6 months for permit review.

4.2.2.7 Additional information needs

Refinements to the anticipated successful dosing level for either alum or Phoslock™ (or both) could be made using additional data characterizing the spatial patterns of total phosphorus concentrations in lake sediments and the fraction of phosphorus in sediments that is labile and/or loosely adsorbed to iron oxides and most likely to be released as orthophosphate into the overlying water column.

4.2.3 In-lake method 11 – Oxygenation via Speece Cone or SDOX

4.2.3.1 Goals and capabilities

The goal of lake oxygenation is to prevent HABs by controlling oxidation-reduction potential (ORP) in the water and bottom sediments and the associated release of orthophosphate (PO_4^{3-}), ammonia (NH_4^+), dissolved manganese (Mn^{2+}), and dissolved iron (Fe^{2+}), sulfate (SO_4^{2-}), and if present, methylmercury (MeHg), from the reservoir sediments during hypoxic (DO < 2 mg/L) or anoxic (DO = 0 mg/L) conditions. Limiting release of nitrogen and phosphorus reduces the potential for HABs because these nutrients are required for algae and cyanobacteria growth (Figure 2-55). Although release of manganese, iron, and/or sulfate from Lake Henshaw bottom sediments has thus far not caused difficulty for the EVWTP to comply with these constituents' secondary maximum contaminant limits (SMCLs) in treated water, continued releases could become problematic. Methylmercury is a toxic metal that bioaccumulates in the aquatic food web. Oxygenation also provides benefits to general lake water quality and fish populations.

Oxygenation systems are designed and operated to maintain positive dissolved oxygen (DO) concentration in the water column and at the sediment/water interface. In deep reservoirs and lakes, it is common for seasonal thermal stratification to occur, where the upper layer of the water column (called the epilimnion) warms in the spring, while the bottom layer (hypolimnion) remains cool. The epilimnion can descend through the water column and the hypolimnion can become smaller as the spring and summer progress. In shallow lakes, the water column may only weakly stratify or not stratify at all, instead remaining a uniform warm temperature throughout the spring and summer until cooling in the fall. Low or zero DO conditions often occur in the hypolimnion of deep reservoirs and lakes during the summer and/or fall, but they can also be present in shallow warm water columns.

The surface water elevation in Lake Henshaw varies depending on several factors including rainfall, groundwater pumping from the Warner Ranch Wellfield, and releases from the dam (Section 2.2.1). The deepest part of the reservoir is a relatively small area located near the dam where the water depth ranges from 25 ft at low levels to 59 ft at a full pool elevation of 2,690 ft. Historically, this maximum water level occurs infrequently, and the lake level more typically ranges from 2,660 to 2,670 ft. The rest of the lake is broad and shallow with typical depths ranging from 8 to 20 ft.

While oxygenation via Speece Cone has been identified as a HABs prevention method that has a strong potential for success in Lake Henshaw (Table 4-1, Section 4.1), further consideration of the oxygenation alternative has focused on two options: the non-pressurized Speece Cone and the pressurized SDOX system. Both methods involve creating an oxygen-rich water stream that is mixed into lake bottom waters, increasing water column and sediment DO levels without disturbing any thermal stratification that may exist. Both systems consist of two main components – the liquid oxygen (LOX) system located above grade near the lake shore, and the oxygen transfer and delivery system. Both systems involve pumping a small portion of the lake volume, called a sidestream, to create super-saturated oxygenated water that is mixed back into the water column.

Speece Cone

For the Speece Cone approach, oxygen transfer into the sidestream typically occurs underwater, where the Speece Cone assembly skid is anchored to the lake floor. The skid is equipped with an intake screen and submersible circulation pump that moves lake water through the Speece Cone at high velocity, where oxygen bubbles are applied. The oxygenated water is discharged at the bottom of the Speece Cone through a pipeline with a diffuser. Figure 4-3. shows the general arrangement for an ECO_2 Speece Cone, which is manufactured by ECO_2 (Indianapolis, IN).

In some installations, the Speece Cone assembly is located onshore instead of underwater. Lake water is withdrawn and directed through the Speece Cone and then returned to the lake through a diffuser system.

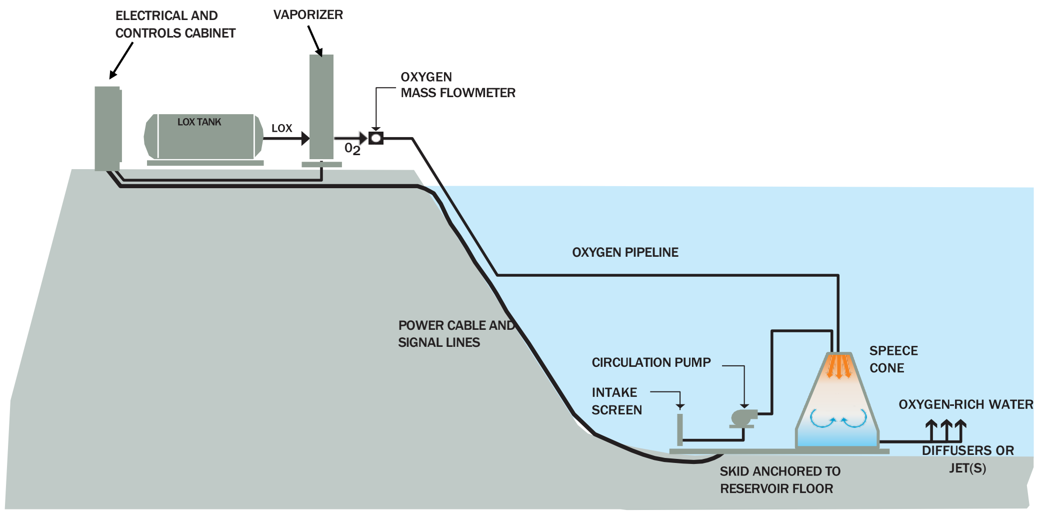


Figure 4-3. Speece Cone schematic.

SDOX

For the SDOX approach, the oxygen transfer into the sidestream occurs above grade near the LOX system (Figure 4-4). The pump lifts water from the lake and the water flows through a pressurized enclosed vessel that is fed with oxygen to create a super-saturated oxygen solution, which is piped back into the lake. Figure 4-4 shows the general arrangement for an SDOX, similar to an SDOX O2®, which is a patented system manufactured by ChartWater BlueInGreen (Fayetteville, AR).

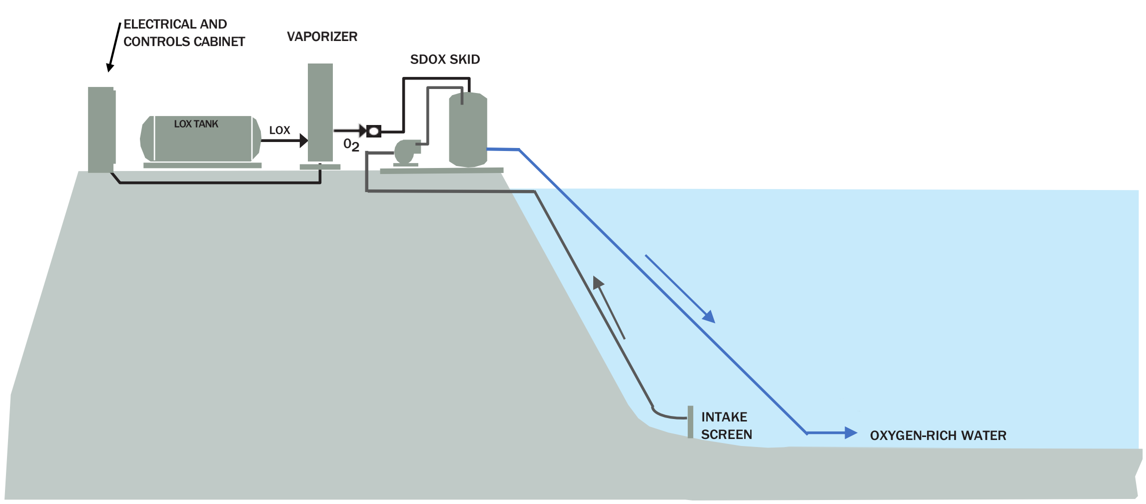


Figure 4-4. SDOX schematic.

4.2.3.2 Implementation considerations

Oxygenation systems require onshore space for the LOX system, road access for LOX tanker truck deliveries, and power for the pumps and controls.

For Lake Henshaw, a Speece Cone could be placed on the lake bottom in the restricted area near the dam or onshore near the LOX system. For the SDOX approach, the system would be located onshore with the LOX system. Potential onshore and in-lake locations are shown in plan-view on Figure 4-5, along with the general bathymetry of the lake. To facilitate the oxygenation of a larger area, two oxygenated plumes would be used, with one oriented toward Carrista Creek and the other toward the upstream San Luis Rey River as shown in the enlarged view on Figure 4-6.

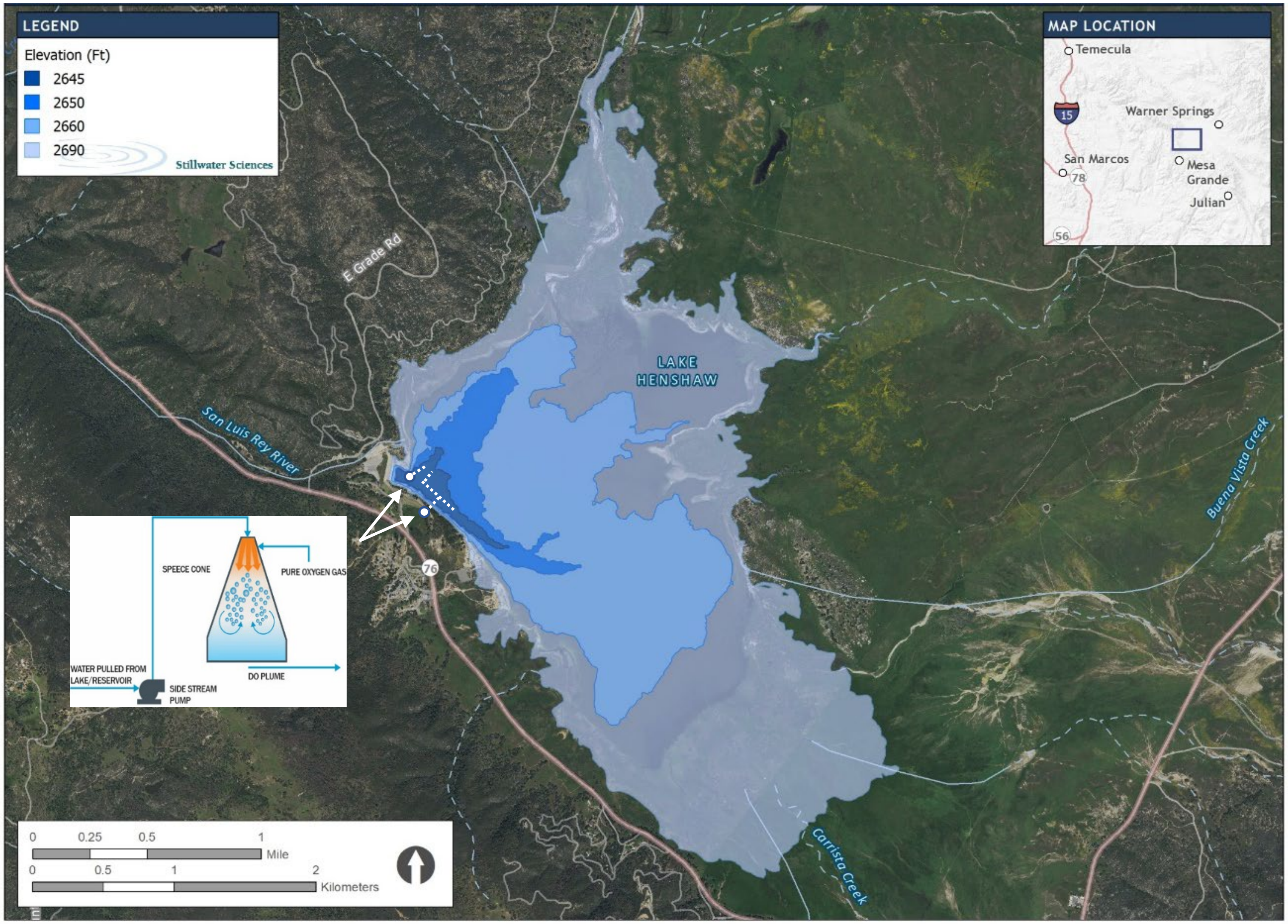


Figure 4-5. Preliminary locations for Speece Cone or SDOX oxygenation systems in Lake Henshaw.

Figure 4-6. Preliminary location for Speece Cone oxygenation system in Lake Henshaw—enlarged view.

Additional chemical treatment

With either the Speece Cone or SDOX approach, an optional alum injection system can be used to enhance orthophosphate removal from the water column by binding and precipitating orthophosphate with aluminum sulfate. The solid precipitate would then deposit into the lake bottom sediments. Alum would be added to the oxygenated stream either within the Speece Cone or just downstream of the SDOX vessel. An alum chemical storage tank and small metering pump can be located onshore near the LOX system.

4.2.3.3 Anticipated implementation schedule

The design, permitting, and construction of an oxygenation system for Lake Henshaw is anticipated to require 2-3 years total, with the construction phase lasting approximately 1 year of the total. The construction schedule would need to be refined in a detailed design phase, with special consideration given to any permitting requirements and associated timeframes.

4.2.3.4 Compatibility

Oxygenation via Speece Cone or SDOX in Lake Henshaw is compatible with the District's water supply objectives as it would improve the quality of water diverted to the District, Escondido, the Rincon Band, and the La Jolla Band by suppressing anoxia in bottom waters and sediments, resulting in decreased nutrient cycling and cyanobacteria growth. Water quality improvements associated with successful oxygenation operations would also be compatible with recreation activities around and on the lake. Oxygenation should improve habitat for the warm water fishery at Lake Henshaw by expanding the volume of oxygenated water available during warmer months of the year, although data are needed to confirm seasonal DO patterns in lake bottom waters (Section 2.2.2.1). Visual and aesthetic characteristics would improve since reduced nutrient cycling would control cyanobacteria blooms, reducing the prevalence of algal scum accumulation.

4.2.3.5 Estimated costs

Capital costs

A Class 5 cost estimate for oxygenation via Speece Cone or SDOX was assembled in accordance with AACE criteria. The accuracy of Class 5 estimates ranges from -50 to +100 percent; therefore, the capital cost estimate applies to either the Speece Cone or the SDOX oxygenation approach. The Lake Henshaw oxygenation cost estimate was scaled from a recent 5 ton/day Speece Cone oxygenation project located in San Pablo Reservoir in northern California, which began construction in January 2022. The Lake Henshaw equipment and installation costs were extracted from the San Pablo Reservoir construction schedule of values, summed, and then scaled for the size of the Lake Henshaw project. Engineering and administrative costs were included as a percentage of construction.

For Lake Henshaw, the estimated system size is 1.6 tons/day, which would oxygenate both the water column (30% of the full 55-ft water depth) and bottom sediments over the moderately deep area of the reservoir (approximately 117 acres). The water column oxygen demand was assumed as 0.2 mg/L/day and the sediment oxygen demand was assumed as 2 $g/m^2/day$. The Lake Henshaw preliminary design criteria should be refined in a future phase of the project using empirically determined oxygen demand of the water column and bottom sediments when those data become available. Table 4-8 presents the San Pablo equipment and installation cost, the scaling factor, and the percentages included in the Lake Henshaw estimate for contingency allowance (40%), engineering costs (15%), start-up and training costs, and various legal/administrative costs typically associated with construction projects.

Table 4-8. Estimated capital costs for a Lake Henshaw oxygenation system via ECO_2 Speece Cone or SDOX O2®(Class 5).

Parameter	Units	San Pablo	Henshaw
Design Capacity	tons/day	5	1.6
Scale factor	-	-	0.46
Capital Cost, 1/2022	$	$ 7,110,000	$ 3,280,000
Engineering, ESDC	15.0%	-	$ 492,000
Contractor General Conditions	15.0%	-	$ 566,000
Start-Up and Training	4.0%	-	$ 174,000
Undesign/Undevelop Contingency	40.0%	-	$ 1,805,000
Building Risk, Liability Auto Insurance	2.0%	-	$ 126,000
Payment and Performance Bonds	1.5%	-	$ 97,000
Construction Cost, 1/2022	**-**	**-**	**$ 6,540,000**

ESDC: Engineering services during construction.

The estimated capital cost is $6.5M in current dollars based on a system sized to deliver 1.6 tons per day of oxygen using LOX delivered by tanker truck as the oxygen source.

O&M costs

Estimated O&M costs for an oxygenation system consist of LOX purchase/delivery costs, power cost for pumping, and labor/general repair costs. Assuming the system operates year-round, the total annual O&M cost is estimated to be approximately $183,000 per year based on the following:

- $72,000/year for LOX (1.6 tons/day at $125/ton of LOX delivered)
- $45,000/year for power ($0.15/kW-hour to run a 45 hp pump motor)
- $66,000/year for labor and general repair (assuming 1% of capital cost)

4.2.3.6 Oxygenation summary

The oxygenation alternatives for Lake Henshaw are summarized in Table 6 below.

Table 4-9. Summary of oxygenation options considered for long-term prevention of HABs in Lake Henshaw.

	ECO_2 Speece Cone	SDOX O2®
Installation Location	submerged or onshore at reduced efficiency	onshore
Full Scale Testing	not readily available	typical service
Capital Cost (-50% to +100%)	$6,540,000	TBD
O&M	$183,000/year	TBD

4.2.3.7 Permit requirements

As lead agency, the District would be required to undertake a project-appropriate CEQA compliance process, and, unless the District is categorically exempt, the project would also require a San Diego County building permit. By siting new facilities judiciously, the District may not need to interface with the California Division of Safety of Dams (DSOD).

4.2.3.8 Additional information needs

The following additional information characterizing year-round lake water quality is needed to accurately size and predict the effectiveness of an oxygenation system for HABs prevention in Lake Henshaw:

1. **To refine system size, the oxygen demand of the water column and the sediment needs to be determined.** Vertical profiles of temperature and DO need to be collected on a regular basis throughout the year to understand oxygen demand within the lake. Replicated sediment samples should be collected at 3 to 4 sites in the lake, including deep and shallow sites, during the period of lowest DO in deeper bottom waters. The samples should be analyzed in the laboratory to provide sediment oxygen demand rates.
2. **To design and locate an effective system, data needs to be collected characterizing the spatial extent of algal blooms to understand where they form and how they spread.** Hypolimnetic oxygenation systems in deep lakes have been proven to mitigate nutrient loading from sediments and reduce algae blooms. Less information is available concerning the effects of oxygenation on broad shallow lakes and the spatial impact of adding oxygen in one location. Understanding where the algal blooms occur is critical information in deciding where to apply oxygen. Numeric modeling of the proposed oxygen application points to estimate the spatial extent of the DO plumes within the lake also would help inform design.
3. **A detailed cost/benefit analysis for both Speece Cone and SDOX approaches should be undertaken.** If the oxygenation alternative is selected, this comparison and cost analysis would guide whether to select one approach before moving forward to final design or to compete them against each other.
4. **A full scale 9-to-12-month operational study would provide exceptionally valuable information on oxygenation effectiveness.** This approach is possible through ChartWater's oxygenation treatment as a service with a dual SDOX O2® system capable of providing a range of 1 to 5 tons/day of oxygen (Figure 4-7). The SDOX O2® equipment and control room are housed within a 20-ft ISO container that can be placed on a gravel pad. The LOX tank and vaporizer system are mounted on an open trailer that can be parked on site (Figure 4-8). The cost for the service, which includes remote monitoring by ChartWater, is approximately $25,000 per month plus the cost of LOX delivered by a local supplier. Additional costs include installing the pump suction and oxygenated water lines in the lake and running power to the site if not already available. Total costs for the study are estimated to range from $400,000 to $600,000, depending on the amount of oxygen used and study length. Currently, Speece Cones are not typically offered or marketed for temporary installations, although full-scale barge installations may be possible and would avoid the need for constructing a concrete pad for a temporary shoreline operational study or submerging the cone within the reservoir in the short-term. Conducting a full-scale operational study would also be a good way to evaluate whether alum addition is needed to reduce internal loading of phosphorus.

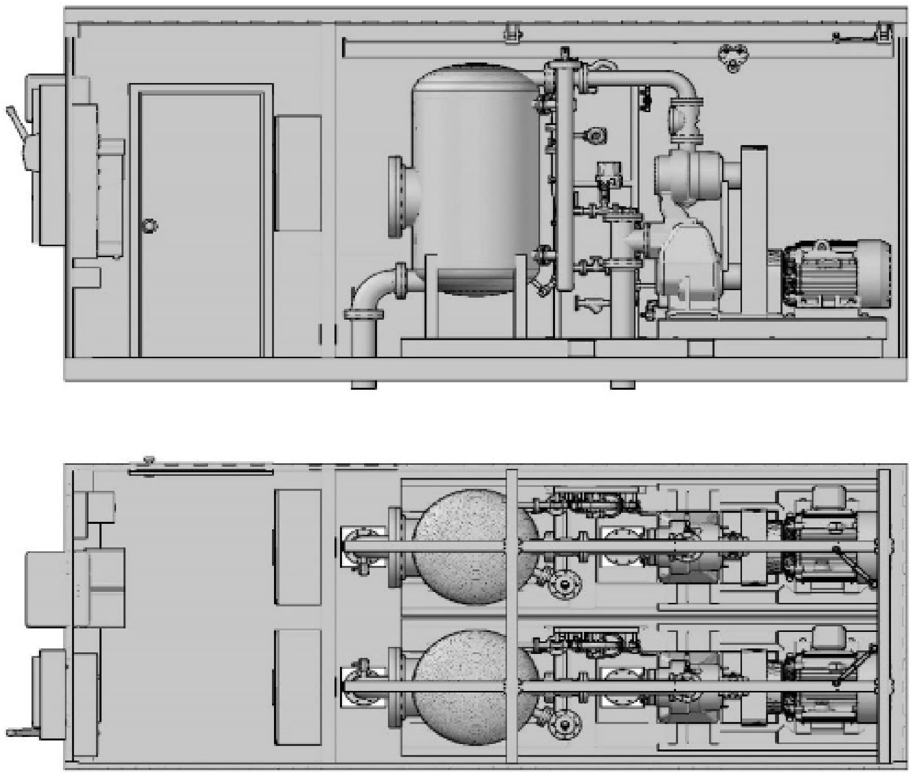


Figure 4-7. Containerized SDOX O2® system.

Figure 4-8. Example trailer-mounted LOX and vaporizer—typical configuration.

4.3 Lake Henshaw Selected Mitigation Methods

4.3.1 Out-of-lake method 8 – Bypass

4.3.1.1 Goals and capabilities

The goal of the bypass pipeline method (bypass) is to provide cyanotoxin-free water downstream of the Henshaw Dam spillway by rerouting groundwater from the Warner Ranch Wellfield around Lake Henshaw and into the San Luis Rey River downstream of Henshaw Dam. The bypass method would bypass flow from the wellfield and provide water for downstream users without interruptions due to HABs. The bypass would provide the District with a reliable and consistent delivery method, with added flexibility regarding the timing of water deliveries, and would reduce evaporative losses that occur within Lake Henshaw, offsetting some costs of implementation. Note that the financial viability of the Local Water System relies on runoff into Lake Henshaw as well as wellfield production, hence the construction of a bypass pipeline would also require a strategy to prevent or mitigate in-lake HABs production. Sections 4.3.1.2 through 4.3.1.8 provide further discussion and analysis of the bypass pipeline as a potential long-term HABs mitigation method for Lake Henshaw.

4.3.1.2 Implementation considerations

Temporary versus permanent bypass use

The bypass can be installed as a *temporary* or *permanent* long-term method. Temporary use of the bypass is defined as a 5-to-10-year service life and would allow the District to convey flows from the Warner Ranch Wellfield to downstream users until one or more long-term in-lake prevention and/or mitigation methods are implemented. Since implementation timing for a temporary bypass would be in the year 2024 or beyond, the temporary bypass is still considered a long-term method (Section 4.1). A temporary bypass could be used in conjunction with short-term and other long-term HABs prevention and mitigation methods (Table 4-10).

Permanent use of a bypass is defined as a 50+ year service life and would allow the District to convey flows long-term from the Warner Ranch Wellfield downstream of the Henshaw Dam spillway in conjunction with short-term and other long-term HABs prevention and mitigation methods (Table 4-10).

Table 4-10. Use of temporary and permanent bypass options with other methods for long-term HABs prevention and mitigation in Lake Henshaw.

Bypass Use in Conjunction with Other Short- and Long-term Prevention and Mitigation Methods		Temporary 5–10 Years Max Service Life (short[1]/long[2])	Permanent 50+ Years Service Life (short/long)
Out-of-lake methods			
1	Source water nutrient control – phosphorus inactivation of Warner Ranch Wellfield outflow	N/A/Yes	N/A/ Yes
In-lake methods			
10	Algaecide treatment	Yes/Yes	Yes/Yes
13	Sediment sealing (e.g., alum, Phoslock™)	N/A/Yes	N/A/Yes
11	Hypolimnetic oxygenation system (HOS)	N/A/Yes	N/A/Yes

[1] Short term is defined as occurring prior to 2024 and long term is defined as occurring in the year 2024 or later.

Preliminary alignment
The existing wellfield infrastructure conveys flows through a series of gravity canals and concrete pipelines that discharge through a dissipation structure into a tributary of the San Luis Rey River (Figure 2-2). The terminus of the wellfield canal is a buried 48-in. inner diameter (ID) reinforced concrete pipe (RCP). From the discharge, raw water is conveyed downstream within the creek bed into Lake Henshaw. The bypass pipeline would start at a constructed diversion structure at the wellfield terminus point. Using sluice gates for flow control, water would be diverted into the bypass.

The preliminary bypass alignment is approximately 17,000 ft (3.2 miles) long, traveling southwest from the wellfield terminus and around the northwestern side of Lake Henshaw, conveying flow over the Lake Henshaw dam spillway (Figure 4-9). The pipeline would terminate at the top of the spillway. A disconnect flange would be provided to allow the last segment of bypass pipeline to be removed and eliminate any impedance of flows over the dam if the spillway is activated. Details regarding anchoring the final pipe segment to the spillway should be considered in a future design phase. The hydraulics of the bypass pipeline are evaluated later in this section.

A creek crossing is required at a tributary of the upper San Luis Rey River (Figure 4-9 and Figure 4-10). This conceptual level alignment assumes trenchless installation methods below the creek bed along with jack and bore tunneling using a drive length of approximately 500 ft long and a depth of 20 ft of cover. Future design phases should evaluate other tunneling methods, including horizontal directional drilling (HDD), and explore whether the creek can be crossed via open-cut construction methods during dry- or low-flow periods in the creek.

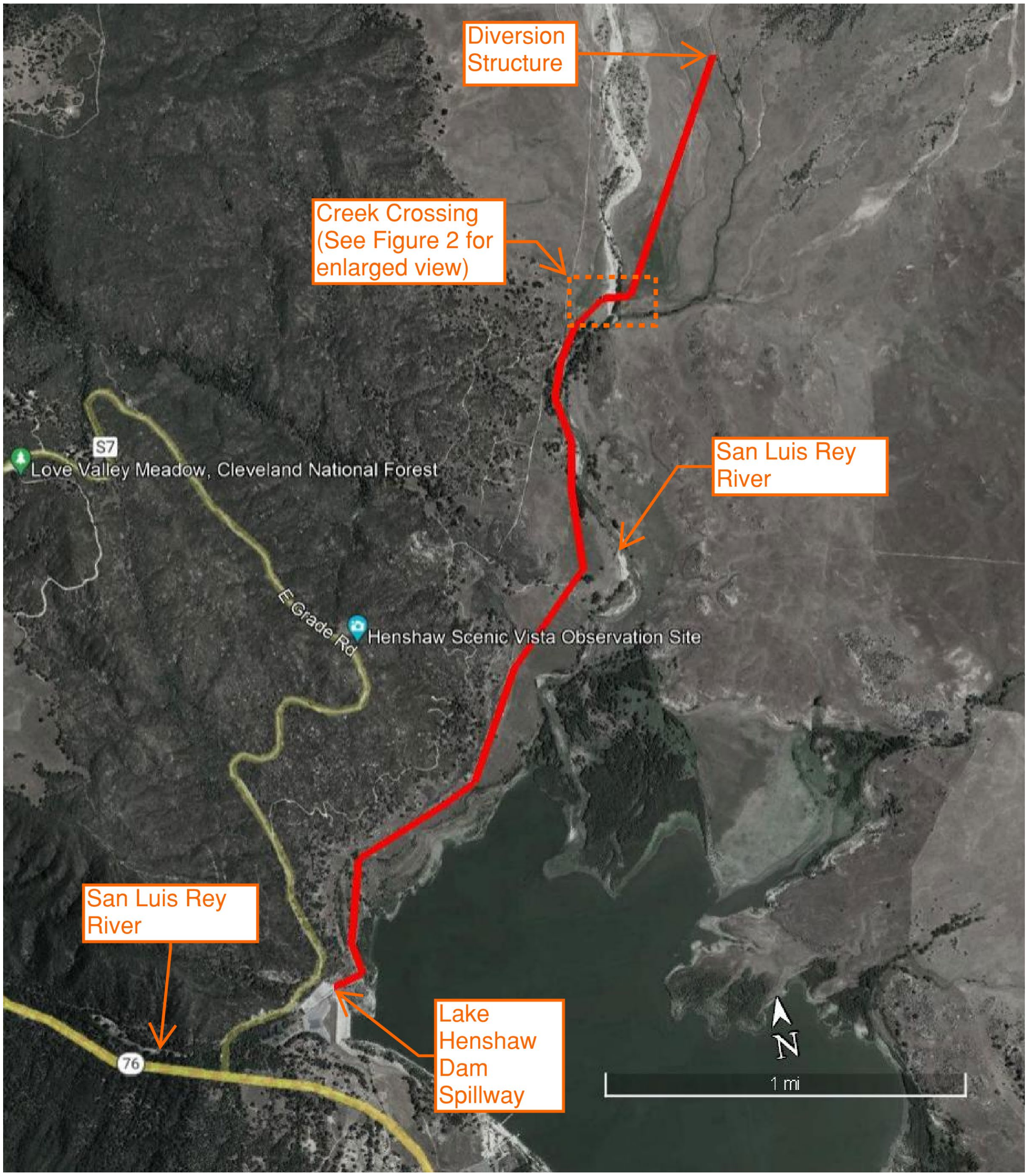


Figure 4-9. Lake Henshaw bypass pipeline preliminary alignment.

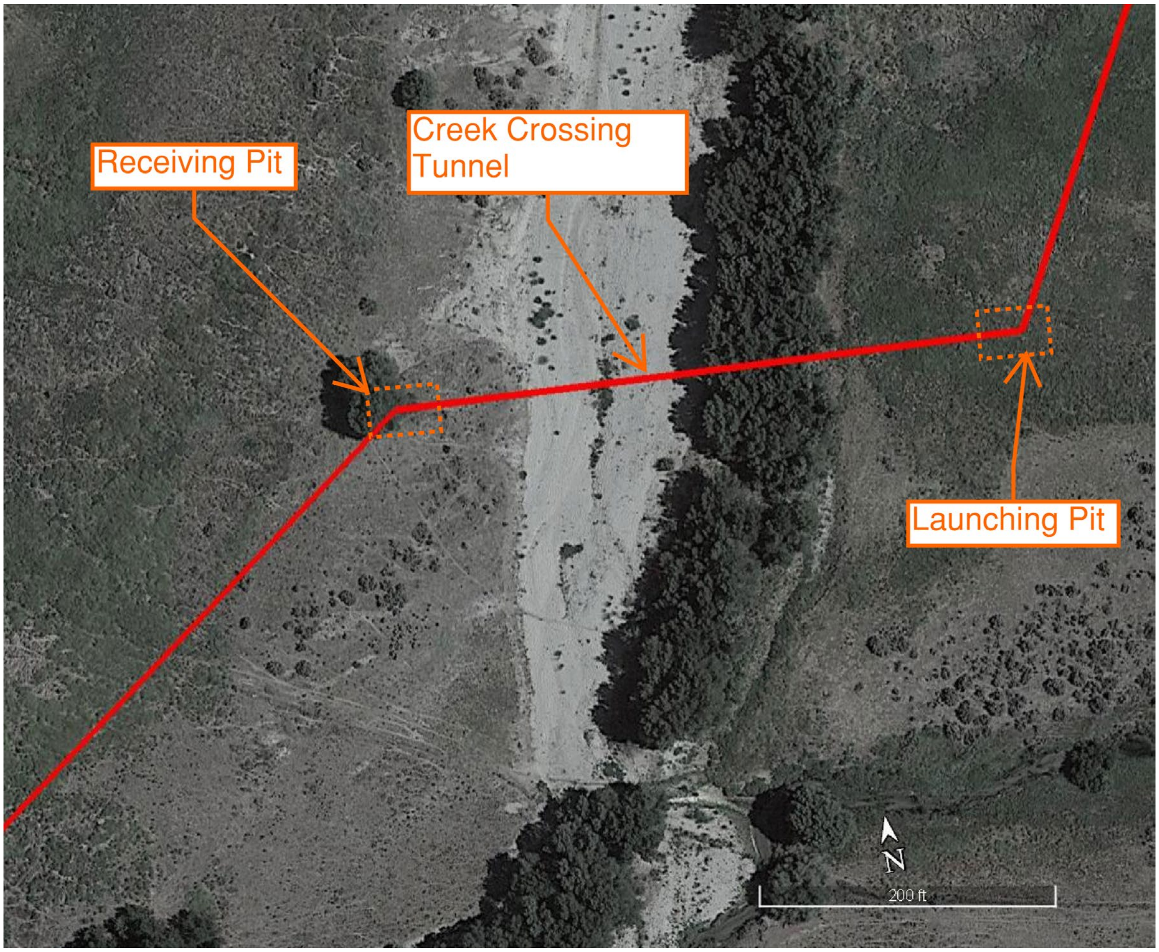


Figure 4-10. Lake Henshaw bypass pipeline creek crossing for preliminary alignment.

Warner Ranch Wellfield

Based on Warner Ranch Wellfield production data provided by the District, historical operations of the wellfield have produced an average annual flow of 7,500 AFY over the course of 12 months since 1953; however, since 2000 average annual production has been less at 6,800 AFY (Table 4-11). Wellfield water outflows are subject to infiltration within the San Luis Rey creek bed upstream of Lake Henshaw and evaporation within the lake itself before waters are released for deliveries to the Bands, Escondido, and the District.

Table 4-11. Warner Ranch Wellfield production over 12 months.

Yield (AFY)	Historical 1953–2020	Current 2000–2020
Minimum	0	300
Average	7,500	6,800
Maximum	18,900	14,700

A recent groundwater modeling effort (Todd Groundwater and Dudek 2018) to characterize the Warner Basin determined that the maximum sustainable yield of the Warner Ranch aquifer is 9,125 AFY. Improvements to the wellfield and its infrastructure would be required to produce flows of 9,125 AFY, including developing deeper wells, since the District has noted that the current wells are pumping near the water table level. The current wells have been in operation for over 40 years so redeveloping the wellfield will be required for its continued, long-term use. While redeveloping the wellfield is distinct from HABs considerations, preliminary costs for developing a new wellfield are presented below to facilitate the District's long term planning efforts.

A new wellfield would entail infrastructure costs, including but not limited to those listed below, as well as life-cycle O&M costs. Section 4.3.1.5 presents a preliminary estimate of costs for a new wellfield, although a future study is required to fully evaluate the existing wellfield and infrastructure improvements that would be required to attain the maximum sustainable yield.

- New wells and pumps
- Canals connecting the new wells to existing wellfield canals
- Possible upsizing of existing canals
- Dirt access roads to the new wells
- Electrical connections to the new wells
- Upgrades to existing electrical system and replacement of existing meters

While a permanent bypass installation would be able to utilize future improvements to increase wellfield production, due to the short service-life of a temporary bypass installation (i.e., 5 to 10 years), wellfield improvements presumably would not be implemented in time and thus the temporary bypass option would be limited to the current wellfield yield (Table 4-11).

Design flows and piping

Design flows

The existing Warner Ranch Wellfield pumping is limited by the depths of the wells and the size of the well pumps. During years when groundwater augmentation of natural runoff is needed, wellfield pumping begins in the spring with continuous operation until October 1 (i.e., start of water year) (see also Section 2.1.2). The District typically releases water deliveries downstream of Henshaw Dam in the summer months, a timing that aligns with the La Jolla Band's recreational use and with the District's and Escondido's peak water demands. Wellfield flow is subject to evaporation losses from Lake Henshaw throughout the remainder of the year. Due to recent HABs events in Lake Henshaw (2020–2021), deliveries were either unable to be made at all or had to be made during the winter months when cyanobacteria blooms had subsided. Delivery amounts during the winter are limited by the local water blending constraints at EVWTP.

The *Groundwater Modeling Report on Warner Ranch Basin* (Todd Groundwater and Dudek 2018) explores the option of producing the wellfield flows during a 4-month window in lieu of the historical 12-month timeframe. This shortened window aligns with the preferred summer delivery timeframe and would effectively limit the amount of evaporation losses from Lake Henshaw.

A bypass pipeline would give the District the flexibility to produce the wellfield flows within a 4-month period or up to a full 12-month period, and thus would provide a consistent and reliable water delivery source. The design flows considered for this assessment of the bypass method are based on the more conservative 4-month period:

- 4,760 AF over 120 days—down-sized alternative for cost saving purposes

- 7,140 AF over 120 days—based on existing annual wellfield production
- 9,125 AF over 120 days—maximum allowable wellfield production

Wellfield improvements would be required for flow periods of 4 months and/or production beyond the current 7,500 AFY limit. Section 4.3.1.5 presents a preliminary range of wellfield improvement costs.

Pipe size

The terminus of the wellfield canal where the bypass tie-in would take place is 48-in. ID RCP. Corresponding pipe sizes for the 4-month design flows are presented in **Table 4-12**. If no wellfield improvements are made, the bypass size would be limited by the existing wellfield production rates (7,500 AFY) and thus the *temporary* bypass pipeline is only proposed in 36-in. ID and 42-in. ID.

Table 4-12. Lake Henshaw bypass pipe sizes and flow rates for design delivery flows.

Pipe Size (in)	Assumed Deliveries: Flow (AF)/Period (days)	Flow Rate (cfs)
36-in. ID	4,760 AF/120 days	20
42-in. ID	7,140 AF/120 days	30
48-in. ID	9,125 AF/120 days	40

In lieu of a single bypass pipeline, future design phases should also evaluate the use of multiple, smaller diameter, parallel pipes. Smaller parallel pipes could provide further cost savings related to materials, installation (i.e., welding), and construction method (i.e., open cut or trenchless). Smaller pipes could be installed above-grade, buried, or placed within a berm. A buried or bermed installation (similar to the existing wellfield canals) should incur less risk due to damage or fires and could be comparable in cost to above-grade pipes due to the smaller size. Environmental conditions, such as the range of temperatures and risk of fire hazard, should be considered in the design of the bypass pipeline.

Pipe material

Multiple pipe materials would be suitable for use as a bypass pipeline. For a *temporary* bypass pipeline, material choice should target a lower material and installation cost. High density polyethylene (HDPE) is recommended as a cost-effective pipe material for temporary use and can be used in above-grade applications for up to 5 to 10 years. Above-grade HDPE is subject to ultraviolet (UV) degradation over time, leading to a shorter service life and higher O&M costs over time so it is not suitable for use above grade in the long term. For HDPE use with raw water, the recommended material is a solid wall, butt welded, PE4710 pipe.

A *permanent* bypass installation should target a material that is durable and performs well in the long run with minimal O&M costs over time. Pipe materials such as steel, ductile iron pipe (DIP), or concrete may perform better in a life-cycle cost-benefit analysis due to the longer service lives and reduced O&M needs of these materials. Above-grade or buried/partially buried applications are also applicable for these materials. HDPE is also suitable for long-term use as long as it is buried and would thus have no exposure to UV.

The ultimate pipe material selection should be based on the desired service life (i.e., temporary versus permanent) and installation method (above-grade, buried, or bermed). This bypass analysis assumes temporary above-grade HDPE for costing purposes. Section 4.3.1.5 presents estimated capital and O&M costs.

Hydraulics

HDPE bypass pipe sizes with IDs of 36-in., 42-in., and 48-in. were evaluated for hydraulic capacity, with corresponding nominal pipe sizes of 42-in., 48-in., and 54-in., respectively, for DR17 PE4710 pipe. Pipe diameters are referred to by their nominal size in the remainder of Section 4.3.1.2. Other pipe material options, such as steel, ductile iron, and plastic pipe, are expected to produce very similar hydraulic results as HDPE. Figure 4-11 shows that there is sufficient head to operate the Lake Henshaw bypass by gravity to convey flows from the Warner Ranch Wellfield diversion structure to the Henshaw Dam spillway for each pipe diameter considered. **Table 4-13** presents a summary of hydraulic conditions and assumptions for the hydraulic grade line (HGL).

Table 4-13. Summary of hydraulic conditions for Lake Henshaw bypass.

Nominal Pipe Size (in.)	Assumed Deliveries: Flow (AF)/Period (days)	Flow Rate (cfs)	Velocity (ft/sec)	Maximum Head (ft)
42-in. (36-in. ID)	4,760 AF/120 days	20	2.70	30
48-in. (42-in. ID)	7,140 AF/120 days	30	3.10	
54-in. (48-in. ID)	9,125 AF/120 days	40	3.20	

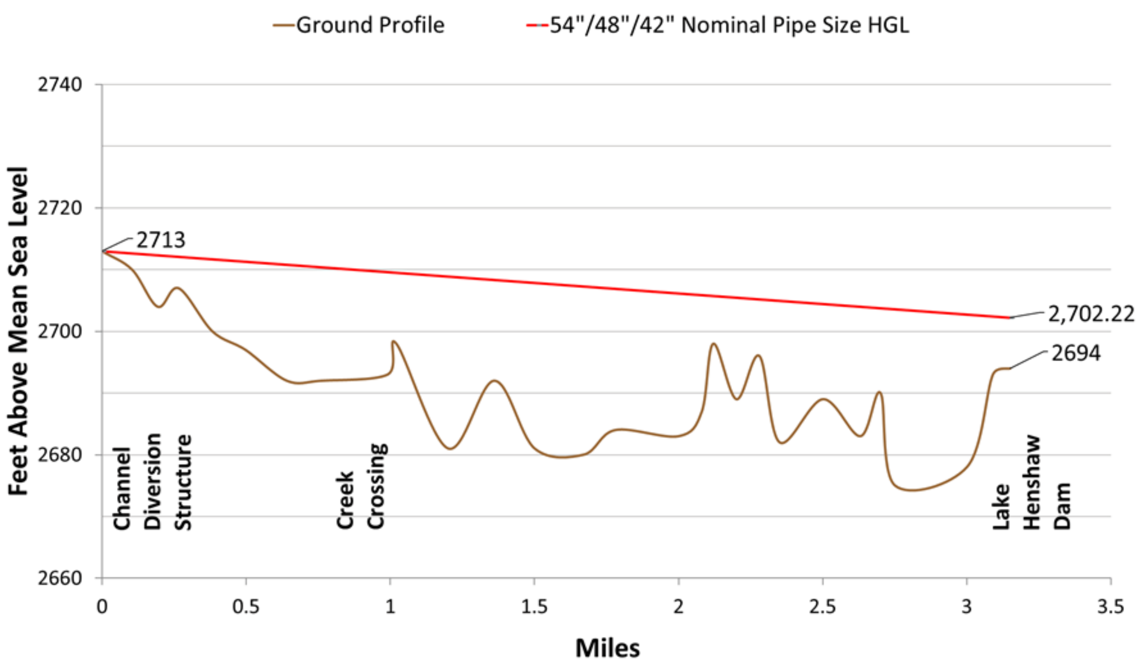


Figure 4-11. Bypass pipeline hydraulic grade line (HGL) for 42-in., 48-in., and 54-in. nominal diameter HDPE.

The following appurtenances are recommended along the bypass pipeline to improve performance and allow for O&M activities:

- Combination air release valves (CARV) at high points to prevent air entrainment in the pipeline.
- Blowoff's (BO) at low points to allow drainage of the pipeline for maintenance.
- A pressure control valve at the end of the pipeline to create backpressure to keep the pipeline full at the intermediate high points along the alignment to prevent valve chatter of the CARVs.

Local yield

Currently, wellfield production plus surface runoff into Lake Henshaw provides local raw water to the District, City of Escondido, and the Bands. Per the Rincon Entitlement, the Bands are allotted a delivery amount each year based on a rolling average of the previous two years of yield from the Warner Basin. After the Rincon Entitlement waters are provided, the remaining yield is conveyed to and treated at EVWTP before being split 50-50 with Escondido. The District's 30-year historical average local yield from the Local Water System is approximately 5,000 AFY.

Table 4-14 shows the potential local yield to the District from the bypass (wellfield only) based on the 4-month production window. Table 4-14 also shows the expected yield if the maximum 9,125 AFY production was achieved using the smaller pipe sizes over a longer time period. Based on cyanotoxin data collected for the period 2020–2021 (i.e., prior to any in-lake treatment of HABs), an approximate 10:1 blend of bypass water and lake water may be sufficient during all but the peak bloom periods to allow release waters from Henshaw Dam to remain under CCHAB posting limits (Table 4-15). Successful in-lake treatment to reduce cyanotoxin concentrations would reduce the need for blending of bypass water and lake water to dilute cyanotoxins in release waters.

Table 4-14. Potential local yield from Lake Henshaw bypass (Warner Ranch Wellfield only).

Nominal Pipe Size (in.)	**42” (36” ID)**			**48” (42” ID)**		**54” (48” ID)**
Wellfield Production Rate	4,760 AF/ 4 months	7,140 AF/ 6 months	9,125 AF/ 8 months	7,140 AF/ 4 months	9,125 AF/ 6 months	9,125 AF/ 4 months
Rincon Entitlement [1] (AFY)	Min: 1,100 Avg: 2,500 Max: 3,600					
Remaining Yield [2] (AFY)	Min: 1,160 Avg: 2,260 Max: 3,660	Min: 4,640 Max: 3,540 Avg: 6,040	Min: 5,525 Avg: 6,625 Max: 8,025	Min: 4,640 Max: 3,540 Avg: 6,040	Min: 5,525 Avg: 6,625 Max: 8,025	Min: 5,525 Avg: 6,625 Max: 8,025
VID Yield (50%) [2,3] (AFY)	Min: 580 **Avg: 1,130** Max: 1,830	Min: 1,770 **Avg: 2,320** Max: 3,020	Min: 2,763 **Avg: 3,313** Max: 4,013	Min: 1,770 **Avg: 2,320** Max: 3,020	Min: 2,763 **Avg: 3,313** Max: 4,013	Min: 2,763 **Avg: 3,313** Max: 4,013
Wellfield Improvements Required?	Yes [4]	Yes	Yes	Yes	Yes	Yes
VID Yield + Henshaw blend [5]	Min: 818 **Avg: 1,368** Max: 2,068	Min: 2,127 **Avg: 2,677** Max: 3,377	Min: 3,219 **Avg: 3,769** Max: 4,469	Min: 2,127 **Avg: 2,677** Max: 3,377	Min: 3,219 **Avg: 3,769** Max: 4,469	Min: 3,219 **Avg: 3,769** Max: 4,469

1 For the period 2000–2021. Minimum Rincon Entitlement corresponds with maximum remaining yield and vice versa.
2 Yield shown does not account for the reduction of evaporative losses in Lake Henshaw; effective yield would be higher. Evaporative losses would still occur downstream in the Escondido Canal and Lake Wohlford.
3 Split 50-50 with Escondido.
4 For a temporary installation with no wellfield improvements, the production would take place at the current rates over 12 months. Note that the lack of a need for wellfield improvements is theoretical for the temporary installation option since the current wells have been in operation for over 40 years. Redeveloping a new wellfield will be required for its continued, long-term use regardless of bypass installation or implementation of other HABs prevention and mitigation methods.
5 10:1 blend from Lake Henshaw would be added prior to delivery of Rincon Entitlement waters. Yield shown here is the District’s yield after deliveries to the Bands and City of Escondido.

Table 4-15. Calculation of dilution factors for a mixture of bypassed groundwater from the Warner Ranch Wellfield and Lake Henshaw water relative to CCHAB cyanotoxin trigger levels.

Microcystin (ug/L)	Dilution Factor[1] to Achieve <0.8 ug/L (No CCHAB Posting[2])	Dilution Factor[1] to Achieve 0.8 to 6 ug/L (CCHAB Caution[2])	Dilution Factor[1] to Achieve 6 to 20 ug/L (CCHAB Warning[2])
0.15	-	-	-
0.5	-	-	-
1	1	-	-
10	13	2	-
100	130	17	5
1,000	1,300	170	50
Anatoxin-a (ug/L)	Dilution Factor[1] to Achieve <0.15 ug/L (No CCHAB Posting[2])	Dilution Factor[1] to Achieve 0.15 to 20 ug/L (CCHAB Caution[2])	Dilution Factor[1] to Achieve 20 to 90 ug/L (CCHAB Warning[2])
0.03	-	-	-
0.15	1	-	-
0.5	4	-	-
1	7	-	-
10	71	-	-
100	710	5	1
1,000	7,100	50	11

[1] Dilution factor calculated as Warner Ranch Wellfield flow: Lake Henshaw flow, rounded to one or two significant figures. Grey shaded cells were used to approximate a 10:1 dilution to account for typical non-peak bloom period microcystin and anatoxin-a concentrations in Lake Henshaw under existing conditions (see also Section 2.2.2.5).

[2] See Table 1-1 for details regarding CCHAB trigger levels for human and animal health.

Based on the aforementioned analysis, the following conclusions can be made regarding the use of a *temporary* or *permanent* bypass for long-term HABs mitigation in Lake Henshaw and the District's local yield:

- Relying on the bypass pipeline alone will not meet historical yields.
- Maximizing Warner Ranch Wellfield production alone will not maintain historical yields.
- Blending bypassed Warner Ranch Wellfield water with untreated Lake Henshaw water will not meet historical yields.
- To maintain historical yields, the bypass method would need to be used in conjunction with one or more in-lake HABs prevention and/or mitigation methods.

4.3.1.3 Anticipated implementation schedule

The design, permitting, and construction of a Lake Henshaw bypass pipeline is anticipated to require 2–5 years, with the construction phase comprising 1–2 years of the total. This estimated timeframe does not include associated improvements to the Warner Ranch Wellfield. The construction schedule would need to be refined in a detailed design phase, with consideration given to any special permitting requirements/timeframes.

4.3.1.4 Compatibility

Bypass implementation is compatible with the District's water supply objectives as it would improve the quality of water diverted to the District, Escondido, the Rincon Band, and the La Jolla Band. Because the bypass method would bypass flow from the wellfield and provide water for downstream users without interruptions due to HABs in Lake Henshaw, this long-term method would provide the District with a reliable and consistent delivery method with added flexibility regarding the timing of water deliveries.

The bypass would also reduce evaporative losses that occur within Lake Henshaw since lake surface area would decrease in years in which there are limited or no runoff events from Warner Basin precipitation that would otherwise increase water levels in the lake. As discussed in Section 2.2.1, monthly evaporative losses can comprise a large fraction of monthly wellfield production, where the median evaporation loss from Lake Henshaw for the period of record (1953–2020) is 335 AF/month (i.e., 50% of values are larger and 50% of values are smaller than this value) which is roughly half of the median wellfield production at 695 AF/month (Table 2-2). A reduction in these evaporative losses would reduce ongoing costs of groundwater pumping and offset some of the bypass implementation costs discussed in Section 4.3.1.5.

A decrease in Lake Henshaw surface area and water depth through use of the bypass method would reduce the extent of habitat for the warm water fishery, including largemouth bass, bluegill and crappie, and a smaller, more shallow lake is likely to experience warmer water temperatures. Further study is needed to understand how a smaller, more shallow lake is likely to affect the current populations of the various warm water fish in Lake Henshaw, and their habitat use by life-history stage (e.g., eggs, juveniles, adults). Lake water temperatures currently exceed 80 ºF for several weeks each year in June through August or September (Figure 2-24 and Figure 2-26), and further increases may push peak water temperatures above 85 ºF and could increase evaporation rates in the warmest months relative to current conditions, further exacerbating lake volume losses and potentially increasing salinity and/or total dissolved solids. Thus, bypass operations would require that the District first determine a minimum pool for Lake Henshaw that would support viability of general lake ecological functions, including habitat for fish species.

With respect to recreational uses of Lake Henshaw, although groundwater inputs of nitrate (NO_3^-) and orthophosphate (PO_4^{3-}) to Lake Henshaw (Section 2.1.3) would cease with a complete bypass, nutrients already present in the Lake Henshaw water column, and those released from the sediments, would concentrate in the smaller lake volume, which would be favorable for cyanobacteria. Thus, in-lake visual and aesthetic characteristics would not be likely to improve through use of the bypass method since large algal accumulations would still occur and cyanotoxins would still be produced in the lake, resulting in health advisory postings for recreational users. If bypass were to be combined with in-lake prevention and/or mitigation methods, any chemical costs for in-lake dosing (e.g., algaecides, alum, Phoslock™, oxygen) would be expected to decrease relative to current cost considerations for these methods, since lake volume, surface area, and sediment area generally would be smaller.

Lastly, although bypass implementation would improve the quality of water diverted to the District, Escondido, the Rincon Band, and the La Jolla Band, and would provide flexibility regarding the timing of water deliveries, it would not provide flexibility with respect to the local water yield received by the District. As discussed in Section 4.3.1.2, bypass pipeline size and wellfield production rates would be the limiting factors for local yield, regardless of wet years or dry years in the Warner Basin. While an approximate 10:1 blend of bypass water and lake water may be sufficient during all but the peak bloom periods to allow release waters from Henshaw

Dam to remain under CCHAB posting limits (Table 4-15), in order to be fully compatible with the maximum potential local water supply, the bypass would need to be used in conjunction with in-lake prevention and/or mitigation methods to reduce or eliminate cyanotoxin concentrations throughout the year so that additional lake water could be released along with bypass water.

4.3.1.5 Estimated costs

Estimated capital and O&M costs are presented below for the temporary 42-in. and 48-in nominal above-grade HDPE bypass pipelines. Permanent bypass costs have been interpolated from the District and City of Escondido's recent San Pasqual Undergrounding Project that was bid in 2021. Preliminary wellfield improvement costs are also presented for planning purposes.

Capital costs

A Class 5 cost estimate was assembled in accordance with the Association for the Advancement of Cost Engineering International (AACE) criteria. A Class 5 estimate is defined as a conceptual level or project viability estimate where engineering is 0 to 2 percent complete. Class 5 estimates provide planning-level cost scopes and allow for a high-level evaluation of methods. Expected accuracy for Class 5 estimates typically ranges from -50 to +100 percent.

The Lake Henshaw bypass cost estimate is based on the proposed preliminary alignment for a temporary 42-in. nominal (36-in. ID) above-grade HDPE installation (Figure 4-9 and Figure 4-10). The temporary bypass cost estimate includes the below items, and does not include any wellfield improvement costs:

- 42-in. nominal (36-in. ID) HDPE pipeline with 3-in wall thickness
- Creek crossing tunnel via jack and bore
- Wellfield terminus diversion structure
- Appurtenances
  - saddle/auger anchors
  - air release valves (4-in. assumed)
  - drain blowoffs (4-in. assumed)
  - disconnecting flange
  - pressure control valve

An alternative diameter of 48-in. nominal (42-in. ID) for the temporary above-grade HDPE bypass pipeline was also evaluated. Table 4-16 provides a comparison of costs between both pipe diameter alternatives. The 48-in. nominal cost was produced by taking the estimated cost of the 42-in. nominal pipe and scaling it up based on cost/in-diameter/linear feet. The cost shown below is for a temporary bypass pipeline project only and does not include improvements to the Warner Ranch Wellfield.

Table 4-16. Estimated capital costs for temporary above-grade HDPE Lake Henshaw bypass pipeline (Class 5).

Nominal Pipe Size (in)	Estimated Pipe-Only Cost ($)	Estimated Total Capital Cost ($)	Estimated Total Cost Range (-50% to +100%)
42 (36-in. ID)	$18M	$22M	$11M–$44M
48 (42-in. ID)	$21M	$25M	$12.5M–$50M

Costs for a permanent bypass method have been projected using recent bids (2021) from the District's and Escondido's San Pasqual Undergrounding Project (SPUP). SPUP includes the construction of approximately 1.5 miles of buried 60-in. steel pipeline with a trenchless tunnel, appurtenances, box culverts, vegetation removal, and existing infrastructure abandonment. These features can be used to project an order of magnitude cost for the permanent bypass method. Items within the SPUP bids not relevant to the Lake Henshaw bypass pipeline, such as the desilting basin, water services, and asphalt pavement, have been excluded from the cost estimates for 42-in., 48-in., and 54-in. nominal bypass pipeline sizes (Table 4-17).

Note that the projected capital costs for a permanent Lake Henshaw bypass pipeline are not specific to the pipeline installation and are intended to provide a high-level and approximate cost range for the different pipe sizes considered. As the projected capital costs in Table 4-17 are likely to be low for the below reasons, it would be prudent to assume higher costs for planning purposes:

- Bypass pipeline cost projections are based on the SPUP winning (lowest) bid. There were eight other bids with a higher price; the median bid was 13% higher and the high bid was 31% over the low bid.
- The SPUP bids are considered to be very competitive. A future bypass pipeline project may not receive nine bids.
- The construction market is very high right now and costs have been increasing at unprecedented rates. It is difficult to predict what will happen in future years, but for the past two years there has been minimal to no relief in escalating costs.

Table 4-17. Projected capital costs for a permanent Lake Henshaw bypass pipeline based on SPUP bids.

Nominal Pipe Size (in.)	42" (36" ID)	48" (42" ID)	54" (48" ID)
Estimated cost	$34M	$38M	$43M

Future design phases should evaluate the following bypass design aspects to determine potential cost savings or increases in the estimated costs shown above. A Class 5 cost estimate should also be developed for a permanent bypass installation.

- Two smaller parallel pipes
- Other pipe materials
- Buried or partially buried installations
- HDD and open-cut creek crossing methods
- Pipeline anchoring at spillway
- Wellfield improvements needed

O&M costs

O&M requirements for a bypass pipeline are dependent on the pipe material selected (e.g., HDPE, steel, DIP, concrete) and installation type (i.e., above-grade versus buried). This bypass analysis assumes a temporary installation of above-grade HDPE pipeline. O&M activities are expected to be minimal in the first five years and would include the following:

- Walking the alignment to inspect the pipe and appurtenances;
- Patching leaks; and,

- Occasional flushing of the pipe due to periods of inactivity (chemicals may be required).

Use of above-grade HDPE pipeline for longer than the 5 to-10-year life span can incur significantly more O&M requirements. As the pipeline continues to be open to the environment, ultraviolet (UV) light begins to degrade the HDPE pipe and biological growth may start to develop in the pipeline. These factors would increase the O&M cost and require the following tasks to maintain a functioning pipeline:

- More frequent and thorough exterior pipeline inspections including caliper (roundness) and UT (wall thickness);
- Interior pipeline inspections including pigging (cleaning and caliper gauging) and 2D laser profiling for deformation/ovality, if unable to access from exterior inspection; and,
- Repairing electrofusion joints.

An annual O&M cost of approximately 1% of the capital cost, or $220,000 per year on average, is assumed over the 5 to 10-year service life. 1% of capital cost is the typical estimate used at this planning level stage, however use of District staff for these O&M activities may reduce the annual cost.

Future design phases should evaluate O&M costs for permanent installations, including the use of other pipe materials, buried or partially buried installations. O&M for other pipe materials, such as steel or DIP, are expected to be less than HDPE and would include at a minimum walking the alignment to inspect the pipe and appurtenances and repairing leaks (for above-grade installations). Occasional flushing of the pipe due to periods of inactivity is expected for either buried or above-grade installations.

Wellfield improvement costs

The Warner Ranch Wellfield improvements required to support a Lake Henshaw bypass pipeline and local yields that exceed the existing wellfield production and 48-in. ID pipe size are dependent on the production goal over a period of time and the size of the bypass pipeline. As noted above, the level of required improvements mean that a new wellfield would need to be developed. Table 4-18 lists initial calculations for the number of wells required and an approximate cost for the different bypass scenarios. The costs shown are preliminary and based on the following assumptions:

- Flow rate of each well is 800 gallons per minute (gpm);
- None of the existing wells of the wellfield will be reused;
- Approximate cost per well is $1M; includes developing the well and the pump;
- Additional 50% for other capital costs – electrical, service road, conveyance expansion, other;
- Additional 25% for design, permitting, and project administration; and,
- Additional 10% for environmental and construction management.

Table 4-18. Warner Ranch Wellfield improvements for yields considered in association with the Lake Henshaw bypass method.

		Temporary Bypass		Permanent Bypass				
Nominal pipe size (in.)		42”	48”	42”	48”			54”
ID (in.)		36”	42”	36”	42”			48”
Target wellfield production (AFY)	Existing: ~7,500	4,760	7,140	9,125[1]	7,140	9,125[1]	9,125	9,125[1]
Production period (months)	12	4	4	12	4	12	4	12
No. of groundwater wells required	0[2]	11[3]	17[4]	7	17[4]	7	21	7
Estimated cost ($ million)	--	$20M	$31M	$13M	$31M	$13M	$39M	$13M
Cost range ($ million)		Minimum : $20M Average : $26M Maximum : $31M		Minimum : $13M Average : $26M Maximum : $39M				

[1] The hydraulics of this scenario for the given pipe size has not been confirmed.
[2] Note that the lack of a need for wellfield improvements is theoretical for the temporary bypass installation option since the current wells have been in operation for over 40 years. Redeveloping a new wellfield will be required for its continued, long-term use regardless of bypass installation or implementation of other HABs prevention and mitigation methods.
[3] The same number of wells would be needed to produce 9,125 AF over 8 months in a 42” pipeline.
[4] The same number of wells would be needed to produce 9,125 AF over 8 months in a 48” pipeline.

The District’s ongoing *Vista Flume Replacement Alignment Study* (FRAS) includes refinement of these planning-level calculations. Future design phases should further evaluate the wellfield improvements needed, develop a Class 5 cost estimate, and perform a cost-benefit analysis to determine the best wellfield improvement project for the District’s needs.

4.3.1.6 Bypass summary

A bypass pipeline allows for assured delivery of pumped groundwater without exposure to delays due to HABs in Lake Henshaw. A bypass pipeline does not affect the overall yield of the Local Water System except to the extent that by avoiding potential delays in release from the lake due to HABs, it avoids the evaporative losses associated with those delays. The 2018 Todd Groundwater Modeling Study noted that by increasing the wellfield capacity to deliver the firm yield (9,125 AF) in a four month period (nominally a 40 cfs production rate), average lake surface area could be reduced significantly, with an average savings in evaporative loss approaching 2,750 AF per year. By designing a bypass pipeline to deliver this 40 cfs of flow, the hypothetical evaporative loss savings would not be lessened by delays (and associated evaporative losses) due to HABs.

Nevertheless, the total yield of the Local Water System depends on the capture and release of stormwater runoff from Lake Henshaw in addition to wellfield production. A bypass pipeline alone would not provide total local water production consistent with historic averages; a successful in-lake HABs mitigation strategy is required to achieve historical local water yields. A reduced local yield is of concern when determining the economic feasibility of the District’s other infrastructure replacements, such as the Vista Flume replacement. If an in-lake HABs mitigation strategy is sufficiently successful to prevent delays in delivery due to HABs, then a bypass pipeline provides little benefit. Hence it is recommended that a phased approach be adopted to try

in-lake treatment methods first. The level of success achieved by these in-lake methods will inform the need for subsequent evaluation of a bypass pipeline.
4.3.1.4

Table 4-19. Summary of bypass pipeline options considered for long-term mitigation of HABs in Lake Henshaw.

	Temporary	Permanent
Life span	5 to 10 years maximum	50+ years
Nominal pipe size (in.)	42" (36" ID) 48" (42" ID)	42" (36" ID) 48" (42" ID) 54" (48" ID)
Flow capacity (AF)	20 cfs (4,760 AF/120 days) 30 cfs (7,140 AF/120 days)	20 cfs (4,760 AF/120 days) 30 cfs (7,140 AF/120 days) 40 cfs (9,125 AF/120 days)
Wellfield improvements needed?	No	Yes – to produce 9,125 AFY yield
Pipe material	HDPE – PE4710 DR 17	Steel DIP Concrete HDPE (buried only) Others TBD
Installation options	Above-grade Tunneling [1]	Above-grade Buried Partially buried Tunneling[1]
Capital cost (-50% to +100%) ($ million)	42": $22M 48": $25M	Est. $34M–$43M
O&M cost	$220,000/year [2]	TBD

Notes:
1 Tunneling is assumed at the upper San Luis Rey River creek crossing. It may be determined in the future that the creek crossing can be constructed with open-cut methods.
2 Annual average over the 5 to 10-year service life. Initial annual O&M cost will be less in the first 5 years of operation. Use of District staff for O&M activities may reduce the annual cost.

4.3.1.7 Permit requirements

As lead agency, the District would be required to undertake a project-appropriate CEQA compliance process for the bypass pipeline implementation, and, unless the District is categorically exempt, the project would also require a San Diego County building permit. Due to the pipeline creek crossing at the San Luis Rey River (Figure 4-9 and Figure 4-10), a lake and streambed alteration agreement (LSAA) would be required for pipeline installation under California Fish and Game Code section 1602, regardless of whether tunneling or open-cut construction methods are ultimately used. A Clean Water Act Section 404 permit would be required if open-cut methods are used, or if the pipeline alignment were to be constructed below the ordinary high operating water level of the lake.

Preliminary queries of California Department of Fish and Wildlife's (CDFW) California Natural Diversity Database (CNDDB) and USFWS Information for Planning and Conservation (IpaC) portal indicate that bypass pipeline construction activities could occur in potential habitat for state- and federally-listed species including, but not limited to: arroyo toad (*Bufo californicus*), western spadefoot (*Spea hammondii*), Stephen's kangaroo rat (*Dipodomys stephensi*), southcoast garter snake (*Thamnophis sirtalis*), tricolored blackbird (*Agelaius tricolor*), least Bell's vireo (*Vireo bellii pusillus*), and southwestern willow flycatcher (*Empidonax trailii extimus*), such that detailed habitat assessments may be required for one or more of these species as part of CEQA compliance (CNDDB 2022; USFWS 2022). Additional review of wildlife permitting considerations associated with future phases of the project may need to include other special-status species designations, such as CDFW Species of Special Concern or State Fully Protect species. The construction footprint of the bypass pipeline alignment may overlap with designated critical habitat for arroyo toad in the West Fork San Luis Rey River and is near designated critical habitat for southwestern willow flycatcher in the San Luis Rey River downstream of Lake Henshaw. Future phases of the project will need to address whether critical habitat may be adversely modified or destroyed by project implementation in coordination with USFWS. Additionally, the potential for effects to existing riparian habitat near the confluence of the San Luis Rey River and Lake Henshaw from reductions in groundwater flows may need to be considered under CEQA.

The District would need to interface with the California Division of Safety of Dams (DSOD) given that the bypass pipeline would terminate at the top of the Henshaw Dam spillway and would convey flow over the spillway, albeit without producing an impedance to flow during spill events.

Lastly, if the bypass pipeline implementation were to make use of federal funds (e.g., Infrastructure Investment and Jobs Act), then NEPA compliance would also be required.

4.3.1.8 Additional information needs

Additional information needed to further rank and prioritize the long-term bypass pipeline method, either as a temporary or permanent installation, for mitigating the effects of HABs on downstream water supply includes the following:

1. Further evaluation and development of Class 5 cost estimate for Warner Ranch Wellfield improvements required to produce the maximum sustainable groundwater yield of 9,125 AFY.
2. Evaluation and development of Class 5 cost estimate for a permanent bypass pipeline, including the assessment of using two smaller, parallel pipes and buried installation methods for potential cost savings.
3. Assessment of the ecological viability of Lake Henshaw at the minimum pool associated with substantially less or no groundwater delivered each year if groundwater is bypassed around the lake, as well as an assessment of potential impacts to recreational uses of the lake including contact and noncontact recreation (e.g., swimming, boating, fishing).
4. Quantification of the anticipated reduced evaporative losses and any associated cost offsets at the bypass minimum pool.
5. Further considerations on the combination of out-of-lake and in-lake solutions for Lake Henshaw, including refinement and/or confirmation of the impacts of each long-term method, or a combination of, to the local yield supply for the District.

4.3.2 In-lake method 10 – Algaecide Treatment

Algaecides were selected by the Project Team as a feasible long-term HABs control method for Lake Henshaw for the following reasons (see also Table 4-1):

- Algaecide application is a well-proven mitigation method for HABs. Approved algaecide chemicals act quickly (i.e., minutes to hours) and can prevent the formation of and interrupt an ongoing HAB and to stop cyanotoxin production. Some active ingredients can also destroy cyanotoxins in the water column (e.g., hydrogen peroxide).
- Little to no capital investment is required for algaecide application, since licensed applicators can be hired by the District to apply the chemicals and undertake monitoring needed to meet permit requirements.
- Costs are generally predictable and there are multiple algaecide products available on the market.
- In June 2021, the District obtained a Statewide Aquatic Weed Control Permit for application of copper sulfate, chelated copper, and sodium carbonate peroxyhydrate (peroxide) to control algae/cyanobacteria in Lake Henshaw.

Algaecide application goals and objectives and other operations considerations for long-term use in Lake Henshaw are the same as those described in Section 3.2.1 for short-term use. The District desires to obtain experience with the use of both copper- and peroxide-based algaecides in the lake over time. The District is currently gaining experience with the use of algaecides in Lake Henshaw and lessons learned will be applicable for the long term. While algaecide treatment remains on the list of long-term methods for mitigation of HABs for the reasons listed above, the success of one or more prevention methods would reduce the need for algaecide treatment in the long term.

4.4 Lake Wohlford Selected Prevention Methods

4.4.1 In-lake method 11 – Oxygenation via Speece Cone or SDOX

Escondido intends to continue operation of the current aeration-mixing system in Lake Wohlford until the new dam is in place, at which point an expansion of the current system or replacement of the current system with a more efficient oxygenation system, such as Speece Cone or SDOX, would be considered. The discussion below provides additional details to inform their future considerations regarding lake oxygenation.

4.4.1.1 Goals and capabilities

The goals, capabilities, and general approach for lake oxygenation are described in Section 4.2.3.1.

Lake Wohlford is a shallow, long lake with a current full pool elevation of 1,445 ft. Water depths currently range from 10 to 35 ft, with typical operating maximum depths at 50–55 ft, but these could increase to 80 ft if the full pool elevation increases to 1,480 ft as planned in the future after the Wohlford Dam raise is completed. The lake currently uses aeration at a single point about 1,200 ft upstream of the dam and Solar Bees for epilimnetic mixing in five locations along the length of the lake.

Figure 4-12 illustrates a one-year time series of water temperature and DO vertical profiles from the deepest portion of the lake just upstream of the dam. The top tile in Figure 4-12 indicates that thermal stratification can occur in late winter/spring (e.g., February to April 2017), but water temperatures are generally uniform throughout the water column in summer through early winter (e.g., May to December 2017). The bottom tile in Figure 4-12 indicates purple-shaded periods of hypolimnetic anoxia (DO at or near 0 mg/L) in March and April 2017, and full water column anoxia in May, August–September, and November 2017, which is characteristic of the longer period of record for water temperature and DO in Lake Wohlford (2016–2020; Figure 2-46a,b Figure 2-47a,b).

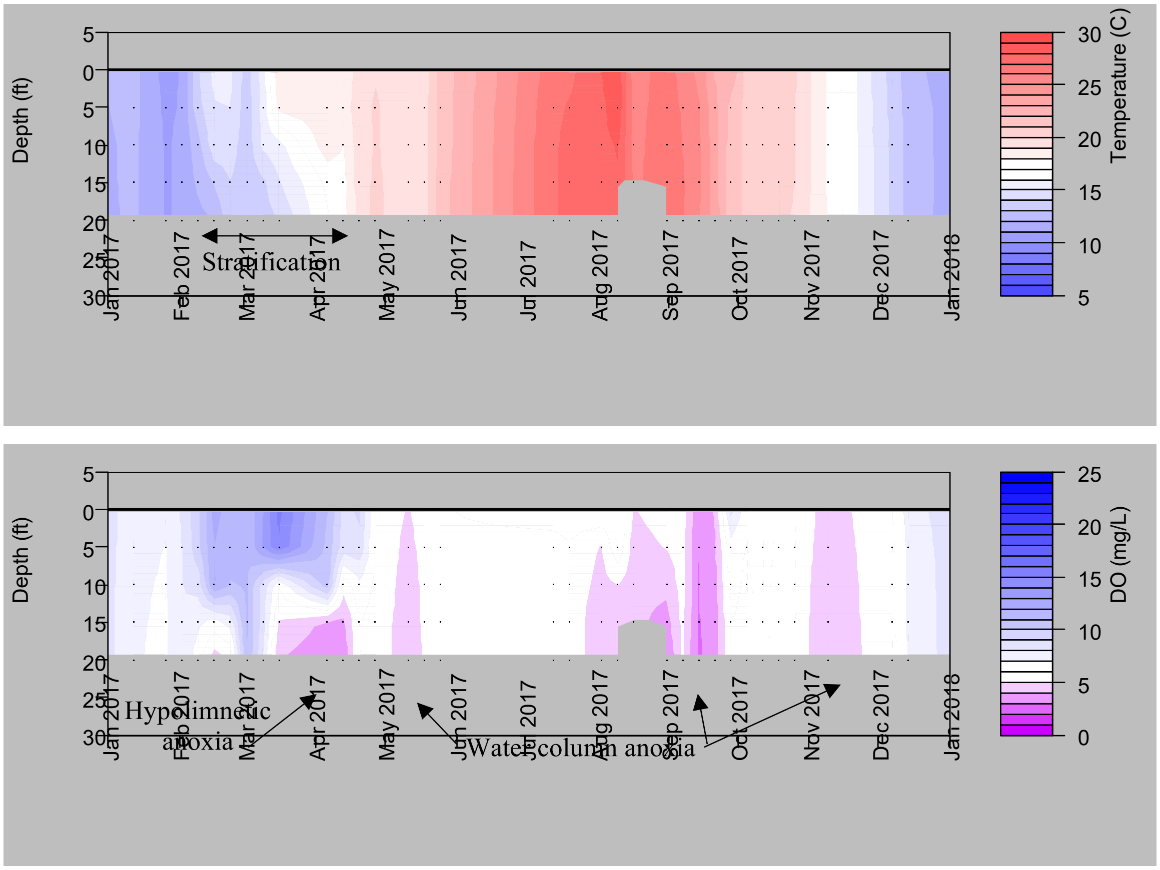


Figure 4-12. Water temperature and dissolved oxygen (DO) vertical profiles in Lake Wohlford in 2017.

The existing thermal and DO profiles (Figure 2-46a,b Figure 2-47a,b, Figure 4-12) indicate that the current aeration system is not able to prevent anoxic conditions in Lake Wohlford throughout the year. A new oxygenation system using Speece Cone or SDOX could dramatically increase the amount of oxygen input to the lake and could reduce or resolve anoxic conditions near the intake. A new oxygenation system sized for the larger, deeper reservoir, once Wohlford Dam is raised, would be an ideal application of hypolimnetic oxygenation, where these systems have been proven to reduce internal nutrient loading from bottom sediments and reduce algae blooms.

Speece and SDOX oxygenation systems differ from aeration because they use high purity oxygen (99% oxygen) and pressure to create a super-saturated oxygen rich water stream in cool water that stays near the lake bottom. This approach is not possible with aeration systems because the latter use a compressor to inject air, which adds gas that is 78% nitrogen and only 21% oxygen into the lake. In addition, aeration systems are relatively inefficient at dissolving oxygen into the water because the dissolution reaction is controlled by surface area between the gas and water and the partial pressure driving the gas from the bubble into the liquid. Injecting a plume of gas bubbles into the water column creates a relatively small surface area for gas dissolution and does not provide efficient oxygen transfer into the water. It is possible that over half of the oxygen in compressed air commonly used in aeration systems is released back to the atmosphere at the water surface. While vertical mixing cells are often created by aeration systems, where surface waters that are super-saturated with photosynthetically produced DO are transported downward to mid-depth and deeper waters, the oxygen demand of the algae that are producing high DO in surface waters during the day can often overwhelm the aeration system at night and even in deeper, shaded waters during the day. In contrast, the SDOX system typically transfers 98% of the injected oxygen directly into the water column, and neither Speece nor SDOX systems rely upon DO produced by algae to be mixed downward into the water column.

A description of Speece Cone and SDOX oxygenation systems can be found in Section 4.2.3.1 along with schematics presented in Figure 4-3. and Figure 4-4.

4.4.1.2 Implementation considerations

As with Lake Henshaw, an oxygenation system at Lake Wohlford would require onshore space for the LOX system, road access for LOX tanker truck deliveries, and power for the pumps and controls.

For Lake Wohlford, the Speece Cone could be placed on the lake bottom in the restricted area near the dam or onshore near the LOX system. For the SDOX approach, the system would be located onshore with the LOX system. Figure 4-13 presents the preliminary in-lake location of the Speece Cone and diffuser along with the general bathymetry of the lake. The lightest blue area represents the future full pool elevation once the Wohlford Dam is raised, while the medium blue represents current full pool. The darkest blue represents the deepest area of the lake which is generally located in the restricted area just upstream of the existing dam.

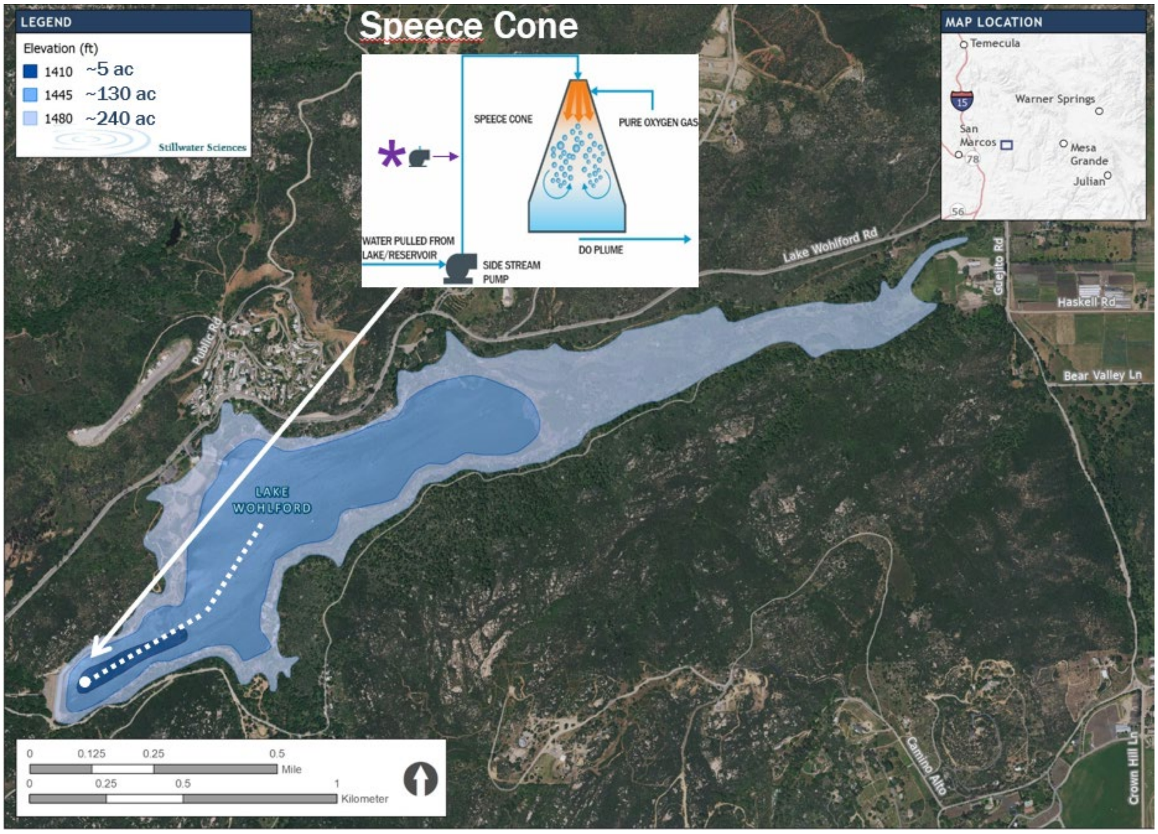


Figure 4-13. Preliminary locations for Speece Cone or SDOX oxygenation systems in Lake Wohlford.

At either Lake Henshaw or Lake Wohlford, it is feasible to use oxygen generators instead of LOX to provide the oxygen feed gas for the Speece Cone or SDOX. Oxygen generators use compressed air as a feed gas and then sieve out the nitrogen which is released back into the atmosphere. The resulting gas is typically 92 to 95% oxygen. Oxygen generators require more maintenance than LOX systems and require power to run the air compressors and pressure/vacuum system involved with generating oxygen. Therefore, oxygen generation systems incur power and maintenance costs instead of the purchase cost of LOX. A cost benefit analysis can be conducted during preliminary design to compare the capital and O&M costs of a LOX system versus an oxygen generation system. The analysis should include non-cost factors regarding the security of the oxygen system needed for each approach. Figure 4-14 shows an example oxygenation system using oxygen generators instead of LOX to supply oxygen to onshore Speece Cones. This river installation in Savannah, Georgia is much larger (14 tons/day) than what is recommended for Lake Wohlford (1.5 to 3.5 tons/day). Also, power costs are substantially lower in Georgia than California.

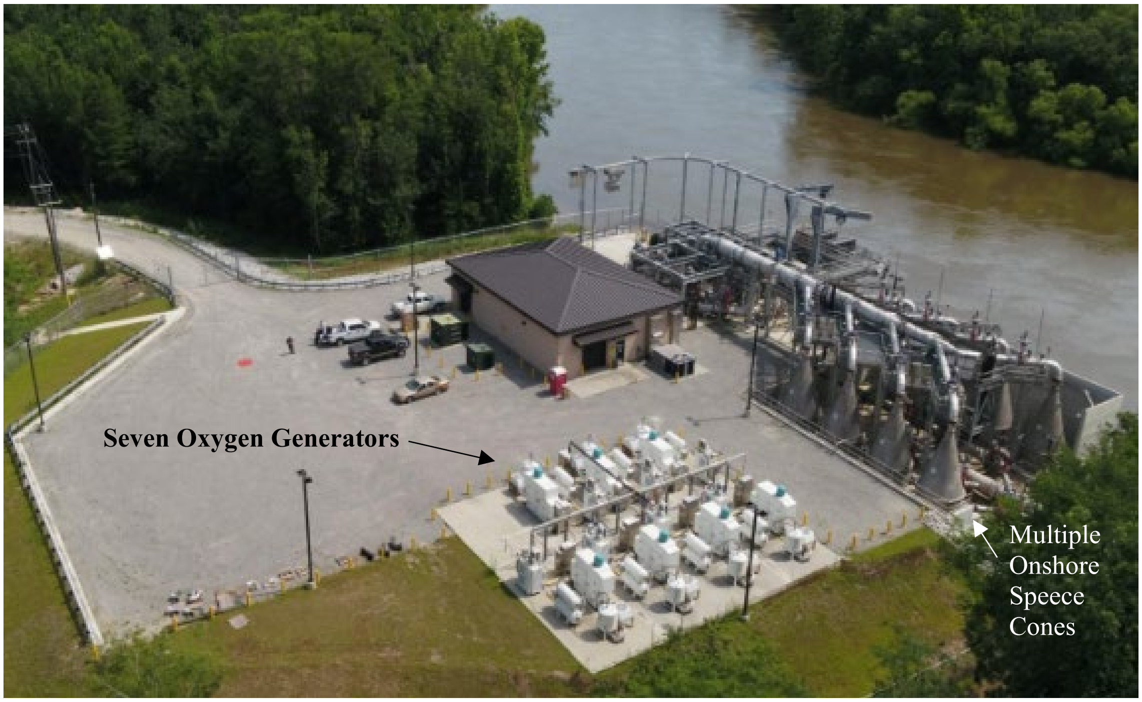


Figure 4-14. Example oxygenation system with onshore Speece Cones and oxygen generators. Photo Reprinted from US Army Corps of Engineers *Startup Run Data Collection & Modeling Report*, Oxygen Injection System Environmental Testing for the Savannah Harbor Expansion Project, March 2021.

Safety and security of the onshore and in lake oxygenation system are critical. Onshore LOX or oxygen generator systems are typically fenced and gated as shown in Figure 4-14. Privacy fencing can also be used to block public view and direct line of sight to equipment. LOX tanks can be selected in the horizontal configuration, which require a larger footprint than vertical tanks but are more easily hidden behind walls or privacy fencing.

Additional chemical treatment

With either the Speece Cone or SDOX approach, an optional alum injection system can be used to enhance orthophosphate removal from the water column by binding and precipitating orthophosphate with aluminum sulfate (alum). The solid precipitate would then deposit in lake bottom sediments. Alum would be added to the oxygenated stream either within the Speece Cone or just downstream of the SDOX vessel. An alum chemical storage tank and small chemical metering pump can be located onshore near the LOX system.

4.4.1.3 Anticipated implementation schedule

The design, permitting, and construction of the oxygenation system for Lake Wohlford is anticipated to require 2-3 years total, with the construction phase lasting approximately 1 year of the total. The construction schedule would need to be refined in a detailed design phase, with special consideration given to any permitting requirements and associated timeframes.

4.4.1.4 Compatibility

Oxygenation via Speece Cone or SDOX in Lake Wohlford is compatible with Escondido's water supply objectives as it would improve the quality of raw water going into the EVWTP by suppressing anoxia in bottom waters and sediments, resulting in decreased nutrient cycling and cyanobacteria growth. Water quality improvements associated with successful oxygenation operations would also be compatible with recreation activities around and on the lake. Oxygenation should improve habitat for the fishery at Lake Wohlford, particularly by expanding the volume of cool oxygenated water available to rainbow trout during warmer months of the year, although data are needed to confirm seasonal DO patterns in lake bottom waters (Section 2.2.2.1). Visual and aesthetic characteristics would improve since reduced nutrient cycling would control cyanobacteria blooms, reducing the prevalence of algal scum accumulation.

4.4.1.5 Cost estimate

Capital costs

A Class 5 cost estimate for oxygenation via Speece Cone or SDOX was assembled in accordance with AACE criteria. The accuracy of Class 5 estimates ranges from -50 to +100 percent; therefore, the capital cost estimate applies to either the Speece Cone or the SDOX oxygenation approach. As described in Section 4.2.3.5, the cost estimate for the Lake Wohlford oxygenation system was scaled from a recent 5 ton/day oxygenation project located in San Pablo Reservoir in northern California, where construction began in January 2022. The Lake Wohlford equipment and installation costs were extracted from the San Pablo Reservoir construction schedule of values, summed, and then scaled for the current and anticipated future size of Lake Wohlford. Engineering and administrative costs were added as a percentage of construction.

For Lake Wohlford, the system size was estimated for both the existing and anticipated future (i.e., raised dam) full pool conditions. For the existing condition, the estimated system size is 1.5 tons/day based on oxygenating the sediment and 30% of the current full pool water depth (35 ft) over the entire area of the reservoir (approximately 127 acres). For the future raised dam condition, the estimated system size is 3.5 tons/day based on oxygenating the sediment and 30% of the future full pool water depth (70 ft) over the entire area of the reservoir (approximately 236 acres). For both conditions, the water column oxygen demand was assumed as 0.2 mg/L/day and the sediment oxygen demand was assumed as 2 g/m^2/day. The Lake Wohlford preliminary design criteria should be refined in a future phase of the project using empirically determined oxygen demand of the water column and bottom sediments, when those data become available. Table 4-20 presents the San Pablo equipment and installation cost, the scaling factor for each condition, and the percentages included in the Lake Wohlford estimate for contingency allowance (40%), engineering costs (15%), and various legal/administrative costs typically associated with construction projects.

Table 4-20. Estimated capital costs for a Lake Wohlford oxygenation system via ECO_2 Speece Cone or SDOX O2® (Class 5).

Parameter	Units	San Pablo	Wohlford (existing full pool)	Wohlford (future full pool)
Design Capacity	tons/day	5	1.5	3.5
Scale factor	-	-	0.45	0.78
Capital Cost, 1/2022	$	$ 7,110,000	$ 3,180,000	$ 5,560,000
Engineering, ESDC	15.0%	-	$ 477,000	$ 834,000
Contractor GC	15.0%	-	$ 549,000	$ 959,000
Start-Up and Training	4.0%	-	$ 168,000	$ 294,000
Undesign/Undevelop Contingency	40.0%	-	$ 1,750,000	$ 3,059,000
Building Risk, Liability Auto Ins	2.0%	-	$ 122,000	$ 214,000
Payment/Performance Bonds	1.5%	-	$ 94,000	$ 164,000
Construction Cost, 1/2022	-	-	$ 6,340,000	$ 11,084,000

For the existing Lake Wohlford, the estimated capital cost is $6.3M in current dollars based on a system sized to deliver 1.5 tons per day of oxygen using LOX delivered by tanker truck as the oxygen source. For the future larger and deeper Lake Wohlford, the estimated capital cost is $11.1M in current dollars based on a 3.5 tons per day oxygenation system using LOX.

O&M costs

Estimated O&M costs for an oxygenation system consist of LOX purchase/delivery costs, power cost for pumping, and labor/general repair costs. Assuming the system operates year-round, the total annual O&M cost for the 1.5 tons/day system is estimated to be approximately $185,000 per year based on the following:

- $70,000/year for LOX (1.5 tons/day at $125/ton of LOX delivered)
- $52,000/year for power ($0.15/kWh to run a 50 hp pump motor)
- $56,000/year for labor and general repair (assuming 1% of capital cost)

For the future larger Lake Wohlford assuming year-round operation, the total annual O&M cost for the 3.5 tons/day system is estimated to be approximately $362,000 per year based on the following:

- $158,000/year for LOX (3.5 tons/day at $125/ton of LOX delivered)
- $93,000/year for power ($0.15/kWh to run a 90 hp pump motor)
- $111,000/year for labor and general repair (assuming 1% of capital cost)

4.4.1.6 Permit requirements

As lead agency, Escondido would be required to undertake a project-appropriate CEQA compliance process, and, unless Escondido is categorically exempt, the project would also require a San Diego County building permit. By siting new facilities judiciously, Escondido may not need to interface with the California Division of Safety of Dams (DSOD).

4.4.1.7 Additional information needs

The following additional information characterizing year-round water quality is needed to accurately size and predict the effectiveness of an oxygenation system for HABs prevention in Lake Wohlford, particularly once the Wohlford Dam has been raised.

1. **To refine system size, the oxygen demand of the water column as well as the sediment need to be determined.** Additional vertical profiles of temperature and DO in other areas of the lake throughout the year would be helpful, particularly once the dam is raised and the lake becomes deeper.
2. **To design and locate an effective system, data needs to be collected characterizing the spatial extent of algal blooms to understand where they form and how they spread.** Hypolimnetic oxygenation systems in deep lakes have been proven to mitigate nutrient loading from sediments and reduce algae blooms. Once the dam is raised, locations where algal blooms occur may change and would be critical information in deciding where to apply oxygen. Also, modeling the proposed oxygen application point to see the spatial extent of the plume within the current lake, should oxygenation be pursued prior to dam raising, and the future larger, deeper, and longer lake, following dam raising, would help inform design.
3. **A detailed cost/benefit analysis for both Speece Cone and SDOX approaches should be undertaken.** If the oxygenation alternative is selected, this comparison and cost analysis would guide whether to select one approach before moving forward to final design or to compete them against each other.
4. **A full scale 9-to-12-month operational study would provide exceptionally valuable information on oxygenation effectiveness.** This approach is discussed for Lake Henshaw in Section 4.2.3.8. For Lake Wohlford, it may be possible to utilize some of the piping already installed in the lake for the existing aeration system by making some modifications. Total costs for the study are estimated to range from $400,000 to $600,000 depending on the amount of oxygen used and study length. Conducting a full-scale operational study would also be a good way to evaluate whether alum addition is needed to reduce internal loading of phosphorus.

4.5 Lake Wohlford Selected Mitigation Methods

4.5.1 In-lake method 7 – Selective Withdrawal

4.5.1.1 Goals and capabilities

The goal of selective withdrawal is to allow Escondido operations staff to withdraw water of consistent quality from within the reservoir through the use of multiple outlets located at different water column depths. Selective withdrawal would serve as HABs mitigation since this method would not prevent a cyanobacteria bloom but would instead provide operational flexibility regarding which layer(s) of water Escondido could withdraw from the reservoir if a bloom should occur (e.g., avoidance of surface waters where cyanotoxins may be elevated). Currently, withdrawals (drafts) from the reservoir occur from the existing inlet/outlet tower, an approximately 85-ft-high reinforced concrete structure located on the left (south) abutment at the upstream toe of the dam (Black and Veatch 2013). The inlet/outlet tower was constructed in 1923–1924 and provides raw water to the EVWTP. The inlet/outlet tower has two operable 36-in. intake gates, one at 1,449 ft elevation and the other at 1,424 ft. A third, low-level intake gate at 1,398 ft is blocked by sediment accumulation and was plugged and abandoned in 2003. The upper intake gate at 1,449 ft is above the currently allowable water level for the lake (i.e., 1,440 ft). Thus, the Lake Wohlford inlet/outlet tower currently has a single operating gate at 1,424 ft, which is approximately 16 ft below the water surface, or in the top third of the generally isothermal water column (e.g., Figure 2-46a,b; Figure 4-12) for typical operating elevations (50–55 ft).

Escondido evaluated three alternatives for an outlet tower once the new dam is in place, including rehabilitation of the existing outlet tower, construction of a new free-standing outlet tower, and construction of a new outlet tower on the face of the new dam. The preferred alternative is a new outlet tower on the upstream face of the new dam. The tower would extend to the dam crest at elevation 1,490 ft, would have four intake gates, and would utilize gate operators at the top of the tower where they would not be affected by flood flow water surface elevations (Black and Veatch 2013).

Construction of a new outlet tower on the upstream face of the new dam would require excavation of a notch in the existing dam (which will remain partially in place) to facilitate flow from the reservoir to the face of the dam. While the top portion of the existing dam is being removed down to approximately 1,450 ft for all alternatives, the preferred outlet tower alternative would include additional excavation down to 1,420 ft to allow unimpeded flow to both lower gates from the reservoir (Black and Veatch 2013).

4.5.1.2 Compatibility with current drinking water reservoir management objectives

Selective withdrawal options for Lake Wohlford in combination with improved water quality via oxygenation (Section 4.4.1) is highly compatible with Escondido's drinking water supply objectives and would maximize operational flexibility to allow consistent delivery of high-quality water to its customers. The new outlet tower configuration would add intake gates, which would enhance operational flexibility over current conditions and would have no impact on Lake Wohlford's recreational use or aesthetics around the reservoir since the infrastructure would be located at the dam face and mostly under water. The new tower system would operate indefinitely into the future. If properly designed and constructed, the risk of failure would be *de minimis*.

4.5.1.3 Estimated costs

The order of magnitude (AACEI Class 5) estimated capital costs for the preferred alternative new outlet tower on Wohlford Dam are $1.8M to $2.6M (Black and Veatch 2013). Note that while the estimated costs include mobilization, bonds, insurance, general construction items and confidence intervals (which also reflect contingency), they were developed to inform outlet tower alternative selection and were intended to be used for comparative purposes only (Black and Veatch 2013). Operation and maintenance costs are not included, although these would be generally minimal and may include subaqueous remotely operated vehicle (ROV) inspection of the structure and removal of any accumulated debris from interfering in gate operations every few years.

4.5.1.4 Anticipated construction/implementation schedules

Construction of construction of a new outlet tower on the face of the new dam would require approximately seven months. Compared with other alternatives considered, the new outlet tower located on the face of the new dam could add one or two months to the overall schedule for the dam replacement project for Lake Wohlford to allow for excavation of the notch on the existing dam. The excavation cannot occur until completion of the new dam (Black and Veatch 2013).

4.5.1.5 Likely permit requirements

As the new outlet tower would be part of the Lake Wohlford dam replacement project, any permit requirements for the tower would be part of permitting activities for the larger project (e.g., CEQA, DSOD). However, the slopes and geometry of the notch to be cut in the existing dam to support full functionality of the new outlet tower will need to be clarified with DSOD and may require a slope stability analysis (Black and Veatch 2013).

4.5.1.6 Additional information needed to further rank and prioritize method

If a selective withdrawal system were to be used independent of oxygenation in Lake Wohlford, then the below additional information would be needed to accurately predict the effectiveness of selective withdrawal for HABs mitigation in Lake Wohlford once the Wohlford Dam has been raised. If oxygenation is successfully implemented in Lake Wohlford, then HABs should be minimized or eliminated and location of the gates relative to the vertical extent of a HAB in the water column would be less critical.

1. **To assess where the intake gates would be located relative to the seasonal thermocline,** more detailed information on the design elevations for the four intake gates included in the new outlet tower is needed along with numerical modeling of water temperature and DO with depth once the dam is raised and the area near the dam becomes deeper.

4.5.2 In-lake method 10 – Algaecide Treatment

Algaecides were selected by the Project Team as a feasible long-term HABs control method for Lake Wohlford for the following reasons (see also Table 4-2):

- Algaecide application is a well-proven mitigation method for HABs. Approved algaecide chemicals act quickly (i.e., minutes to hours) and can prevent the formation of and interrupt an ongoing HAB and to stop cyanotoxin production. Some active ingredients can also destroy cyanotoxins in the water column (e.g., hydrogen peroxide).
- Little to no capital investment is required for algaecide application, since licensed applicators can be hired by the Escondido to apply the chemicals and undertake monitoring needed to meet permit requirements.
- Costs are generally predictable and there are multiple algaecide products available on the market.
- For the past several years, Escondido has been successfully applying a chelated copper algaecide to control algae/cyanobacteria in Lake Wohlford.

While algaecide treatment remains on the list of long-term methods for mitigation of HABs for the reasons listed above, the success of the oxygenation prevention method and/or the selective withdrawal mitigation method would reduce the need for algaecide treatment in the long term.

5 WATER QUALITY MONITORING PLAN

5.1 Monitoring Approach

Monitoring is a necessary first step for determining whether project goals and objectives were achieved. As there are a variety of reasons why a project may not meet the originally conceived goals and/or objectives, successful monitoring programs can include multiple types of monitoring. As shown in Table 5-1, each monitoring type focuses on a different aspect of the project and necessarily involves a specific time frame for monitoring activities. Spatial scales for each type of monitoring depend on the particular objective or hypothesis being addressed.

Table 5-1. Monitoring types for compliance, status and trends, and the assessment of post-project success.

Type of Monitoring	Question Addressed	Time Frame
Compliance	Does the project meet the terms of required permits?	Variable, depends on permit terms
Status and trends		
Synoptic	What are the characteristics of the project area as defined by a set of key variables at a particular moment in time?	Medium term
Routine	What is the status and what are the trends for a set of key variables at regular time intervals?	Long term
Rapid response	Are key variables across the project area crossing pre-determined thresholds that then trigger a management action at a particular moment in time?	Short term
Post-project success		
Implementation	Was the project implemented as planned?	Short term
Effectiveness	Was the project effective at meeting water quality improvement objectives?	Medium to long term
Validation	Are the basic assumptions behind the project conceptual model valid?	Medium to long term

For this project, compliance monitoring is aligned with the short- and long-term (as needed) algaecide application mitigation method and the required statewide aquatic weed control permit, which is discussed in Section 5.4.1 for the Warner Ranch Wellfield, Section 5.5.2.3 for Lake Henshaw, and Section 5.7.2.3 for Lake Wohlford. Further, Appendix C presents the Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch. Compliance monitoring may also be required for long-term phosphorus inactivation methods in Lake Henshaw source waters (Section 4.2.1) and in-lake waters (Section 4.2.2) if alum and/or Phoslock™ treatments are used.

Synoptic monitoring in lakes Henshaw and Wohlford focuses on the establishment of a relatively small number of routine monitoring *index sites* in order to provide data that are representative of other locations of interest, reducing the total number of samples that must be collected on a regular basis (e.g., weekly, monthly, quarterly) in order to characterize the broader study area. Synoptic monitoring is discussed in Section 5.5.1 for Lake Henshaw and Section 5.7.1 for Lake Wohlford.

Routine and rapid response monitoring are discussed for both Lake Henshaw (Section 5.5.2) and Lake Wohlford (Section 5.7.2) to provide data that triggers when the District and Escondido would move from an *operational strategies window*, before a HAB occurs and when multiple options for reservoir operation are still available, to an *early warning window*, when monitoring data suggest that a HAB may be developing, and, as needed, to a *treatment window*, when algaecide application would occur prior to a HAB becoming out of control. While routine monitoring may continue indefinitely, albeit as updated through adaptive management (Section 5.3), rapid response monitoring would only continue insofar as algaecide treatment remains a HABs mitigation method used by the District and/or Escondido.

With respect to the types of monitoring used to assess post-project success, implementation monitoring activities would be appropriate if the District and/or Escondido opt to implement one or more long-term prevention or mitigation methods that possess complex design features (e.g., oxygenation system, bypass pipeline, selective withdrawal outlet tower), where as-built conditions would need to be compared against planning specifications to determine whether the project was implemented as designed. Future iterations of this HABs Management and Mitigation Plan should be updated with relevant implementation monitoring elements, as needed.

For the short- and long-term, effectiveness monitoring in this HABs Management and Mitigation Plan focuses on whether algaecide application has the desired effect of reducing cyanotoxin concentrations below pre-determined thresholds in either or both lakes. Detailed discussion of algaecide effectiveness monitoring is presented in Section 5.5.2 for Lake Henshaw and Section 5.7.2 for Lake Wohlford. Future iterations of this HABs Management and Mitigation Plan should be updated with relevant effectiveness monitoring elements for any other long-term methods that are selected for implementation.

Validation monitoring tests the fundamental cause-effect linkages and mechanisms in the conceptual model (Figure 2-55) and will ultimately support long-term adaptive management of water quality in lakes Henshaw and Wohlford. Future iterations of this plan should be updated with validation monitoring elements that focus on unknowns in the conceptual model:

1. Whether infrequent runoff events are responsible for a substantial portion of nutrient loading to Lake Henshaw and/or Lake Wohlford;
2. Whether Lake Henshaw thermally and/or chemical stratifies such that localized regions of low DO occur in bottom waters and at the sediment/water interface that support high levels of internal loading of bioavailable nutrients; and,
3. The ultimate source of cyanotoxins (i.e., specific planktonic and/or benthic cyanobacteria species) in both Lake Henshaw and Lake Wohlford.

Continued studies to address the additional information needs identified for each of the potential long-term prevention and mitigation methods discussed in Section 4 may result in refinements to the above list.

5.2 Reservoir Management Windows of Opportunity and the Use of Operational Triggers for Short-term HABs Mitigation

The ability to track progression of a potentially cyanotoxin-causing bloom requires sampling early in HAB development, when water concentrations of cyanobacteria and cyanotoxins may still be relatively low. Because HABs can develop rapidly (i.e., over the course of several days), early warning that a HAB may be forming in Lake Henshaw and/or Lake Wohlford is critical for

successful application of algaecides as the current short-term mitigation strategy. Although licensed applicators can be hired to apply the chemicals and undertake monitoring needed to meet permit requirements, the contracting, selection and procurement of the appropriate chemical(s), transport of the chemical(s) to the site, and deployment of specialized equipment and personnel to treat a large lake, require lead time that typically extends two to four weeks. Thus, appropriate triggers are needed to allow the District and Escondido to move from an *operational strategies window*, before a HAB occurs and when multiple options for reservoir operation are still available, to an *early warning window*, when monitoring data suggest that a HAB may be developing, and, as needed, to a *treatment window*, when algaecide application would occur prior to a HAB becoming out of control (
Figure **5-1**).

For short-term HABs mitigation, this monitoring plan uses operational triggers to transition from routine (weekly) monitoring at a small number of index sites during the reservoir operational strategies window, to rapid response monitoring at a greater number of sites and at a higher frequency (sub-weekly) during the early warning window, and, as needed, to algaecide treatment. The triggers for these transitions involve specified increases in water quality parameters (including total and/or cyanobacteria cell counts, cyanotoxin concentrations, and/or other *in situ* water quality parameters) between consecutive samples collected at one or more monitoring sites, with greater relative increases required to move from the early warning window to the treatment window. The operational triggers also indicate whether it is too late to apply algaecides to Lake Henshaw or Lake Wohlford because the bloom is too dense to be effectively treated with an algaecide, and when to reduce monitoring efforts because the HAB has been eliminated. Additional details are presented in sections 5.5.2 and 5.7.2.

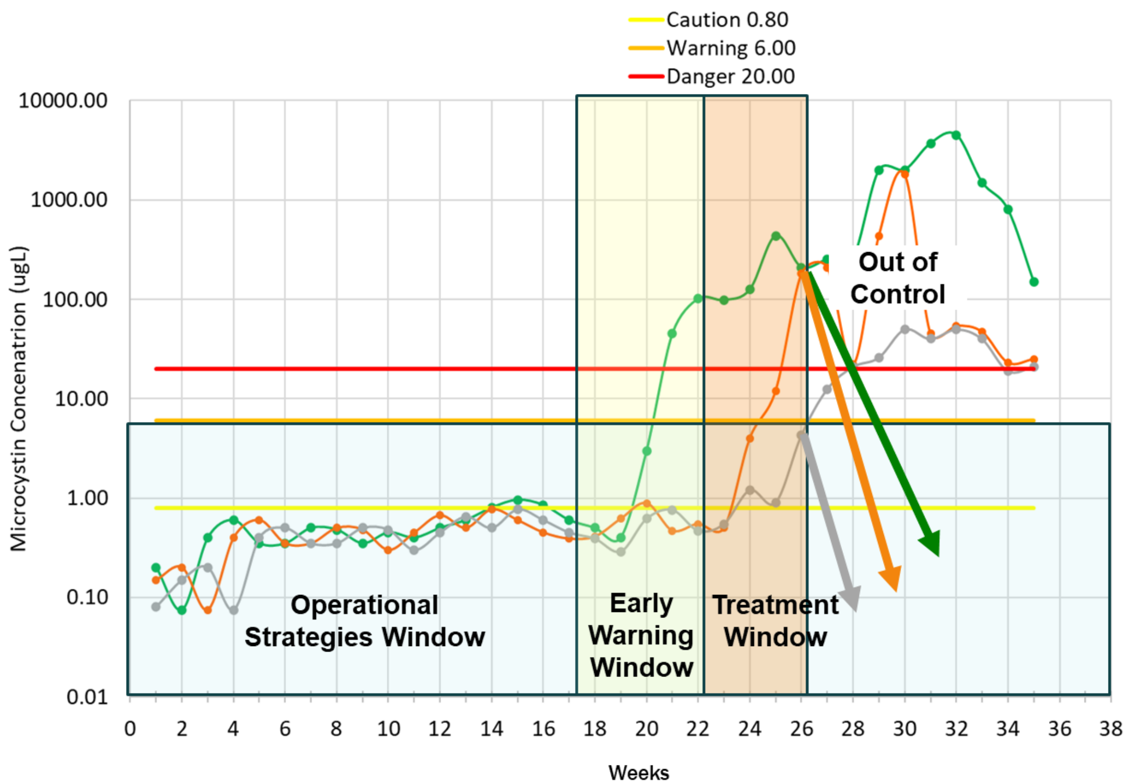


Figure 5-1. Windows of opportunity for HABs management with hypothetical microcystin response in Lake Henshaw and Lake Wohlford. Colored arrows indicate algaecide treatment with rapid decreases in microcystin concentrations compared with an out-of-control bloom condition. A similar type of relationship may apply for other cyanotoxins.

5.3 Importance of Adaptive Management

Effective long-term management of complex water resources, including the Local Water System (Figure 2-55), requires that the analysis and interpretation of water quality monitoring data occur periodically in order to support the scientific "learning while doing" inherent to adaptive management. Knowledge gained during previous monitoring periods should be used to evaluate whether monitoring goals and objectives are being met through the monitoring program and this evaluation may result in recommendations for revisions. While this water quality monitoring plan has been developed to assist the District and Escondido in the long-term management of the Local Water System using a detailed process of existing data compilation and analysis (Section 2), consideration of potential short-term HAB mitigation strategies for Lake Henshaw and the selection of algaecides as the most appropriate short-term strategy (Section 3), and screening of potential long-term HAB prevention strategies for Lake Henshaw and Lake Wohlford (Section 4), implementation of this plan will provide new information about the physical, chemical, and biological processes controlling HABs in the water system, and this new information may change future monitoring efforts. For example, data collected to characterize HAB development and the Lake Henshaw response to algaecide application may change the number of monitoring sites, the specific parameters being monitored, and/or the established operational triggers needed to effectively manage HABs in this waterbody and its receiving waters. Additionally, Escondido is currently controlling HABs in Lake Wohlford through routine monitoring and periodic algaecide application using established triggers (Section 5.7.2) and refinements to this approach may not be necessary until Lake Henshaw HAB conditions change and/or the Wohlford Dam is raised, creating a larger and deeper Lake Wohlford. Lastly, while multiple water quality monitoring techniques are included in this plan, open source cyanobacterial remote sensing data and new approaches for less expensive and/or less logistically complicated analyses of cyanotoxins are becoming more available with time. This water quality monitoring plan assumes that water quality monitoring data collected by the District and Escondido will be reviewed regularly to aid in adaptive decision making, and the data will be compiled, analyzed, and included in periodic reports. Consistent with an adaptive management approach, results and recommendations for plan revisions, as applicable and justified by knowledge gained during the previous monitoring period, will be included as part of future reporting for the Local Water System.

5.4 Warner Wellfield

5.4.1 Monitoring required under the Statewide Aquatic Weed Control Permit

Coverage under the California Statewide General National Pollutant Discharge Elimination System (NPDES) Permit for Residual Aquatic Pesticide Discharges to Waters of the United States from Algae and Aquatic Weed Control Applications Permit # CAG990005 requires that certain monitoring activities be conducted in association with aquatic herbicide/algaecide application to control nuisance aquatic macrophytes and/or filamentous algae in the ditch system. The *Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch* (Marine Biochemists 2021) includes an Aquatic Pesticide Monitoring and Reporting Program (APMRP) for the Warner Wellfield that is compliant with NPDES requirements.

The APMRP addresses the following two key questions:

1. Does the residual algaecides and aquatic herbicides discharge cause an exceedance of the receiving water limitations?
2. Does the discharge of residual algaecides and aquatic herbicides, including active ingredients, inert ingredients, and degradation byproducts, in any combination cause or contribute to an exceedance of the "no toxics in toxic amount" narrative toxicity objective?

Prior to, during, and following algaecide treatment, monitoring required for the APMRP will include the parameters listed in Table 5-7 (Section 5.5.2.3). The three types of required monitoring for each treatment event include background monitoring (BG) immediately upstream of the treatment area just prior to the treatment event (i.e., up to 24 hours in advance of treatment); event monitoring (EM) directly downstream of the treatment location, in flowing waters outside of the treatment location itself; and post-event monitoring (PE) at the terminus of the ditch system before the water is released in the San Luis Rey River leading to Lake Henshaw, within one week after the treatment event. One full set of three samples (i.e., BG, EM and PE samples) will be collected during each treatment event along with quality control samples. For products that contain sodium carbonate peroxyhydrate, no treatment event monitoring for residuals is required before or after treatment because peroxyhydrate breakdown products are water and bicarbonate. For application of all other algaecides and aquatic herbicides listed in the NPDES permit, monitoring will be conducted for each active ingredient utilized at the time of treatment event.

APMRP details regarding monitoring procedures for in-field and grab sampling, laboratory analyses, monitoring records, calibration and maintenance of instrumentation, maximum allowable copper concentrations in the Warner Wellfield following treatment (if copper is the active ingredient in the applied algaecide), and reporting are presented in Appendix C.

5.5 Lake Henshaw

5.5.1 Synoptic monitoring to establish index sites and rapid response monitoring sites

5.5.1.1 Background

Water quality is currently monitored in Lake Henshaw at up to three sites in the lake and one site at the lake outlet (Table 5-2, Figure 5-2). Grab samples are collected on a weekly basis for nutrient species (Section 2.2.2.3), cyanotoxins (Section 2.2.5), and cyanobacteria identification (PTOX) (Section 2.2.3). Variability in microcystin and anatoxin-a concentrations at the buoy line bottom and fishing dock sites indicates that the two open water samples at these locations typically are significantly indistinguishable (Figure 2-25). While the existing monitoring sites and frequency of monitoring are providing the District with an understanding of the seasonal variability in nutrients, cyanotoxins, and cyanobacteria species at locations in the deepest portion of Lake Henshaw nearest the reservoir outlet and the primary recreational access point along the southwestern shoreline, there is currently a lack of data characterizing the spatial variability of cyanotoxins and cyanobacteria throughout the reservoir. In order to effectively control a HAB using algaecides as a short-term mitigation strategy, it is critical to understand whether the HAB is developing at locations other than the sites currently being monitored and could be rapidly moved around the lake via wind and/or lake circulation.

For routine monitoring activities, the establishment of a relatively small number of *index sites* is intended to provide data that are representative of other locations of interest, reducing the total number of samples that must be collected on a regular basis (e.g., weekly, monthly, quarterly) in order to characterize the broader study area. For cyanobacteria and cyanotoxins in lakes and reservoirs, surface accumulations can occur in open water and shoreline locations during periods of high solar insolation and low wind, and they can occur along downwind shoreline locations when winds increase, or they can be relatively isolated in enclosed bays. Thus, HAB-related index sites should include both open water and shoreline locations, and preferably be in easily accessible locations to facilitate regular monitoring. Routine monitoring for HABs management in Lake Henshaw will occur as part of an *operational strategies window* (Figure 5-1) when multiple strategies for reservoir operation and water delivery are available to the District.

While longer sampling intervals are important for documenting chemical and biological changes in a lake or reservoir over time, they are insufficient as an early warning mechanism for rapid algal bloom development and associated cyanotoxin events that can develop over just a few days. When a HAB is developing, additional data are often necessary to characterize the spatial variability of cyanotoxins and cyanobacteria throughout the waterbody, to help inform the necessary extent of any short-term mitigation strategies such as algaecide treatment, and to provide a broader basis for assessing the success of long-term water quality improvements in the reservoir as a whole. In order for monitoring to provide the necessary early warning that a HAB may be forming, this type of monitoring often involves a greater number of *rapid response monitoring* sites and a higher frequency of monitoring than routine monitoring. Rapid response monitoring occurs during an *early warning window* (Figure 5-1) that is focused on the potential for elevated cyanotoxin concentrations to rapidly change reservoir operations and water delivery.

Table 5-2. Lake Henshaw water quality monitoring sites.

Site ID	Location	Latitude	Longitude
H-S	Southwestern shoreline at beach adjacent to fishing dock	33.23496°N	116.75617°W
H-FD	Southwestern shoreline at the in-water end of the fishing dock	33.23544°N	116.75568°W
H-BLS	Buoy line at dam in surface waters	33.23963°N	116.76174°W
H-BL	Buoy line at dam in bottom waters	33.23963°N	116.76174°W
H-R	Dam release channel approximately 10 ft upstream of flow measurement weir (point of release to San Luis Rey River)	33.23923°N	116.76594°W

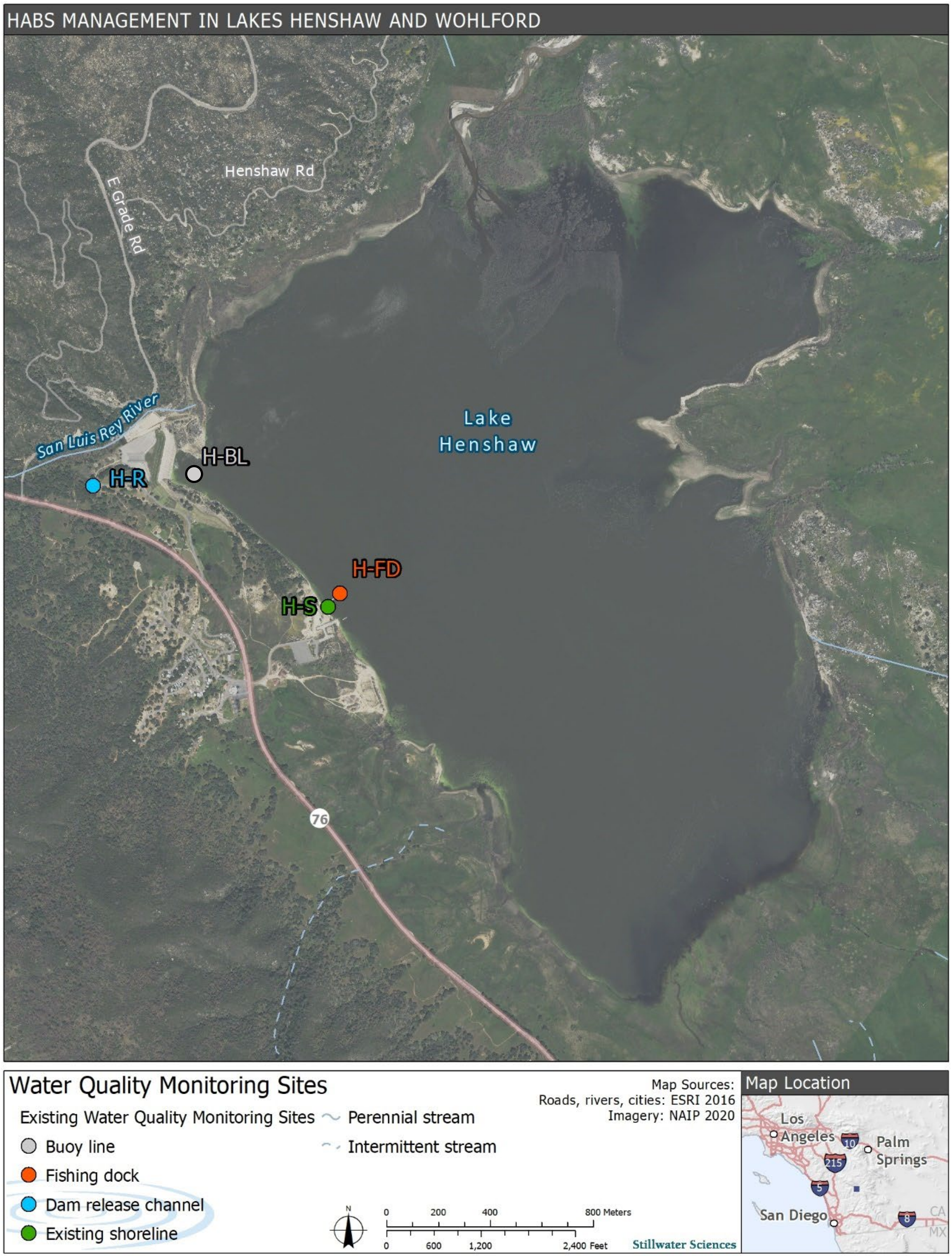


Figure 5-2. Existing water quality monitoring sites in Lake Henshaw. Shoreline (H-S; green), Fishing dock (H-FD; orange), Buoy line bottom (H-BL; grey), Buoy line surface (H-BLS; grey), and Release (H-R; light blue).

5.5.1.2 Monitoring goals and objectives

The goal of the synoptic monitoring is to establish a set of representative monitoring sites (index sites and rapid response monitoring sites) in Lake Henshaw that characterize the range of spatial and seasonal variability of common HAB indicators and cyanotoxins and provide an early warning that HAB development may be occurring.

The following monitoring objective has been developed to meet the above goals:

- Conduct multi-day seasonal synoptic surveys of Lake Henshaw surface waters to characterize *in situ* water quality conditions, make observations of cyanobacteria surface scums/accumulations, and quantify chlorophyll-a and relative levels of toxin producing cyanobacteria at multiple open water and shoreline sites.

5.5.1.3 Monitoring area

The monitoring area for establishing index sites and rapid response monitoring sites is Lake Henshaw, including approximately 7 miles of shoreline and approximately 1.2 square miles of open water areas, to establish the most informative locations for early warning/rapid response monitoring.

5.5.1.4 Monitoring design

Monitoring sites

Synoptic monitoring sites will include five shoreline sites, including the existing shoreline site, five open water sites, including the existing buoy line surface site, and the fishing dock site, for a total of 11 sites (Figure 5-3). The specific locations of the other shoreline and open water sites will be determined based on field conditions and will be roughly evenly distributed throughout the open water area of the lake and along the approximately 7 miles of shoreline. Three of the five open water sites will be in relatively deep water locations. All monitoring sites will be accessible by boat, such that the shoreline site locations may adjust with lake water level, moving along a transect perpendicular to the shoreline.

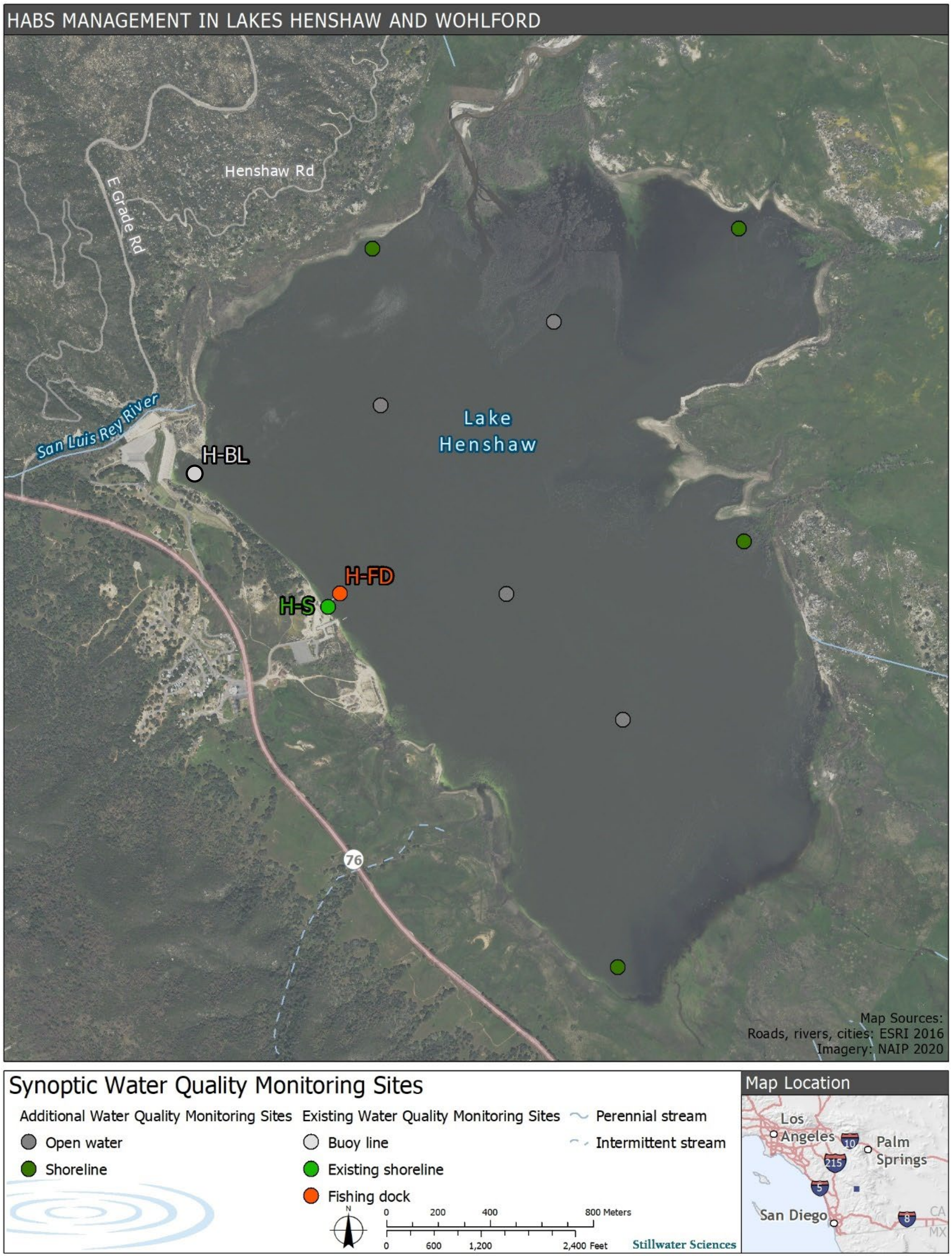


Figure 5-3. Approximate locations of monitoring sites to establish index sites and rapid response monitoring sites in Lake Henshaw. Existing shoreline (H-S; green), fishing dock (H-FD; orange), and buoy line surface (H-BLS; grey) sites will be included.

Monitoring frequency and parameters

Synoptic monitoring will be conducted three times during the year, to represent the late spring/early summer algal bloom period (May/June), the late summer/early fall peak algal bloom period (August/September), and the winter period when an algal bloom may not be active (November/December). During each seasonal period, monitoring will be conducted over 2–3 days. *In situ* water quality parameters will be collected as daytime vertical profiles at the fishing dock site, three deep water sites (including the buoy line site), and two shallow water sites (Table 5-3). Additionally, two sondes will be used to collect continuous data (i.e., every 15 minutes) for a minimum of a 48-hour monitoring period at the buoy line site and one of the two open water sites that are in relatively shallow water locations.

Table 5-3. Water quality parameters to be monitored during Lake Henshaw seasonal synoptic surveys associated with the use of algaecides as a short-term mitigation strategy.

Parameter	Location	Number of Samples
Latitude and longitude	Monitoring site	11
Wind speed and prevailing wind direction (average hourly for the duration of the survey period) [1]	Lake-wide	1
In situ water quality instantaneous daytime measurements (water temperature, dissolved oxygen, conductivity, pH, turbidity and/or chlorophyll-*a* and BGA-phycocyanin) [2]	Vertical profile	6 including H-FD, H-BL, 2 open water deep (>5 ft) sites, and 2 open water shallow (< 5 ft) sites
In situ water quality diel measurements (water temperature, dissolved oxygen, conductivity, pH, turbidity and/or chlorophyll-*a* and BGA-phycocyanin) over 48-hour period [1]	~ 0.5 ft above sediment water interface	2 including H-BL and 1 open water shallow (< 5 ft) site
Secchi depth	Water column	11 including H-FD, H-BL, H-S
Chlorophyll-a	Surface	11 including H-FD, H-BL, H-S
Cyanobacteria surface scum/accumulation	Surface	11 including H-FD, H-BL, H-S
Cyanobacteria identification and photograph (PTOX)	Surface	11 including H-FD, H-BL, H-S
Total anatoxin-producing cyanobacteria (qPCR)	Surface	11 including H-FD, H-BL, H-S
Total microcystin-producing cyanobacteria (qPCR)	Surface	11 including H-FD, H-BL, H-S

1 Data available at 10-minute increments at nearby Warners (WARSD) or Mataguay (MGYSD) sites (https://mesowest.utah.edu/index.html).

2 Chlorophyll-*a* and BGA-phycocyanin are also available as *in situ* parameters, depending on the instrument used. Collection of chlorophyll-*a* as *in situ* and grab sample measurements will support determination of an empirical relationship between the *in situ* and laboratory-analyzed parameters.

5.5.1.5 Monitoring methods

***In situ* water quality**

The District will measure *in situ* water quality parameters (water temperature, dissolved oxygen, conductivity, pH, turbidity [and/or chlorophyll-*a* and BGA-phycocyanin]) as instantaneous daytime measurements using a pre-calibrated, multi-probe sonde with designated sensors (Table

5-4). The sonde will be deployed from a boat and *in situ* measurements will be recorded at 1-ft depth intervals and within 0.5 ft from the bottom sediments. For the *in situ* diel measurements of water quality parameters (water temperature, dissolved oxygen, pH, turbidity [may also include chlorophyll-a or BGA-phycocyanin]), the sondes will be deployed approximately 0.5 ft above the sediment water interface for collection of continuous data (i.e., every 15 minutes) for a minimum of a 48-hour monitoring period at each site. Secchi depth readings will be collected at each reservoir site as a separate measure of transparency and to ascertain the approximate depth of the photic zone.

Table 5-4. *In situ* water quality methods for Lake Henshaw synoptic monitoring.

Parameter	Method	Units	MDL
Water temperature	EPA 170.1	degrees Celsius (°C)	0.1
DO	SM 4500-O(G)	milligrams per liter (mg/L)	0.1
Conductivity	SM 2510-B	micro Siemens per centimeter (uS/cm)	1.0
pH	SM 4500-H	standard unit of pH (s.u.)	0.1
Turbidity (Nephelometric)	EPA 180.1; ISO 7027-1; SM 2130-B	Nephelometric Turbidity Unit (NTU) or Formazin Nephelometric Unit (FNU)	0.1
Chlorophyll-*a* or BGA-phycocyanin	Optical luminescence	Reflective Fluorescence Unit (RFU) or micrograms per liter (ug/L)	0.01
Secchi depth (Secchi disk)	N/A	meters (m)	0.1

DO = dissolved oxygen
EPA = Environmental Protection Agency
MDL = method detection limit
SM = Standard Methods

Analytical water quality

The District will analyze water quality constituents and cyanobacteria samples using methods presented in Table 5-5. Analytical samples will be collected as surface grab samples in the top 1 ft of the water column.

Table 5-5. Analytical water quality methods for Lake Henshaw synoptic monitoring.

Constituent	Method	MDL	Hold Time
Chlorophyll-*a*	EPA 445.0	0.5 ug/L	8 hours (refrigerated 4°C)*
Cyanobacteria identification and photograph (PTOX)	Microscopy	--	24 hours (refrigerated 4°C)
Total anatoxin-producing cyanobacteria (qPCR)	Molecular (quantitative polymerase chain reaction [qPCR])	1 copy/mL	4 days at 4°C
Total microcystin-producing cyanobacteria (qPCR)			

* Sample should be analyzed as soon as possible
EPA = Environmental Protection Agency
MDL = method detection limit

Quality assurance/Quality control

For instantaneous *in situ* water quality measurements, the sonde will be calibrated at the start of each sampling day and checked at the end of the day. For 48-hour deployments, the sondes will be calibrated before they are deployed and checked immediately after deployment. Calibration data will be recorded on a calibration log that includes measurement quality objective criteria for real-time comparison. All analytical samples will be collected, handled and delivered to the analytical laboratory consistent with standard methods (Table 5-5). Appropriate QA/QC methods and documentation will be followed. All sample bottles will be rinsed with water from the water body to be collected from. QA/QC in the field will be assured by accurate and thoroughly completed sample labels, field sheets, and chain of custody and sample log forms. Sample labels will include sample identification code, date, time, site name, sampling location, collector's name, sample type and preservative, if applicable.

Data analysis and reporting

Monitoring data will be compiled and organized by seasonal event, parameter/constituent, and/or monitoring location. Figures and tables will be developed to display and evaluate spatial trends by season, with particular comparisons between shoreline sites and between open water sites to identify representative index sites and rapid response monitoring sites within those two groupings. A short technical memorandum will be developed to summarize the results of the monitoring and to inform monitoring discussed in Section 5.5.2.

5.5.1.6 Cost estimate

Estimated costs for synoptic monitoring in Lake Henshaw are summarized in Table 5-6 and assume that project management, field work, coordination with the analytical laboratory, data entry and QA/QC, and data analysis and reporting would be undertaken by a contractor.

Table 5-6. Estimated monitoring costs for Lake Henshaw synoptic monitoring to establish index and rapid response monitoring sites for short-term HABs mitigation using algaecides.

Task	Labor Hours		Labor Cost (full)		Field Expenses Cost		Contract Laboratory Expenses [e]	Total Cost
	Contractor	District Staff	Contractor	District Staff	Contractor	District Staff		
Task 1 Project Management [a]	36	12	$5,300	$1,500	$0	$0	$0	$6,800
Task 2 Seasonal Synoptic Field Surveys [b, c]	320	12	$37,700	$1,500	$14,400	$0	$5,700	$59,300
Task 3 Data Analysis and Reporting [d]	97	24	$12,500	$3,000	$0	$0	$0	$15,500
Total	**453**	**48**	**$55,500**	**$6,000**	**$14,400**	**$0**	**$5,700**	**$81,600**

[a] Assumes monthly invoicing and up to three as-needed progress meetings with the District PM conducted by teleconference. Meetings up to 0.5-hour each. Hourly labor estimates assume 2022 rates.

[b] Synoptic monitoring cost estimate assumes three sampling events at three days each; late spring/early summer (May/June), late summer/early fall (August/September), and winter (November/December). Cost estimate assumes contractor conducts field work and coordination with analytical laboratory. Hourly labor estimates assume 2022 rates.

[c] Contractor field expenses include travel, lodging, and monitoring equipment (i.e., one water quality *in situ* meter and two water quality sondes, Secchi disk), and sample shipping to the analytical laboratory. Costs assume that the District will provide the boat and outboard motor for field work.

[d] Cost estimate assumes contractor conducts data entry, QA/QC, data analysis, and reporting and is based on 2022 rates.

[e] Expenses include chlorophyll-a at $40/sample; cyanobacteria identification and photograph (PTOX) at $40/sample; qPCR total microcystin-producing cyanobacteria at $42/sample; qPCR total anatoxin-a-producing cyanobacteria at $42/sample.

5.5.1.7 Responsibilities to implement monitoring, analysis, and reporting

The District may elect to have synoptic monitoring undertaken by a contractor. Analytical water quality samples collected during seasonal monitoring events will be shipped by District or contractor field staff to the analytical laboratory or laboratories possessing the appropriate expertise. Overall data analysis and reporting may also be undertaken by the contractor at the District's direction.

5.5.2 Routine, rapid response, effectiveness, and permit-related monitoring associated with the use of algaecides as a short-term mitigation strategy

5.5.2.1 Monitoring goals and objectives

The goal of routine monitoring is to provide the District with evidence that a HAB may be developing in Lake Henshaw, while the goal of rapid response monitoring is to allow sufficient response time for the successful implementation of algaecide as a short-term HAB mitigation strategy. HAB mitigation success will involve using the least amount of algaecide necessary to interrupt cyanotoxin production in the lake, consistent with limitations dictated by the *Warner Wellfield and Lake Henshaw Aquatic Pesticide Application Plan* (Marine Biochemists 2021) and the District's interest in minimizing the use of chemicals in the lake. See also Section 5.5.1.1 for a discussion of the need for rapid response monitoring.

The following monitoring objectives have been developed to meet the above goals:

- Conduct routine monitoring of a targeted set of in situ parameters (i.e., water temperature), analytical constituents (i.e., chlorophyll-a, cyanobacteria identification and photograph [PTOX], occasional total and/or cyanobacteria cell counts), and cyanotoxin screening (i.e., microcystin, anatoxin-a) in Lake Henshaw surface waters;
- Conduct once annual screening for the cyanotoxin cylindrospermopsin in Lake Henshaw surface waters15;
- Conduct rapid response monitoring, as needed, of a larger set of variables, including wind speed and direction (lake-wide), in situ parameters (i.e., water temperature, dissolved oxygen, conductivity, pH, turbidity), analytical constituents (i.e., chlorophyll-a, cyanobacteria identification and photograph [PTOX]), and quantified cyanotoxins (i.e., microcystin, anatoxin-a), to rapidly characterize the extent of the algal bloom and the distribution of cyanotoxins throughout lake surface waters and aid in establishing appropriately located and sized algaecide treatment zones;
- Conduct algaecide effectiveness monitoring, as needed, following treatment; and,
- Conduct monitoring required under the Statewide Aquatic Weed Control Permit.

5.5.2.2 Monitoring area

The monitoring area for routine and rapid response monitoring is Lake Henshaw.

[15] To date, this toxin has not been detected in Lake Henshaw (Section 2.2.4).

5.5.2.3 Monitoring design

Monitoring sites

The location of routine and rapid response monitoring sites will be determined as described in Section 5.5.1. The number of routine monitoring sites is expected to range from 1–3 index sites including at least one shoreline site and one open water site. The number of rapid response monitoring sites is expected to range from 3–5 shoreline and 2–3 open water sites, but the ultimate number of sites will depend on the results of the monitoring described in Section 5.5.1.

Monitoring frequency, parameters, and operational triggers

Monitoring associated with the use of algaecides as a short-term mitigation strategy will include routine monitoring, rapid response monitoring, algaecide effectiveness monitoring, and monitoring required under the Statewide Aquatic Weed Control Permit (Table 5-7). Additional details are presented below for each type of monitoring, including the associated operational triggers.

Table 5-7. Water quality parameters to be monitored, monitoring frequency, and number of sites for each type of Lake Henshaw monitoring associated with the use of algaecides as a short- and long-term mitigation strategy.

Parameter	Type/ Location	Routine Monitoring		Rapid Response Monitoring		Algaecide Effectiveness Monitoring		Statewide Aquatic Weed Control Permit	
		Frequency	Number of Sites	Frequency	Number of Sites	Frequency	Number of Sites	Frequency	Number of Sites
Wind speed	Lake-wide	Weekly	1	Every 1–7 days	1	Weekly	1	–	–
Prevailing wind direction	Lake-wide	Weekly	1	Every 1–7 days	1	Weekly	1	–	–
In situ water temperature	Surface	Weekly	1–3 [a]	–	–	–	–	Three times (BG, EM, PE) [c]	1
In situ dissolved oxygen, conductivity, pH, turbidity)	Surface	–	–	–	–	–	–	Three times (BG, EM, PE) [c]	1
In situ water quality (water temperature, dissolved oxygen, pH, turbidity) [d]	Vertical profile	–	–	Every 1–7 days	1–3 [b] (deep water sites only)	Weekly	1–3 [b] (deep water sites only)	–	–
Secchi depth	Water column	Weekly	1–3 [a]	Every 1–7 days	5–8 [b]	Weekly	5–8 [b]	–	–
Total hardness ($CaCO_3$)	Surface grab	–	–	–	–	–	–	Three times (BG, EM, PE) [c]	1
Total copper [e]	Surface grab	–	–	–	–	–	–	Three times (BG, EM, PE) [c]	1
Chlorophyll-a [d]	Surface grab	Bi-weekly	1–3 [a]	Weekly	5–8 [b]	Weekly	5–8 [b]	–	–
Cyanobacteria surface scum/accumulation	Visual	Weekly	1–3 [a]	Every 1–7 days	5–8 [b]	Weekly	5–8 [b]	Three times (BG, EM, PE) [c]	1
Cyanobacteria genus identification and photograph (PTOX)	Surface	Weekly	1–3 [a]	–	–	Weekly	5–8 [b]	–	–

Parameter	Type/ Location	Routine Monitoring		Rapid Response Monitoring		Algaecide Effectiveness Monitoring		Statewide Aquatic Weed Control Permit	
		Frequency	**Number of Sites**	**Frequency**	**Number of Sites**	**Frequency**	**Number of Sites**	**Frequency**	**Number of Sites**
Total and/or cyanobacteria cell counts and identification of potential toxin producing species	Surface grab	Every six weeks [d]	1–3 [a]	Once prior to treatment	1–3	–	–	–	–
Total anatoxin-a	Surface grab	Weekly	1–3 [a]	Every 1–7 days	5–8 [b]	Weekly	5–8 [b]	–	–
Total cylindrospermopsins	Surface grab	Annually	1–3 [a]	As needed	As needed	As needed	As needed	–	–
Total microcystins/ nodularins	Surface grab	Weekly	1–3 [a]	Every 1–7 days	5–8 [b]	Weekly	5–8 [b]	–	–

[a] Routine monitoring sites will include at least one shoreline site and one open water site.

[b] The number of rapid response monitoring sites is expected to range from 3–5 shoreline and 2–3 open water sites, but the ultimate number of sites will depend on the results of the monitoring described in Section 5.4.1.

[c] BG = background monitoring prior to treatment; EV = event monitoring during treatment; PE = post-event monitoring following treatment (see also page 175).

[d] The District may elect to undertake *in situ* continuous monitoring of chlorophyll-a (ug/L or RFU) rather than laboratory-analyzed chlorophyll-a, as a long-term cost-saving measure. If this occurs, then *in situ* chlorophyll-a can replace laboratory-analyzed chlorophyll-a in the set of routine monitoring parameters once a relationship between the *in situ* and laboratory-analyzed parameters has been empirically determined. Establishment of this relationship should involve a sample size of at least 12 per season, including both open water and shoreline sites, and for three seasons (i.e., late spring/early summer algal bloom period (May/June), the late summer/early fall peak algal bloom period (August/September), and the winter period when an algal bloom may not be active [November/December]).

[e] If copper is the active ingredient in algaecide product.

[d] Sample collection weeks for total and/or cyanobacteria cell counts should correspond with weeks when chlorophyll-a surface grabs are collected.

Routine monitoring
Routine monitoring will include the parameters listed in Table 5-7. Routine monitoring will occur at 1–3 index sites on a weekly basis unless otherwise noted in Table 5-7.

Triggers for Moving to Rapid Response Monitoring (see also Figure 5-4)
If the District has not met all water delivery obligations for the current water year, and it is not possible to delay any planned releases by weeks or months to allow a potentially long-lasting HAB to run its course, then the trigger for moving to rapid response monitoring is the following:

- *Detection of anatoxin-a at one or more index sites; or*
- *Microcystin ≥ 0.5 ug/L collected from one or more index sites.*

Knowledge gained during previous monitoring periods should be used to evaluate whether monitoring goals and objectives are being met through the Lake Henshaw water quality monitoring program, and this evaluation may result in recommendations for revisions to the routine monitoring parameters and operational triggers (see also Section 5.1).

Rapid response monitoring
Rapid response monitoring will include the parameters listed in Table 5-7. Rapid response monitoring will occur at shoreline and open water sites for the rapid response monitoring parameters listed in Table 5-7 as soon as possible upon reaching the rapid response monitoring triggers.

The District may elect to use rapid screening tests for routine monitoring of one or more cyanotoxins (see Table 5-9). Currently, rapid tests are available for microcystin/nodularins and cylindrospermopsin, but not for anatoxin-a (Appendix D).

If the District uses rapid screening tests for routine monitoring, then rapid response monitoring will begin within 2–3 days of the rapid screening test results and within 24 hours of receiving confirmation of the rapid screening test results from the analytical laboratory. This timeline may require expedited processing by the analytical laboratory.

If the District does not use rapid screening tests for routine monitoring, then rapid response monitoring will begin within 1−3 days of receiving analytical laboratory results.

Rapid response monitoring at Lake Henshaw will occur every 1−7 days thereafter, unless otherwise noted, depending on the course of the HAB event.

Triggers for Moving to Algaecide Treatment (see also Figure 5-4)
Unless precluded by other operational considerations, the trigger for moving to algaecide treatment is either of the below at any single rapid response monitoring site:

- *Detection of anatoxin-a; or*
- *Microcystin ≥ 0.8 ug/L.*

If triggers for moving to algaecide treatment are met, then up to three total cell count samples, including identification of potential toxin producing species, will be collected from Lake Henshaw as soon as possible to inform the minimum dose of algaecide that is anticipated to be effective.

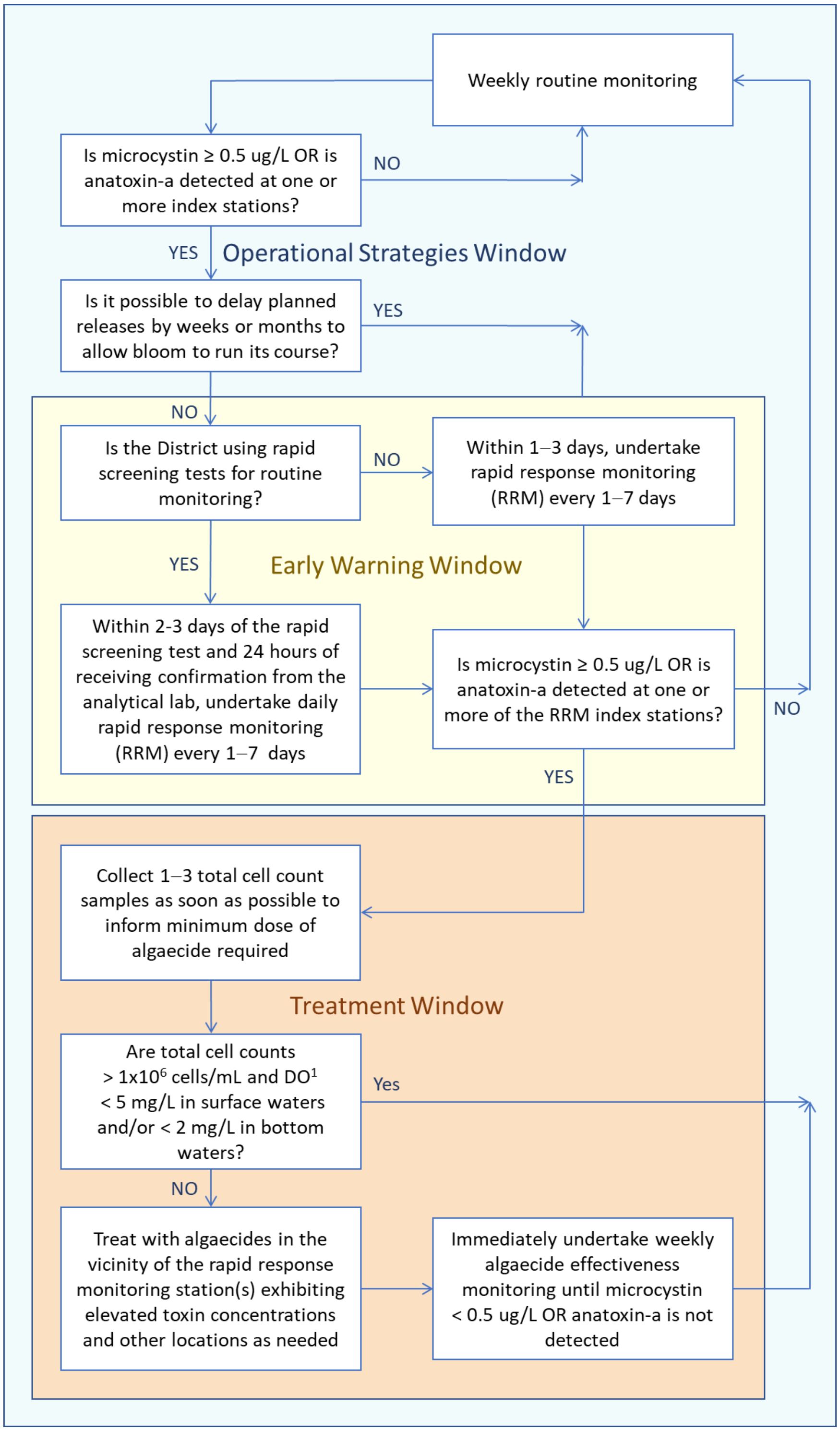
For large-scale treatment within the lake, if dissolved oxygen measured before 0900 at multiple rapid response monitoring sites is generally less than 5 mg/L in surface waters and/or less than 2 mg/L in bottom waters, and total cell counts are greater than $1x10^6$ cells/mL, then the HAB may be too dense to be effectively treated with an algaecide (see Figure 5-1) and/or a fish kill is possible. In this case, the District will return to routine monitoring on a weekly basis to track progression of the HAB. If total cell count data are not available to inform selection of the minimum effective algaecide dose, then dosing will default to a moderate level. Note that cyanobacteria cell counts provide valuable information regarding the intensity of a cyanobacteria bloom, although they do not indicate the intensity of other phytoplankton (e.g., green algae, diatoms, golden algae) that also may be present and affecting water quality in the lake.

Algaecide treatment will occur in the vicinity of the rapid response monitoring site(s) exhibiting elevated toxin concentrations and/or cell counts, and, as needed, in other locations that exhibit cyanobacteria surface scums/accumulations and/or cell counts that are indicative of a bloom. *Algaecide treatment will occur within 1−7 days of the trigger.* Treatment will occur consistent with the final APAP for Lake Henshaw and the Warner Ranch (Vista Irrigation District 2021).

Algaecide effectiveness monitoring

Following algaecide treatment, algaecide effectiveness monitoring will include the parameters listed in Table 5-6. Unless otherwise noted in Table 5-6, algaecide effectiveness monitoring will be conducted at all rapid response monitoring sites and for all rapid response monitoring parameters to determine the response of the HAB. Algaecide effectiveness monitoring will begin immediately following algaecide application and will occur on a weekly basis until:

- *Anatoxin-a is not detected; and*
- *Microcystin is at stable background levels (e.g., <0.5 ug/L).*



[1] Dissolved oxygen (DO) monitoring results on the day of planned treatment or one day prior.

Figure 5-4. Cyanotoxin routine, rapid response, and algaecide effectiveness monitoring framework and operational triggers flowchart for Lake Henshaw. Operational strategies window (blue), early warning window (yellow), and treatment window (orange) are discussed further in Section 5.1.

Monitoring required under the Statewide Aquatic Weed Control Permit

Coverage under the California Statewide General National Pollutant Discharge Elimination System (NPDES) Permit for Residual Aquatic Pesticide Discharges to Waters of the United States from Algae and Aquatic Weed Control Applications Permit # CAG990005 requires that certain monitoring activities be conducted in association with algaecide application to control nuisance blooms. The *Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch* (Marine Biochemists 2021) includes an Aquatic Pesticide Monitoring and Reporting Program (APMRP) for Lake Henshaw that is compliant with NPDES requirements.

The APMRP addresses the following two key questions:

1. Does the residual algaecides and aquatic herbicides discharge cause an exceedance of the receiving water limitations?
2. Does the discharge of residual algaecides and aquatic herbicides, including active ingredients, inert ingredients, and degradation byproducts, in any combination cause or contribute to an exceedance of the "no toxics in toxic amount" narrative toxicity objective?

Prior to, during, and following algaecide treatment, monitoring required for the APMRP will include the parameters listed in Table 5-6. The three types of required monitoring for each treatment event include background monitoring (BG) in the treatment area just prior to the treatment event (i.e., up to 24 hours in advance of treatment); event monitoring (EM) proximally adjacent to the treatment location, in the portion of the treatment area that is exposed to the algaecide plume immediately or shortly after the treatment event; and post-event monitoring (PE) adjacent to the treatment area within one week after the treatment event. One full set of three samples (i.e., BG, EM and PE samples) will be collected during each treatment event along with quality control samples. For products that contain sodium carbonate peroxyhydrate, no treatment event monitoring for residuals is required before or after treatment because peroxyhydrate breakdown products are water and bicarbonate. For application of all other algaecides and aquatic herbicides listed in the NPDES permit, monitoring will be conducted for each active ingredient utilized at the time of treatment event.

APMRP details regarding monitoring procedures for in-field and grab sampling, laboratory analyses, monitoring records, calibration and maintenance of instrumentation, maximum allowable copper concentrations in Lake Henshaw following treatment (if copper is the active ingredient in the applied algaecide), and reporting are presented in Appendix C.

5.5.2.4 Monitoring methods

***In situ* water quality**

The District will measure *in situ* water quality parameters (water temperature, dissolved oxygen, conductivity, pH, turbidity [or chlorophyll-*a* or BGA-phycocyanin]) using a pre-calibrated multi-probe sonde with designated sensors (Table 5-8). The sonde will be deployed from a boat and *in situ* measurements will be recorded at 1-ft depth intervals and within 0.5 ft from the bottom sediments. Secchi depth readings will be collected at each reservoir site as a separate measure of transparency and to ascertain the approximate depth of the photic zone.

Table 5-8. *In situ* water quality methods for algaecide-related monitoring in Lake Henshaw.

Parameter	Method	Units	MDL
Water temperature	EPA 170.1	degrees Celsius (°C)	0.1
DO	SM 4500-O(G)	milligrams per liter (mg/L)	0.1
Conductivity	SM 2510-B	micro Siemens per centimeter (uS/cm)	1.0
pH	SM 4500-H	standard unit of pH (s.u.)	0.1
Turbidity (Nephelometric) [a]	EPA 180.1; ISO 7027-1; SM 2130-B	Nephelometric Turbidity Unit (NTU)/Formazin Nephelometric Unit (FNU)	0.1
Chlorophyll-*a* and BGA-phycocyanin [b]	Optical luminescence	Reflective Fluorescence Unit (RFU) or micrograms per liter (ug/L)	0.01
Secchi depth (Secchi disk)	N/A	meters (m)	0.1

DO = dissolved oxygen
EPA = Environmental Protection Agency
MDL = method detection limit
SM = Standard Methods

[a] Chlorophyll-a or BGA-phycocyanin may be substituted for turbidity.

[b] The District may elect to undertake *in situ* continuous monitoring of chlorophyll-a (ug/L or RFU), rather than laboratory-analyzed chlorophyll-a, as a long-term cost-saving measure. If this occurs, then *in situ* chlorophyll-a can replace laboratory-analyzed chlorophyll-a in the set of routine monitoring parameters (Table 5-6) once a relationship between the *in situ* and laboratory-analyzed parameters has been empirically determined. Establishment of this relationship should involve a sample size of at least 12 per season, including both open water and shoreline sites, and for three seasons (i.e., late spring/early summer algal bloom period (May/June), the late summer/early fall peak algal bloom period (August/September), and the winter period when an algal bloom may not be active [November/December]).

Analytical water quality

The District will collect analytical water quality constituents and cyanobacteria samples using methods presented in Table 5-9. Analytical samples will be collected as grab samples in the top 1 ft of the water column (i.e., at the surface).

Table 5-9. Analytical water quality methods for algaecide-related monitoring in Lake Henshaw.

Constituent	Method	MDL	Sample Volume	Container Type	Hold Time
Chlorophyll-*a*	EPA 445.0	0.5 ug/L	125 mL	HDPE	8 hours (refrigerated 4°C)*
Cyanobacteria identification and photograph (PTOX)	Microscope	--	100 mL	HDPE	24 hours (refrigerated 4°C)
Total and/or cyanobacteria cell counts and identification of potential toxin producing species	Microscope	--	1 L	HDPE	24 hours (refrigerated 4°C)
Total microcystins/ nodularins	ELISA	0.15 ug/L	100 mL [1]	HDPE	4 days (refrigerated 4°C)
	LC/MS	0.04 ug/L			
	Rapid screening test [1]	≥ 0.5 ug/L	25 mL	HDPE	4 hours (refrigerated 4°C)
Total anatoxin-a	ELISA	0.17 ug/L	100 mL[1]	HDPE	4 days (refrigerated 4°C)
	LC/MS	0.04 ug/L			
	Rapid screening test [1]	--	25 mL	HDPE	4 hours (refrigerated 4°C)
Total cylindro-spermopsins	ELISA	0.05 ug/L	100 mL[1]	HDPE	4 days (refrigerated 4°C)
	LC/MS	0.04 ug/L			
	Rapid screening test [1]	≥ 0.5 ug/L	25 mL	HDPE	4 hours (refrigerated 4°C)

* Sample should be analyzed as soon as possible
EPA = Environmental Protection Agency
MDL = method detection limit
HDPE = high-density polyethylene
[1] If a rapid screening test is used, the monitoring protocol would involve collecting a sufficient sample volume in the field to run the rapid screening test at the District offices and to send a confirmatory grab sample to the analytical laboratory for expedited confirmation via ELISA or LC/MS. Currently rapid screening tests are not available for anatoxin-a, but they are understood to be under development (see also Appendix D).

Quality assurance/quality control

For *in situ* water quality, the sonde will be calibrated at the start of each sampling day and checked at the end of the day. Calibration data will be recorded on a calibration log that includes measurement quality objective criteria for real-time comparison. All analytical samples will be collected, handled and delivered to the analytical laboratory consistent with standard methods (Table 5-9). Appropriate QA/QC methods and documentation will be followed. All sample bottles will be rinsed with water from the water body to be collected from. QA/QC in the field will be assured by accurate and thoroughly completed sample labels, field sheets, and chain of custody and sample log forms. Sample labels will include sample identification code, date, time, site name, sampling location, collector's name, sample type, and preservative, if applicable.

Data analysis and reporting

Monitoring data will be compiled and organized by monitoring event, parameter/constituent, and/or monitoring location. Figures and tables will be developed to display and evaluate spatial and temporal trends, with particular comparisons between rapid response monitoring and algaecide effectiveness monitoring. Monitoring data will be analyzed and interpreted at minimum on an annual basis (see also Section 5.1).

5.5.2.5 Cost estimate

Estimated costs for monitoring associated with the use of algaecides as a short-term mitigation strategy are summarized in Table 5-10 and assume that project management, field work, coordination with the analytical laboratory, data entry and QA/QC, and data analysis would be undertaken by the District.

Table 5-10. Estimated annual monitoring costs for monitoring associated with the use of algaecides as a short-term mitigation strategy in Lake Henshaw.

Task	Labor Hours		Labor Cost (full)		Field Expenses Cost		Contract Laboratory Expenses [f]	Total Cost
	Contractor	District Staff	Contractor	District Staff	Rental Equipment [e]	District Staff		
Task 1 Routine Monitoring [a]	0	208	$0	$26,000	$0	$0	$75,600	$101,600
Task 2 Rapid Response Monitoring [b]	0	48	$0	$6,000	$3,400	$0	$26,100	$35,500
Task 3 Algaecide Effectiveness Monitoring [c]	0	64	$0	$8,000	$1,400	$0	$11,900	$21,300
Task 4 Statewide Aquatic Weed Control Permit Monitoring [d]	Costs included in algaecide application contract							
Total	**0**	**320**	**$0**	**$40,000**	**$4,800**	**$0**	**$113,600**	**$158,400**

[a] Assumes weekly monitoring at each of three index monitoring sites. Hourly labor estimates assume 2022 rates.
[b] Assumes two rapid response monitoring events per year. Each event includes sample collection at 9 monitoring sites, 3x per event, with samples collected every 1–3 days. Hourly labor estimates assume 2022 rates.
[c] Assumes two post-algaecide treatment monitoring events per year. Each event includes sample collection at nine monitoring sites, one time per event, with samples collected once per week. Hourly labor estimates assume 2022 rates.
[d] Assumes contractor conducts algaecide application and required monitoring and reporting for the Statewide aquatic weed control permit.
[e] Assumes equipment rental expenses are $350 per day plus $600 shipping each way per event.
[f] Expenses include chlorophyll-a at $40/sample; cyanobacteria cell counts at $225/sample; cyanobacteria identification and photograph (PTOX) at $40/sample; ELISA total microcystin-producing cyanobacteria at $105/sample; ELISA total anatoxin-a-producing cyanobacteria at $130/sample; ELISA cylindrospermopsin-producing cyanobacteria at $130/sample.

5.5.2.6 Responsibilities to implement monitoring, analysis, and reporting

In situ monitoring and grab sample collection associated with the use of algaecides as a short-term mitigation strategy will be undertaken by District staff. Analytical water quality samples will be shipped by District staff to one or more laboratories that possess the appropriate expertise. Overall data analysis and reporting would also be undertaken by the District, with consultant support at the District's discretion.

5.6 San Luis Rey River and Escondido Canal

5.6.1 Routine monitoring

5.6.1.1 Monitoring goals and objectives

The goal of routine monitoring in the San Luis Rey River downstream of Lake Henshaw and the Escondido Canal is to provide the District with evidence that a HAB is affecting release waters that flow downstream through a campground operated by the La Jolla Band of Indians on their reservation lands, through the Rincon Reservation lands, and (for diverted flows) through the Escondido Canal and into Lake Wohlford. While monitoring in Lake Henshaw (Section 5.5) and Lake Wolford (Section 5.7) currently is focused on use of algaecides as a short-term mitigation strategy, monitoring in the San Luis Rey River and the Escondido Canal between Lake Henshaw and Lake Wohlford is focused on periods when releases are occurring, regardless of whether an algaecide treatment has recently taken place in Lake Henshaw.

The following monitoring objective has been developed to meet the above goal:

- Conduct routine monitoring of cyanotoxin concentrations (i.e., microcystin, anatoxin-a) along a longitudinal transect from the Lake Henshaw release to the canal inlet at Lake Wohlford.

5.6.1.2 Monitoring area

The monitoring area for routine monitoring is the San Luis Rey River from the Lake Henshaw release to the canal inlet at Lake Wohlford.

5.6.1.3 Monitoring design

Monitoring sites

The location of monitoring sites is provided in Table 5-11.

Table 5-11. San Luis Rey River and Escondido Canal monitoring sites.

Monitoring Responsibility	Site ID	Location	Latitude	Longitude
District	H-R	Dam release channel approximately 10 ft upstream of flow measurement weir (point of release to San Luis Rey River)	33.23923°N	116.76594°W
District	H-RR	Rey River Ranch	33.23963°N	116.76174°W
La Jolla Band	SWSLRCC8	La Jolla Band Campground Lower Site	33.23923°N	116.76594°W
La Jolla Band	DD/SWSLRK12	Diversion Dam/La Jolla Band Diversion Dam Site	33.27224°N	116.82998°W
District	PG	Paradise Grates	33.27230°N	116.84936°W

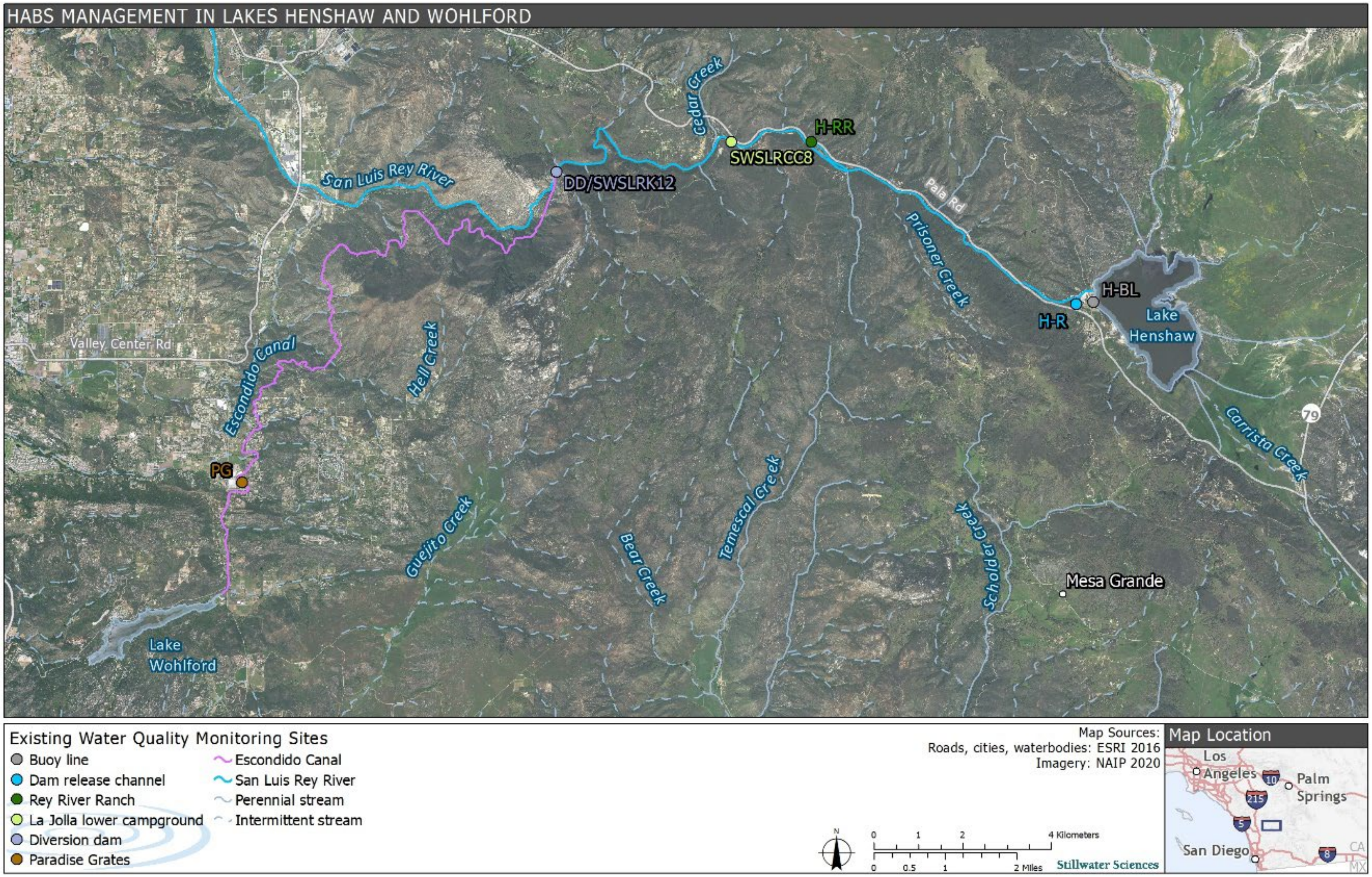


Figure 5-5. San Luis Rey River and Escondido Canal monitoring sites.

Monitoring frequency, parameters, and operational triggers
Routine monitoring will include sampling and analysis of microcystin and anatoxin-a concentrations using methods presented in Table 5-9. Routine monitoring will occur at all District sites in Table 5-11 on at least a weekly basis while flows are being released from Lake Henshaw. Routine monitoring at the San Luis Rey River and the Escondido Canal sites should occur on the same dates as monitoring of the Lake Henshaw sites to allow for analysis of a longitudinal transect between Lake Henshaw and Lake Wohlford.

5.7 Lake Wohlford

5.7.1 Synoptic monitoring to establish index sites and rapid response monitoring sites

5.7.1.1 Background

Cyanotoxins are currently monitored in Lake Wohlford at three sites in the lake (Table 5-12, Figure 5-6). Routine grab samples are collected on a monthly basis for microcystin (Section 2.4.2.4) and the results are used to determine when to shift to weekly rapid response monitoring and potentially algaecide application using the triggers discussed in Section 5.7.2. While the existing monitoring sites and frequency of monitoring are providing Escondido with sufficient information to control HABs in Lake Wohlford under current conditions, there may be a need to characterize the seasonal and spatial variability of cyanotoxins and cyanobacteria throughout the lake should HAB conditions in the upstream Lake Henshaw change and/or the Wohlford Dam be raised to create a larger and deeper lake. Under either of these conditions, or should the current algaecide treatment approach no longer adequately control HABs in Lake Wohlford, it would be critical to understand whether the existing routine and rapid response monitoring sites need to be expanded or moved to different locations using synoptic monitoring.

For routine monitoring activities, the establishment of a relatively small number of *index sites* is intended to provide data that are representative of other locations of interest, reducing the total number of samples that must be collected on a regular basis (e.g., weekly, monthly, quarterly) in order to characterize the broader study area. For cyanobacteria and cyanotoxins in lakes and reservoirs, surface accumulations can occur in open water and shoreline locations during periods of high solar insolation and low wind, and they can occur along downwind shoreline locations when winds increase, or they can be relatively isolated in enclosed bays. Thus, HAB-related index sites should include both open water and shoreline locations, and preferably be in easily accessible locations to facilitate regular monitoring. Routine monitoring for HABs management in Lake Wohlford is currently occurring as part of an *operational strategies window* (Figure **5-1**) when multiple strategies for reservoir operation and water delivery are available to Escondido.

While longer sampling intervals are important for documenting chemical and biological changes in a lake or reservoir over time, they are insufficient as an early warning mechanism for rapid algal bloom development and associated cyanotoxin events that can develop over just a few days. When a HAB is developing, additional data are often necessary to characterize the spatial variability of cyanotoxins and cyanobacteria throughout the waterbody, to help inform the necessary extent of any short-term mitigation strategies such as algaecide treatment, and to provide a broader basis for assessing the success of long-term water quality improvements in the reservoir as a whole. In order for monitoring to provide the necessary early warning that a HAB may be forming, this type of monitoring often involves a greater number of *rapid response monitoring* sites and a higher frequency of monitoring than routine monitoring. Rapid response monitoring is currently occurring during an *early warning window* (Figure 5-1) that is focused on the potential for elevated cyanotoxin concentrations to rapidly change reservoir operations and water treatment at the adjacent EVWTP.

Table 5-12. Lake Wohlford microcystin monitoring sites.

Location	Latitude	Longitude
Canal inlet	33°10'30.36"N	116°59'25.34"W
Near boat dock	33°10'23.40"N	116°59'56.16"W
Tower near dam	33°10'0.98"N	117° 0'14.87"W

Figure 5-6. Existing microcystin monitoring sites in Lake Wohlford.

5.7.1.2 Monitoring goals and objectives

The goal of future synoptic monitoring would be to establish a set of newly representative monitoring sites (index sites and rapid response monitoring sites) in Lake Wohlford that characterize the range of spatial and seasonal variability of common HAB indicators and cyanotoxins and provide an early warning that HAB development may be occurring.

The following monitoring objectives have been developed to meet the above goals:

- Conduct multi-day seasonal synoptic surveys of Lake Wohlford surface waters to characterize *in situ* water quality conditions, make observations of cyanobacteria surface scums/accumulations, and quantify chlorophyll-a and relative levels of toxin producing cyanobacteria at multiple open water and shoreline sites.

5.7.1.3 Monitoring area

The monitoring area for establishing index sites and rapid response monitoring sites under future synoptic monitoring is Lake Wohlford.

5.7.1.4 Monitoring design

Monitoring sites

Based on the current size of Lake Wohlford, synoptic monitoring sites will include three shoreline sites, including the existing near boat dock site, three open water sites, including a site proximal to the existing aeration diffuser, and the existing canal inlet, and tower near dam sites, for a total of eight sites (Figure 5-7). Once the dam has been raised, the number and location of sites may need to be changed to best represent the new condition. The specific locations of the shoreline and open water sites will be determined based on field conditions and will be roughly evenly distributed along the longitudinal transect of the lake. All monitoring sites will be accessible by boat, such that the shoreline site locations may adjust with lake water level, moving along a transect perpendicular to the shoreline.

Figure 5-7. Approximate locations of monitoring sites to establish index sites and rapid response monitoring sites in Lake Wohlford. Existing canal inlet, near boat dock, and tower near dam sites (white circles) will be included. One open water site nearest the existing aeration diffuser (orange circle) will also be included.

Monitoring frequency and parameters

Synoptic monitoring will be conducted three times during the year, to represent the late spring/early summer algal bloom period (May/June), the late summer/early fall peak algal bloom period (August/September), and the winter period when an algal bloom may not be active (November/December). During each seasonal period, monitoring will be conducted over two days. *In situ* water quality parameters will be collected as daytime vertical profiles at two shallow water sites and three deep water sites, including the tower near the dam site and the site nearest the aeration diffuser (Table 5-13). Additionally, two sondes will be used to collect continuous data (i.e., every 15 minutes) for a minimum of a 48-hour monitoring period in between the tower near dam and the site nearest the aeration diffuser, and at the canal inlet site.

Table 5-13. Water quality parameters to be monitored during Lake Wohlford seasonal synoptic surveys associated with the use of algaecides as a short-term mitigation strategy.

Parameter	Location	Number of Samples
Latitude and longitude	Monitoring site	5
Wind speed and prevailing wind direction (average hourly for the duration of the survey period) [1]	Lake-wide	1
In situ water quality instantaneous daytime measurements (water temperature, dissolved oxygen, conductivity, pH, turbidity and/or chlorophyll-*a* and BGA-phycocyanin) [2]	Vertical profile	5 including 3 deep water (>5 ft) sites (tower near the dam, nearest the aeration diffuser), 2 open water shallow (< 5 ft) sites
In situ water quality diel measurements (water temperature, dissolved oxygen, conductivity, pH, turbidity and/or chlorophyll-*a* and BGA-phycocyanin) over 48-hour period [1]	~ 0.5 ft above sediment water interface	2 including a site in between the tower near dam and the site nearest the aeration diffuser, and at the canal inlet site
Secchi depth	Water column	9 including 3 existing sites
Chlorophyll-a	Surface	9 including 3 existing sites
Cyanobacteria surface scum/accumulation	Surface	9 including 3 existing sites
Cyanobacteria identification and photograph (PTOX)	Surface	9 including 3 existing sites
Total anatoxin-producing cyanobacteria (qPCR)	Surface	9 including 3 existing sites
Total microcystin-producing cyanobacteria (qPCR)	Surface	9 including 3 existing sites

[1] Data available at 10-minute increments at the Lake Wohlford (LKWSD) site (https://mesowest.utah.edu/index.html).
[2] Chlorophyll-*a* and BGA-phycocyanin are also available as *in situ* parameters, depending on the instrument used. Collection of chlorophyll-a as in situ and grab sample measurements will support determination of an empirical relationship between the in situ and laboratory-analyzed parameters.

5.7.1.5 Monitoring methods

In situ water quality

Escondido will measure *in situ* water quality parameters (water temperature, dissolved oxygen, conductivity, pH, turbidity [may also include chlorophyll-*a* or BGA-phycocyanin]) as instantaneous daytime measurements using a pre-calibrated multi-probe sonde with designated sensors (Table 5-14). The sonde will be deployed from a boat and *in situ* measurements will be recorded at 1-ft depth intervals and within 0.5 ft from the bottom sediments. For the *in situ* diel measurements of water quality parameters (water temperature, dissolved oxygen, pH, turbidity [may also include chlorophyll-a and BGA-phycocyanin]), the sondes will be deployed approximately 0.5 ft above the sediment water interface for collection of continuous data (i.e., every 15 minutes) for a minimum of a 48-hour monitoring period at each site. Secchi depth readings will be collected at each reservoir site as a separate measure of transparency and to ascertain the approximate depth of the photic zone.

Table 5-14. *In situ* water quality methods for Lake Wohlford synoptic monitoring.

Parameter	Method	Units	MDL
Water temperature	EPA 170.1	degrees Celsius (ºC)	0.1
DO	SM 4500-O(G)	milligrams per liter (mg/L)	0.1
Conductivity	SM 2510-B	micro Siemens per centimeter (uS/cm)	1.0
pH	SM 4500-H	standard unit of pH (s.u.)	0.1
Turbidity (Nephelometric)	EPA 180.1; ISO 7027-1; SM 2130-B	Nephelometric Turbidity Unit (NTU)/Formazin Nephelometric Unit (FNU)	0.1
Chlorophyll-*a* and BGA-phycocyanin	Optical luminescence	Reflective Fluorescence Unit (RFU) or micrograms per liter (ug/L)	0.01
Secchi depth (Secchi disk)	N/A	meters (m)	0.1

DO = dissolved oxygen
EPA = Environmental Protection Agency
MDL = method detection limit
SM = Standard Methods

Analytical water quality

Escondido will analyze water quality constituents and cyanobacteria samples using methods presented in Table 5-15. Analytical samples will be collected as surface grab samples in the top 1 ft of the water column.

Table 5-15. Analytical water quality methods for Lake Wohlford synoptic monitoring.

Constituent	Method	MDL	Hold Time
Chlorophyll-*a*	EPA 445.0	0.5 ug/L	8 hours (refrigerated 4°C)*
Cyanobacteria identification and photograph (PTOX)	Microscopy	--	24 hours (refrigerated 4°C)
Total anatoxin-producing cyanobacteria (qPCR)	Molecular (quantitative polymerase chain reaction [qPCR])	1 copy/mL	4 days at 4°C
Total microcystin-producing cyanobacteria (qPCR)			

* Sample should be analyzed as soon as possible
EPA = Environmental Protection Agency
MDL = method detection limit

Quality assurance/Quality control

For instantaneous *in situ* water quality measurements, the sonde will be calibrated at the start of each sampling day and checked at the end of the day. For 48-hour deployments, the sondes will be calibrated before they are deployed and checked immediately after deployment. Calibration data will be recorded on a calibration log that includes measurement quality objective criteria for real-time comparison. All analytical samples will be collected, handled and delivered to the analytical laboratory consistent with standard methods (Table 5-5). Appropriate QA/QC methods and documentation will be followed. All sample bottles will be rinsed with water from the water body to be collected from. QA/QC in the field will be assured by accurate and thoroughly completed sample labels, field sheets, and chain of custody and sample log forms. Sample labels will include sample identification code, date, time, site name, sampling location, collector's name, sample type, and preservative, if applicable.

Data analysis and reporting

Monitoring data will be compiled and organized by seasonal event, parameter/constituent, and/or monitoring location. Figures and tables will be developed to display and evaluate spatial trends by season, with particular comparisons between shoreline sites and between open water sites to identify representative index sites and rapid response monitoring sites within those two groupings. A short technical memorandum will be developed to summarize the results of the monitoring and to inform monitoring discussed in Section 5.7.2.

5.7.1.6 Cost estimate

Estimated costs for synoptic monitoring in Lake Wohlford are summarized in Table 5-16 and assume that project management, field work, coordination with the analytical laboratory, data entry and QA/QC, and data analysis and reporting would be undertaken by a contractor.

Table 5-16. Estimated monitoring costs for Lake Wohlford synoptic monitoring to establish index and rapid response monitoring sites for a future new condition in Lake Wohlford.

Task	Labor Hours		Labor Cost (full)		Field Expenses Cost		Contract Laboratory Expenses [e]	Total Cost
	Contractor	Escondido Staff	Contractor	District Staff	Contractor	District Staff		
Task 1 Project Management [a]	9	12	$1,400	$1,500	$0	$0	$0	$2,900
Task 2 Seasonal Synoptic Field Surveys [b, c]	215	12	$25,400	$1,500	$14,400	$0	$4,700	$46,000
Task 3 Data Analysis and Reporting [d]	97	24	$12,500	$3,000	$0	$0	$0	$15,500
Total	**321**	**48**	**$39,300**	**$6,000**	**$14,400**	**$0**	**$4,700**	**$64,400**

[a] Assumes monthly invoicing and up to three as-needed progress meetings with Escondido PM conducted by teleconference. Meetings up to 0.5-hour each. Hourly labor estimates assume 2022 rates.

[b] Synoptic monitoring cost estimate assumes three sampling events; late spring/early summer (May/June), late summer/early fall (August/September), and winter (November/December). Cost estimate assumes contractor conducts field work and coordination with analytical laboratory. Hourly labor estimates assume 2022 rates.

[c] Contractor field expenses include travel, lodging, and monitoring equipment (i.e., three water quality sondes, Secchi disk), and sample shipping to the analytical laboratory. Costs assume that Escondido will provide the boat and outboard motor for field work.

[d] Cost estimate assumes contractor conducts data entry, QA/QC, data analysis, and reporting and is based on 2022 rates.

[e] Expenses include chlorophyll-a at $40/sample; cyanobacteria identification and photograph (PTOX) at $40/sample; qPCR total microcystin-producing cyanobacteria at $42/sample; qPCR total anatoxin-a-producing cyanobacteria at $42/sample.

5.7.1.7 Responsibilities to implement monitoring, analysis, and reporting

Escondido may elect to have the synoptic monitoring undertaken by a contractor. Analytical water quality samples collected during seasonal monitoring events will be shipped by Escondido or contractor field staff to the analytical laboratory or laboratories possessing the appropriate expertise. Overall data analysis and reporting may also be undertaken by the contractor at Escondido's direction.

5.7.2 Routine, rapid response, effectiveness, and permit-related monitoring associated with the use of algaecides as a short-term mitigation strategy

5.7.2.1 Monitoring goals and objectives

The goal of routine monitoring is to provide Escondido with evidence that a HAB may be developing in Lake Wohlford, while the goal of rapid response monitoring is to allow sufficient response time for the successful implementation of algaecide as a short-term HAB mitigation strategy. HAB mitigation success will involve using the least amount of algaecide necessary to interrupt cyanotoxin production in the lake, consistent with limitations dictated by the *Aquatic Pesticide Application Plan for Lakes Wohlford and Dixon and Associated Waterways* (SCS Engineers 2014) and Escondido's interest in minimizing the use of chemicals in the lake. See also Section 5.5.1.1 for a discussion of the need for rapid response monitoring.

The following monitoring objectives have been developed to meet the above goals:

- Conduct routine monitoring of anatoxin-a and microcystin in Lake Wohlford surface waters;
- Conduct once annual screening for cylindrospermopsin in Lake Wohlford surface waters16;
- Conduct rapid response monitoring, as needed, of anatoxin-a and microcystin in Lake Wohlford surface waters at more frequent intervals than under routine monitoring and at additional monitoring sites in the water treatment plant;
- Conduct algaecide effectiveness monitoring, as needed, following treatment; and,
- Conduct monitoring required under the Statewide Aquatic Weed Control Permit.

5.7.2.2 Monitoring area

The monitoring area for routine and rapid response monitoring is Lake Wohlford and, in the case of rapid response monitoring, in the water treatment plant.

5.7.2.3 Monitoring design

Monitoring sites

The location of routine and rapid response monitoring sites will be determined as described in Section 5.7.1. The number of routine monitoring sites is expected to range from 1–3 index sites. The number of rapid response monitoring sites will depend on the results of the monitoring described in Section 5.5.1 but is expected to range from 1–3 sites.

[16] To date, these cyanotoxins have not been detected in Lake Wohlford (Section 2.2.4).

Monitoring frequency, parameters, and operational triggers

Monitoring associated with the use of algaecides as a short-term mitigation strategy will include routine monitoring, rapid response monitoring, algaecide effectiveness monitoring, and monitoring required under the Statewide Aquatic Weed Control Permit (Table 5-15). Additional details are presented below for each type of monitoring, including the associated operational triggers.

Table 5-17. Water quality parameters to be monitored, monitoring frequency, and number of sites for each type of Lake Wohlford monitoring associated with the use of algaecides as a short- and long-term mitigation strategy.

Parameter	Type/ Location	Routine Monitoring		Rapid Response Monitoring				Algaecide Effectiveness Monitoring		Statewide Aquatic Weed Control Permit	
		Frequency in Lake	Number of Sites in Lake	Frequency in Lake	Number of Sites in Lake	Frequency in EVWTP	Number of Sites in EVWTP	Frequency in Lake	Number of Sites in Lake	Frequency in Lake	Number of Sites in Lake
Cyanobacteria surface scum/ accumulation	Visual	Monthly	1–3	Weekly	1–3	–	–	Weekly	1–3	–	–
In situ water temp., dissolved oxygen, conductivity, pH, turbidity)	Surface	–	–	–	–	–	–	–	–	Three times (BG, EM, PE) [c]	1
Total hardness ($CaCO_3$)	Surface grab	–	–	–	–	–	–	–	–	Three times (BG, EM, PE) [c]	1
Total copper [a]	Surface grab	–	–	–	–	–	–	–	–	Three times (BG, EM, PE) [c]	1
Total anatoxin-a	Surface grab	–	–	Weekly	1–3	Daily for 10 days	1–3 [b] (MR, SW, FE)	Weekly	1–3		
Total cylindro-spermopsins	Surface grab	Annually	1–3	As needed	As needed	As needed	As needed	As needed	As needed		
Total microcystins/ nodularins	Surface grab	Monthly	1–3	Weekly	1–3	Daily for 10 days	1–3 [b] (MR, SW, FE)	Weekly	1–3		

[a] If copper is the active ingredient in algaecide product.
[b] Samples will be collected at MR = mixed raw water location in the WTP. Sample collection at SW = settled water and FE = filter effluent water locations will be dependent on whether microcystin concentrations at MR exceed 0.3 ug/L or anatoxin-a at MR is detected (see also page 194).
[c] BG = background monitoring prior to treatment; EV = event monitoring during treatment; PE = post-event monitoring following treatment (see also page 194).

Routine monitoring

Routine monitoring will include the parameters listed in Table 5-15. Routing monitoring will occur at 1–3 index sites on a monthly basis unless otherwise noted in Table 5-15.

Triggers for Moving to Rapid Response Monitoring (see also Figure 5-8)

If anatoxin-a is detected in Lake Henshaw release water[17] *and water is being released from Lake Wohlford to the water treatment plant, or it is anticipated for release in the next two weeks, then Escondido will move to rapid response monitoring for anatoxin-a.*

If microcystin ≥ 0.3 ug/L in Lake Henshaw release water[17] *and water is being released from Lake Wohlford to the water treatment plant, or it is anticipated for release in the next two weeks, then Escondido will move to rapid response monitoring for microcystin.*

If microcystin ≥ 0.3 ug/L in Lake Wohlford and water is being released from Lake Wohlford to the water treatment plant, or it is anticipated for release in the next two weeks, then Escondido will move to rapid response monitoring for microcystin and anatoxin-a.

Note that if microcystin ≥ 0.3 ug/L in Lake Wohlford, then in addition to moving to rapid response monitoring, Escondido will also begin algaecide treatment in the lake (see below).

Knowledge gained during previous monitoring periods should be used to evaluate whether monitoring goals and objectives are being met through the Lake Wohlford water quality monitoring program, and this evaluation may result in recommendations for revisions to the above routine monitoring parameters and operational triggers (see also Section 5.1).

Rapid response monitoring

Rapid response monitoring will include the parameters listed in Table 5-15 and will occur in Lake Wohlford and in the water treatment plant.

In-lake rapid response monitoring will occur at rapid response monitoring sites for the parameters in Table 5-15 within 1−3 days of the rapid response monitoring trigger. Rapid response monitoring at Lake Wohlford will occur weekly.

Rapid response monitoring in the water treatment plant will occur in the mixed raw water, settled water, and filter effluent water for the parameters in Table 5-15 within 1−3 days of the rapid response monitoring trigger. Rapid response monitoring in the water treatment plant will occur daily for 10 days. At the end of the 10 days, if cyanotoxin concentrations are not detected in the filter effluent water, then rapid response monitoring can cease. If cyanotoxin concentrations are detected in the filter effluent water on days 8, 9, or 10, then rapid response monitoring in Lake Wohlford will continue on a weekly basis and rapid response monitoring in the water treatment plant will continue daily until concentrations are not detected for 3 days in a row. If at any time microcystin or anatoxin-a are detected in the filter effluent sample, then treatment of Lake Wohlford water will cease.

[17] Vista Irrigation District currently conducts cyanotoxin monitoring of the Lake Henshaw release water on a weekly basis (see also Section 5.4.2).

Triggers for Moving to Algaecide Treatment (see also Figure 5-8)

The trigger for moving to algaecide treatment is if a sample at one or more index sites or rapid response monitoring sites exhibits either of the below:

- *Detection of anatoxin-a*
- *Microcystin concentration ≥ 0.3 ug/L*

Algaecide treatment will occur in the vicinity of the rapid response monitoring site(s) exhibiting elevated toxin concentrations and, as needed, in other locations that exhibit cyanobacteria surface scums/accumulations. Escondido currently has sufficient experience with algaecide dosing in Lake Wohlford such that the amount of cyanotoxin concentration at one or more index sites is sufficient information to set the effective algaecide dose prior to treatment. *Algaecide treatment will occur within 1 week of the trigger.* Treatment will occur consistent with the final APAP for Lake Wohlford and Lake Dixon (SCS Engineers 2014).

Once the new dam is in place, and Lake Wohlford storage has returned to 6,800 AF, additional as-needed monitoring will be conducted as part of rapid response monitoring prior to algaecide application. The additional monitoring may include *in situ* dissolved oxygen and total and/or cyanobacteria cell counts (cells/mL), including identification of potential toxin producing species, at any rapid response monitoring site where the trigger occurs as soon as possible (i.e., next day if using rapid screening methods for cyanotoxins, or within 2−3 days if using laboratory methods). *In situ* dissolved oxygen measurements and total cell counts would be collected at rapid response monitoring sites to indicate whether the HAB is too dense and too widespread to be effectively treated with an algaecide. If dissolved oxygen measured before 9 am at is generally less than 5 mg/L in surface waters and/or less than 2 mg/L in bottom waters, and if total cell counts are greater than $1x10^6$ cells/mL at multiple rapid response monitoring sites, then the HAB is likely past the effective treatment window (see Figure 5-1) and/or a fish kill is possible. In this case, algaecide treatment will not occur, water from Lake Wohlford will not be released to the water treatment plant, and Escondido will return to rapid response monitoring in the lake to track progression of the HAB. Note that cyanobacteria cell counts provide valuable information regarding the intensity of a cyanobacteria bloom, although they do not indicate the intensity of other phytoplankton (e.g., green algae, diatoms, golden algae) that also may be present and affecting water quality in the lake.

Algaecide effectiveness monitoring

Following algaecide treatment, algaecide effectiveness monitoring will include the parameters listed in Table 5-15. Unless otherwise noted in Table 5-15, algaecide effectiveness monitoring will be conducted at all rapid response monitoring sites and for all rapid response monitoring parameters to determine the response of the HAB. Algaecide effectiveness monitoring will begin immediately following algaecide application and will occur on a weekly basis until (see also Figure 5-8):

- *Anatoxin-a is not detected, and*
- *Microcystin is < 0.3 ug/L.*

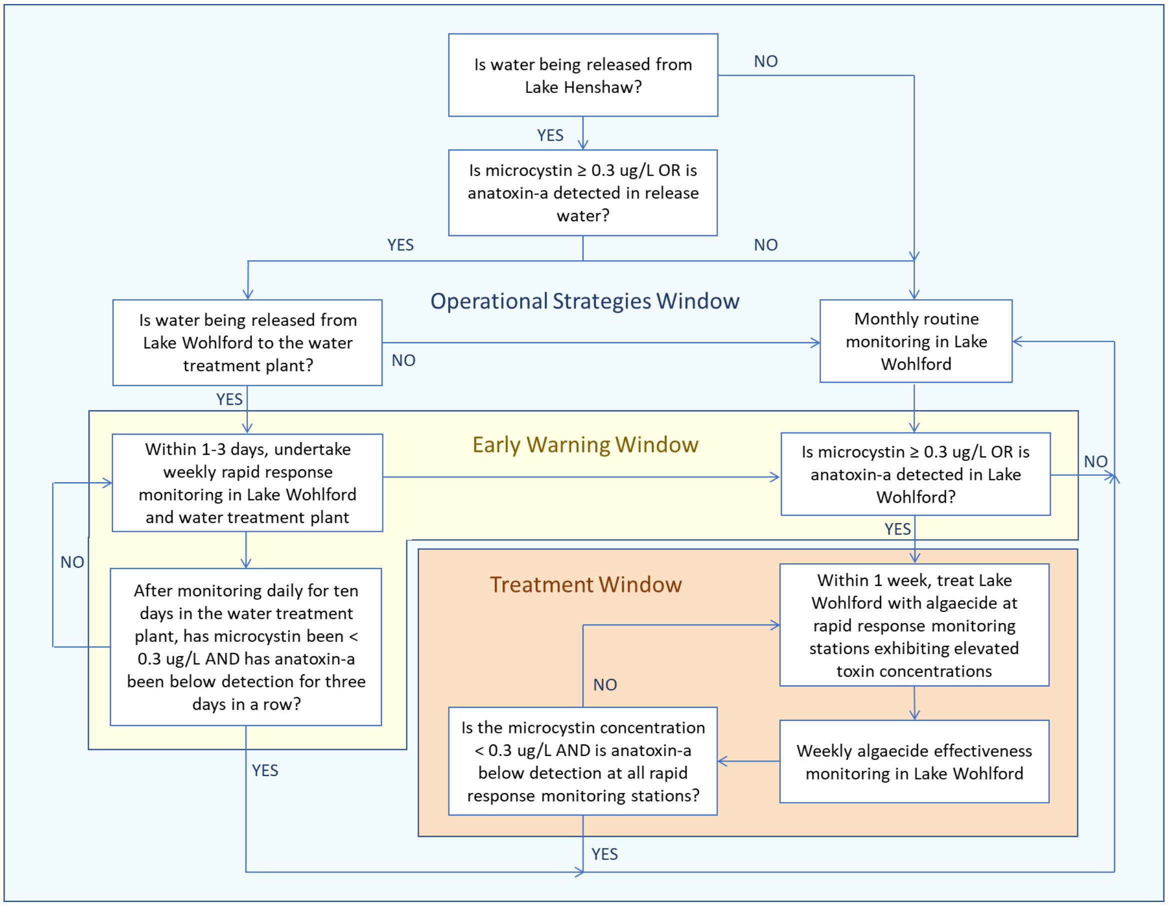


Figure 5-8. Cyanotoxin routine, rapid response, and algaecide effectiveness monitoring framework and operational triggers flowchart for Lake Wohlford. Operational strategies window (blue), early warning window (yellow), and treatment window (orange) are discussed further in Section 5.1.

Monitoring required under the Statewide Aquatic Weed Control Permit

Coverage under the California Statewide General National Pollutant Discharge Elimination System (NPDES) Permit for Residual Aquatic Pesticide Discharges to Waters of the United States from Algae and Aquatic Weed Control Applications Permit # CAG990005 requires that certain monitoring activities be conducted in association with algaecide application to control nuisance blooms. The *Aquatic Pesticide Application Plan for Lakes Wohlford and Dixon and Associated Waterways* (SCS Engineers 2014) includes a Monitoring and Reporting Program for Lake Wohlford that is compliant with NPDES requirements.

Prior to, during, and following algaecide treatment, monitoring required for the Monitoring and Reporting Program will include the parameters listed in Table 5-15. The three types of required monitoring for each treatment event include background monitoring (BG) in the treatment area just prior to the treatment event (i.e., up to 24 hours in advance of treatment); event monitoring (EM) proximally adjacent to the treatment location, in the portion of the treatment area that is exposed to the algaecide plume immediately or shortly after the treatment event; and post-event monitoring (PE) adjacent to the treatment area within one week after the treatment event. One full set of three samples (i.e., BG, EM and PE samples) will be collected during each treatment event along with quality control samples.

Monitoring and Reporting Program details regarding monitoring procedures for in-field and grab sampling, laboratory analyses, monitoring records, calibration and maintenance of instrumentation, maximum allowable copper concentrations in Lake Wohlford following treatment (if copper is the active ingredient in the applied algaecide), and reporting are presented in Appendix C.

5.7.2.4 Monitoring methods

Analytical water quality

Escondido will collect analytical water quality constituents using methods presented in Table 5-18. Analytical samples will be collected as grab samples in the top 1 ft of the water column (i.e., at the surface).

Table 5-18. Analytical water quality methods for algaecide-related monitoring in Lake Wohlford.

Constituent	Method	MDL	Hold Time
Total microcystins/nodularins	ELISA or LC/MS	0.041 ug/L	4 days (refrigerated 4°C)
Total anatoxin-a	ELISA or LC/MS	0.03 ug/L [a]	4 days (refrigerated 4°C)
Total cylindrospermopsins	ELISA or LC/MS	0.09 ug/L [a]	4 days (refrigerated 4°C)

MDL = method detection limit

[a] Analytical laboratory provides only a method reporting limit (MRL) for this constituent.

Quality assurance/Quality control

All analytical samples will be collected, handled and delivered to the analytical laboratory consistent with standard methods (Table 5-18). Appropriate QA/QC methods and documentation will be followed. All sample bottles will be rinsed with water from the water body to be collected from. QA/QC in the field will be assured by accurate and thoroughly completed sample labels, field sheets, and chain of custody and sample log forms. Sample labels will include sample identification code, date, time, site name, sampling location, collector’s name, sample type, and preservative, if applicable.

Data analysis and reporting

Monitoring data will be compiled and organized by monitoring event, parameter/constituent, and/or monitoring location. Figures and tables will be developed to display and evaluate spatial and temporal trends, with particular comparisons between rapid response monitoring and algaecide effectiveness monitoring. Monitoring data will be analyzed and interpreted at minimum on an annual basis (see also Section 5.1).

5.7.2.5 Cost estimate

Estimated costs for monitoring associated with the use of algaecides as a short-term mitigation strategy are summarized in Table 5-19 and assume that project management, field work, coordination with the analytical laboratory, data entry and QA/QC, and data analysis would be undertaken by Escondido.

Table 5-19. Estimated annual monitoring costs for monitoring associated with the use of algaecides as a short-term mitigation strategy in Lake Wohlford.

Task	Labor Hours		Labor Cost (full)		Field Expenses Cost		Contract Laboratory Expenses [f]	Total Cost
	Contractor	District Staff	Contractor	District Staff	Rental Equipment [e]	District Staff		
Task 1 Routine Monitoring [a]	0	208	$0	$26,000	$0	$0	$12,800	$38,800
Task 2 Rapid Response Monitoring [b]	0	48	$0	$6,000	$0	$0	$9,500	$15,500
Task 3 Algaecide Effectiveness Monitoring [c]	0	64	$0	$8,000	$0	$0	$3,200	$11,200
Task 4 Statewide Aquatic Weed Control Permit Monitoring [d]	Costs included in algaecide application contract							
Total	**0**	**320**	**$0**	**$40,000**	**$0**	**$0**	**$25,500**	**$65,500**

[a] Assumes monthly monitoring at each of three index monitoring sites. Hourly labor estimates assume 2022 rates.
[b] Assumes two rapid response monitoring events per year. Each event includes sample collection at 5 monitoring sites, 3x per event, with samples collected every 1–3 days. Hourly labor estimates assume 2022 rates.
[c] Assumes two post-algaecide treatment monitoring events per year. Each event includes sample collection at five monitoring sites, one time per event, with samples collected once per week. Hourly labor estimates assume 2022 rates.
[d] Assumes contractor conducts algaecide application and required monitoring and reporting for the Statewide aquatic weed control permit.
[e] Assumes no equipment rental expenses for cyanotoxin monitoring.
[f] Expenses include ELISA total microcystin-producing cyanobacteria at $105/sample; ELISA total anatoxin-a-producing cyanobacteria at $130/sample; ELISA total cylindrospermopsin-producing cyanobacteria at $130/sample.

5.7.2.6 Responsibilities to implement monitoring, analysis, and reporting

In situ monitoring and grab sample collection associated with the use of algaecides as a short-term mitigation strategy will be undertaken by Escondido staff. Analytical water quality samples will be shipped by Escondido staff to one or more laboratories that possess the appropriate expertise. Overall data analysis and reporting would also be undertaken by Escondido, with consultant support at Escondido's discretion.

6 REFERENCES

Anderson, M.A. 2002. Impacts of alum addition on water quality in Lake Elsinore. Final report to Lake Elsinore and San Jacinto Watersheds Authority. http://www.mywatersheds.com/the-watershed/lesjwas-approach/additional-watershed-studies/

Afsar, A., and S. Groves. 2009. Eco-toxicity assessment of Phoslock. PWS Report Number TR 022/09. Phoslock Water Solutions, Ltd. http://www.phoslock.eu/media/7407/Eco-toxicity-Assessment-Report-May-2009.pdf

Beutel, M.W., 2000a. Dynamics and control of nutrient, metal and oxygen fluxes at the profundal sediment-water interface of lakes and reservoirs. Doctoral dissertation, University of California, Berkeley.

Beutel, M.W., 2000b. Lake Elsinore Sediment-water Interface Study. Report to Montgomery Watson and Alex Horne Associates, October 2000.

Beutel, M.W. and S. Defeo. 2021. Lake Henshaw and Lake Wohlford Sediment Flux Study. Report to Stillwater Sciences, December 2021.

Black and Veatch. 2013. Draft Lake Wohlford Dam Replacement Tower Selection Summary. Prepared for City of Escondido, February 12, 2013.

Carlson, R.E. 1977. A trophic state index for lakes. Limnology and Oceanography. 22:361-369.

California Water Quality Monitoring Council. 2022a. California HABs portal. https://fhab.sfei.org/

California Water Quality Monitoring Council. 2022b. My Water Quality: California Harmful Algal Blooms (HABs). https://mywaterquality.ca.gov/habs/resources/habs_response.html

CNDDB (California Natural Diversity Database). 2022. California Department of Fish and Game, Biogeographic Data Branch, Sacramento. http://www.dfg.ca.gov/whdab/html/cnddb.html

CH2MHill. 2004. Lake Elsinore nutrient removal study. Final Report. Prepared for Lake Elsinore & San Jacinto Watersheds Authority. http://www.mywatersheds.com/the-watershed/lesjwas-approach/additional-watershed-studies/

Cooke, G. D., E. B. Welch, and S. Nichols. 2005. Restoration and management of lakes and reservoirs, Third edition. Taylor and Francis Group. New York, New York. 591 p.

EPA (U.S. Environmental Protection Agency). 1978. Report on Lake Henshaw, San Diego County, California, EPA Region IX. Working Paper No. 748. Corvallis Environmental Research Laboratory, Corvallis, Oregon and Environmental Monitoring and Support Laboratory, Las Vegas, Nevada.

EPA (U.S. Environmental Protection Agency). 2012. Nitrates. Website https://archive.epa.gov/water/archive/web/html/vms57.html. [Accessed April 2022].

EPA. 2021. Cyanobacteria Assessment Network Application (CyAN app). https://www.epa.gov/water-research/cyanobacteria-assessment-network-application-cyan-app

Fastner, J., U. Neumann, B. Wirsing, J. Weckesser, C. Wiedner, B. Nixdorf, and I. Chorus. 1999. Microcystins (hepatotoxic heptapeptides) in german fresh water bodies. Environ Toxicol 14: 13–22.

Forsberg C., and S. O. Ryding. 1980. Eutrophication parameters and trophic state indices in 30 Swedish waste-receiving lakes. Arch Hydrobiol 89: 189–207.

Groundwater, T., and Dudek. 2018. Groundwater Modeling Report on Warner Ranch Basin.

Hellström, T. 1996. An empirical study of nitrogen dynamics in lakes. Water Environment Research 68: 3-125.

Herrmann, H., J. Nolde, S. Berger, and S. Heise. 2016. Aquatic ecotoxicity of lanthanum – a review and an attempt to derive water and sediment quality criteria. Ecotoxicology and Environmental Safety. 124: 213–238.

Horne, A. J., and C. R. Goldman. 1994. Limnology. McGraw-Hill, Inc.

Lürling, M., and Y. Tolman. 2010. Effects of lanthanum and lanthanum-modified clay on growth, survival and reproduction of *Daphnia magna*. Water Research 44: 309–319.

Marine Biochemists. 2021. Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch. Prepared by Marine Biochemists, Anaheim, California for Vista Irrigation District, Vista, California.

Marine Biochemists. 2021. Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch. Anaheim, California.

MWH. 2016. Watershed Sanitary Survey Update, Escondido/Vista Irrigation District.

Nurnberg, G. K. 1994. Phosphorus release from anoxic sediments: What we know and how we can deal with it. Limnetica 10: 1–4.

RWQCB (California Regional Water Quality Control Board). 2016. Water Quality Control Plan for the San Diego Basin. Prepared by RWQCB, San Diego Region, California.

San Diego Regional Water Quality Control Board. 2021. Water Quality Control Plan for the San Diego Basin. San Diego, California. https://www.waterboards.ca.gov/sandiego/water_issues/programs/basin_plan/

SCS Engineers. 2014. Aquatic Pesticide Application Plan. Lakes Wohlford and Dixon and associated waterways, City of Escondido, Escondido, California. Prepared by SCS Engineers, San Diego, California for State Water Resources Control Board, Sacramento, California.

Smith, D. 2021. Site Visit Notes Clarification.

Stillwater Sciences and Brown and Caldwell. 2016. Key factors influencing the production of taste & odor compounds and soluble manganese in San Pablo Reservoir. Prepared by Stillwater Sciences, Berkeley, California, with assistance from Brown and Caldwell, Walnut Creek, California; Dr. Marc Buetel, University of California, Merced; and, Dr. William Taylor, Walnut Creek, California for East Bay Municipal Water District, Oakland, California.

Todd Groundwater and Dudek. 2018. Warner Valley Basin Groundwater Flow Model Development and Calibration. Final. Prepared for Vista Irrigation District, Vista, California.

University of Utah. 2022. MesoWest. https://mesowest.utah.edu/

USFWS. 2022. Information for Planning and Conservation (IPaC) portal. https://ipac.ecosphere.fws.gov/

Vista Irrigation District. 2019. Regular Meeting of the Board Directors, 2-6-2019. Vista Irrigation District.

Vista Irrigation District. 2021. Site Visit Interview.

Welch, E. B., and H. Gibbons. 2005. World-wide use of alum in lake restoration. Presented at the North American Lake Management Conference, Madison, Wisconsin, November.

World Health Organization (WHO). 2020a. Cyanobacterial toxins: anatoxin-a and analogues. Background document for development of WHO Guidelines for drinking-water quality and Guidelines for safe recreational water environments. Geneva, Switzerland. WHO/HEP/ECH/WSH/2020.1

World Health Organization (WHO). 2020b. Cyanobacterial toxins: microcystins. Background document for development of WHO Guidelines for drinking-water quality and Guidelines for safe recreational water environments. Geneva, Switzerland. WHO/HEP/ECH/WSH/2020.6

Zeise, L. 2021. Recommendations for interim notification levels for saxitoxins, microcystins, and cylindrospermopsin. OEHHA memorandum to D. Polhemus. Deputy Director, Division of Drinking Water State Water Resources Control Board.

Appendices

Appendix A

Warner Wellfield Loading Calculations

Loading Calculations from Wellfield into Lake Henshaw
Submitted 1/14/2022

Legend

Multiple sample dates in month, monthly average was used

No wellfield production, groundwater concentration not contributing to Lake Henshaw

[1] The concentration was averaged for same-day sampling events across all District wells

Nitrate as Nitrogen

Well Sampling Date	Month End Date	Wellfield Production (AF/month)	Wellfield Production (MG/month)	Ave. NO_3 Concentration (mg/L)[1]	NO_3 Loading (lb/month)	NO_3 Loading (kg/month)
2/13/1951	2/28/1951	*No Data*		0.1	*N/A*	
9/2/1954	9/30/1954	1085.5	353.7	0.2	729.6	330.9
8/5/1955	8/31/1955	875.3	285.2	1.1	2634.7	1195.1
9/13/1956	9/30/1956	836.5	272.6	0.5	1077.5	488.7
3/28/1957	3/31/1957	927.0	302.1	0.0	0.0	0.0
5/18/1957	5/31/1957	1368.8	446.0	0.0	0.0	0.0
6/26/1957	6/30/1957	1625.4	529.6	0.3	1148.4	520.9
12/1/1957	12/31/1957	1759.5	573.3	0.2	963.5	437.0
12/16/1957	12/31/1957	1759.5	573.3	0.2	See Above	
7/13/1959	7/31/1959	1003.0	326.8	0.5	1362.8	618.2
2/2/1960	2/29/1960	849.8	276.9	0.5	1154.7	523.8
1/4/1961	1/31/1961	825.8	269.1	0.1	157.1	71.3
4/27/1962	4/30/1962	961.9	313.4	1.5	3795.5	1721.6
10/3/1963	10/31/1963	894.2	291.4	0.4	877.2	397.9
11/20/1963	11/30/1963	804.4	262.1	0.8	1777.2	806.1
12/3/1964	12/31/1964	679.4	221.4	1.0	1877.7	851.7
1/1/1968	1/31/1968	18.6	6.1	0.3	17.1	7.8
5/22/1968	5/31/1968	741.9	241.7	1.2	2505.4	1136.4
9/25/1968	9/30/1968	987.0	321.6	0.6	1636.1	742.1
5/21/1969	5/31/1969	0.0	0.0	4.1	0.0	0.0
1/27/1970	1/31/1970	0.0	0.0	0.5	0.0	0.0
7/29/1970	7/31/1970	0.0	0.0	0.7	0.0	0.0
12/24/1970	12/31/1970	965.3	314.5	0.6	1482.1	672.3
1/6/1971	1/31/1971	961.2	313.2	0.3	762.4	345.8
1/27/1971	1/31/1971	961.2	313.2	0.2	See Above	
1/1/1984	1/31/1984	2.2	0.7	3.6	21.2	9.6
7/1/1984	7/31/1984	0.0	0.0	4.0	0.0	0.0
1/1/1985	1/31/1985	0.0	0.0	3.9	0.0	0.0
7/1/1985	7/31/1985	653.1	212.8	3.4	6034.8	2737.4
1/1/1986	1/31/1986	1126.8	367.2	1.2	3777.6	1713.5
7/1/1986	7/31/1986	1153.3	375.8	3.6	11282.7	5117.8
1/1/1987	1/31/1987	1091.5	355.7	1.3	3930.1	1782.7
7/1/1987	7/31/1987	962.1	313.5	6.6	17255.7	7827.2
1/1/1988	1/31/1988	981.8	319.9	9.7	25880.0	11739.2
5/22/1989	5/31/1989	1774.3	578.1	5.7	27668.9	12550.6
10/17/1989	10/31/1989	1603.0	522.3	7.0	30392.5	13786.0
5/15/1990	5/31/1990	1267.5	413.0	7.8	26840.0	12174.6
10/30/1990	10/31/1990	1120.9	365.2	6.4	19467.0	8830.2
5/20/1991	5/31/1991	1302.8	424.5	6.0	21318.0	9669.8
10/28/1991	10/31/1991	1444.9	470.8	5.9	23045.6	10453.5
11/3/1992	11/30/1992	776.1	252.9	5.4	11459.3	5198.0
5/25/1993	5/31/1993	0.0	0.0	3.7	0.0	0.0
11/14/1994	11/30/1994	0.0	0.0	4.8	0.0	0.0

6/19/1995	6/30/1995	0.0	0.0	6.1	0.0	0.0
10/30/1995	10/31/1995	0.0	0.0	4.3	0.0	0.0
1/1/1996	1/31/1996	0.0	0.0	6.4	0.0	0.0
5/13/2002	5/31/2002	1066.3	347.4	1.3	3844.2	1743.7
10/22/2002	10/31/2002	1024.3	333.8	1.2	3401.5	1542.9
5/20/2003	5/31/2003	808.3	263.4	1.4	3150.5	1429.1
11/6/2003	11/30/2003	662.1	215.7	1.6	2815.8	1277.3
5/11/2004	5/31/2004	627.2	204.4	0.9	1575.4	714.6
6/24/2004	6/30/2004	595.1	193.9	0.7	2826.0	1281.9
6/28/2004	6/30/2004	595.1	193.9	0.9	See Above	
6/29/2004	6/30/2004	595.1	193.9	1.6		
10/26/2004	10/31/2004	437.4	142.5	1.1	1285.4	583.1
5/24/2005	5/31/2005	21.3	6.9	0.8	46.4	21.0
11/1/2005	11/30/2005	40.0	13.0	0.8	82.4	37.4
5/9/2006	5/31/2006	41.3	13.5	0.8	87.1	39.5
10/24/2006	10/31/2006	11.2	3.6	0.9	28.7	13.0
7/9/2007	7/31/2007	1251.8	407.9	1.2	3967.3	1799.6
12/19/2007	12/31/2007	1070.6	348.8	1.2	3589.4	1628.2
6/10/2008	6/30/2008	947.9	308.9	1.2	3164.7	1435.5
10/20/2008	10/31/2008	901.3	293.7	1.2	2920.8	1324.9
5/11/2009	5/31/2009	694.6	226.3	1.2	2236.8	1014.6
12/9/2009	12/31/2009	712.8	232.3	1.1	2144.6	972.8
5/11/2010	5/31/2010	676.9	220.6	1.0	1876.3	851.1
12/6/2010	12/31/2010	595.2	193.9	1.1	1813.9	822.8
5/10/2011	5/31/2011	8.7	2.8	0.6	14.0	6.4
11/3/2011	11/30/2011	8.6	2.8	0.7	16.8	7.6
6/25/2012	6/30/2012	864.3	281.6	1.1	2654.1	1203.9
12/18/2012	12/31/2012	907.0	295.5	1.3	3144.0	1426.1
6/5/2013	6/30/2013	733.3	238.9	1.3	2524.1	1144.9
12/4/2013	12/31/2013	776.9	253.1	1.3	2715.4	1231.7
4/21/2014	4/30/2014	724.2	236.0	0.8	2401.0	1089.1
4/22/2014	4/30/2014	724.2	236.0	1.3	See Above	
6/18/2014	6/30/2014	633.9	206.5	1.3	2288.9	1038.3
12/19/2014	12/31/2014	696.7	227.0	1.4	2679.0	1215.2
6/18/2015	6/30/2015	667.5	217.5	1.4	2547.6	1155.6
12/21/2015	12/31/2015	627.3	204.4	1.5	2513.4	1140.1
5/31/2016	5/31/2016	566.1	184.4	1.5	2282.4	1035.3
12/6/2016	12/31/2016	493.3	160.7	1.4	1826.1	828.3
6/27/2017	6/30/2017	16.8	5.5	0.5	23.4	10.6
12/6/2017	12/31/2017	9.5	3.1	0.5	14.2	6.4
6/27/2018	6/30/2018	536.9	174.9	1.0	1477.8	670.3
1/22/2019	1/31/2019	506.7	165.1	1.0	1308.0	593.3
6/4/2019	6/30/2019	5.3	1.7	0.7	8.0	3.6
6/24/2019	6/30/2019	5.3	1.7	0.0	See Above	
12/17/2019	12/31/2019	5.4	1.8	0.8	12.2	5.5

Conversion	
8.34	lb/gal
3.069	AF/MG
0.4536	kg/lb

All Years	Loading (kg/month)	**Yrs 2000-2019**	Loading (kg/month)
Minimum	0.00	Minimum	3.62
Maximum	13786.04	Maximum	1799.57
Average	2067.65	Average	851.19
Sample Size, n	81	Sample Size, n	38

OrthoPhosphate

Well Sampling Date	Month End Date	Wellfield Production (AF/month)	Wellfield Production (MG/month)	Ave. OrthoP Concentration (mg/L) [1]	OrthoP Loading (lb/month)	OrthoP Loading (kg/month)
6/24/2004	6/30/2004	595.10	193.907	0.048	58.76	26.65
6/28/2004	6/30/2004	595.10	193.907	0.012	See Above	
6/29/2004	6/30/2004	595.10	193.907	0.049		
4/21/2014	4/30/2014	724.20	235.973	0.055	150.88	68.44
4/22/2014	4/30/2014	724.20	235.973	0.088	See Above	
6/4/2019	6/30/2019	5.34	1.740	0.049	0.70	0.32

Iron

Well Sampling Date	Month End Date	Wellfield Production (AF/month)	Wellfield Production (MG/month)	Ave. Fe Concentration (mg/L) [1]	Fe Loading (lb/month)	Fe Loading (kg/month)
3/28/1957	3/31/1957	927.00	302.053	0.000	0	0.00
5/18/1957	5/31/1957	1368.80	446.008	0.000	0	0.00
12/1/1957	12/31/1957	1759.50	573.314	0.000	0	0.00
10/3/1963	10/31/1963	894.20	291.365	0.000	0	0.00
1/1/1968	1/31/1968	18.60	6.061	0.000	0	0.00
5/22/1968	5/31/1968	741.90	241.740	0.000	0	0.00
1/27/1970	1/31/1970	0.00	0.000	0.000	0	0.00
7/29/1970	7/31/1970	0.00	0.000	360.000	0	0.00
12/24/1970	12/31/1970	965.30	314.532	0.000	0	0.00
1/6/1971	1/31/1971	961.20	313.196	65.000	383.10	173.78
1/27/1971	1/31/1971	961.20	313.196	310.000	See Above	
6/24/2004	6/30/2004	595.10	193.907	6.400	6.79	3.08
6/28/2004	6/30/2004	595.10	193.907	2.000	See Above	
6/29/2004	6/30/2004	595.10	193.907	6.400		
4/21/2014	4/30/2014	724.20	235.973	7.300	10.30	4.67
4/22/2014	4/30/2014	724.20	235.973	4.200	See Above	
6/4/2019	6/30/2019	5.34	1.740	14.950	0.22	0.10

Manganese

Well Sampling Date	Month End Date	Wellfield Production (AF/month)	Wellfield Production (MG/month)	Ave. Mn Concentration (mg/L) [1]	Mn Loading (lb/month)	Mn Loading (kg/month)
2/13/1951	2/28/1951	*No Data*		25.000	*N/A*	
3/28/1957	3/31/1957	927.00	302.053	0.000	0	0.00
12/1/1957	12/31/1957	1759.50	573.314	0.000	0	0.00
10/3/1963	10/31/1963	894.20	291.365	0.000	0	0.00
1/1/1968	1/31/1968	18.60	6.061	0.000	0	0.00
5/22/1968	5/31/1968	741.90	241.740	0.000	0	0.00
1/27/1970	1/31/1970	0.00	0.000	0.000	0	0.00
7/29/1970	7/31/1970	0.00	0.000	0.000	0	0.00
12/24/1970	12/31/1970	965.30	314.532	0.000	0	0.00
1/6/1971	1/31/1971	961.20	313.196	0.000	0.00	0.00
1/27/1971	1/31/1971	961.20	313.196	0.000	See Above	
6/24/2004	6/30/2004	595.10	193.907	0.200	0.37	0.17
6/28/2004	6/30/2004	595.10	193.907	0.290	See Above	
6/29/2004	6/30/2004	595.10	193.907	0.130		
4/21/2014	4/30/2014	724.20	235.973	0.400	0.79	0.36
4/22/2014	4/30/2014	724.20	235.973	0.400	See Above	
6/4/2019	6/30/2019	5.34	1.740	1.790	0.03	0.01

Appendix B

Assessment of March 2022 Algaecide Treatment Effectiveness for Lake Henshaw

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TECHNICAL MEMORANDUM

DATE:	April 28, 2022
TO:	Don Smith, Vista Irrigation District
FROM:	Maia Singer, Avi Kertesz, and Wayne Swaney, Stillwater Sciences
SUBJECT:	Assessment of March 2022 algaecide treatment effectiveness for Lake Henshaw

1 INTRODUCTION

In March 2020, the Vista Irrigation District (District) began monitoring for the presence of cyanobacteria and cyanotoxins in Lake Henshaw after being alerted to the potential presence of harmful algal blooms (HABs) in the lake by remote sensing data. Since then, routine monitoring and laboratory analysis have confirmed the presence of elevated levels of the cyanotoxins microcystin and anatoxin-a at multiple sites in the lake and in water released to the downstream San Luis Rey River.

The District is currently developing a Draft HABs Management and Mitigation Plan, which outlines protocols for identifying early HAB development and actions that can be taken to minimize cyanotoxin production and associated delays to water deliveries in the short term, while longer-term alternatives are developed and implemented to prevent future blooms. As part of Draft HABs Management and Mitigation Plan development, application of copper- and/or peroxide-based algaecides has been identified as the most feasible short-term HABs control method for Lake Henshaw for the following reasons:

- Algaecide application is a well-proven mitigation method for HABs. Approved algaecide chemicals act quickly (i.e., minutes to hours) and can prevent the formation of and interrupt an ongoing HAB and to stop cyanotoxin production.
- Little to no capital investment is required for algaecide application, since licensed applicators can be hired by the District to apply the chemicals and undertake monitoring needed to meet permit requirements.
- Costs are generally predictable and there are multiple algaecide products available on the market.

In June 2021, the District obtained a Statewide Aquatic Weed Control Permit for application of copper sulfate, chelated copper, and sodium carbonate peroxyhydrate (peroxide) to control HABs in Lake Henshaw. The District desires to obtain experience with the use of both copper- and peroxide-based algaecides in the lake over time.

Throughout 2021, persistent cyanotoxin concentrations above the California Cyanobacteria Harmful Algal Bloom (CCHAB) Network "caution" thresholds (i.e., 0.8 µg/L and detection for microcystin and anatoxin-a, respectively) in Lake Henshaw hindered the District's ability to meet water delivery obligations to groups represented by the San Luis Rey Indian Water Authority (IWA), including the La Jolla and Rincon Bands, and the City of Escondido. Cyanotoxin concentrations in the lake dropped below the CCHAB caution thresholds in early 2022 and the District subsequently released water from Henshaw Dam. However, persistent low-level microcystin concentrations (<0.5 µg/L) and several subsequent anatoxin-a detections both in Lake Henshaw and at downstream sampling sites in the San Luis Rey River prompted the District to initiate the first algaecide treatment of Lake Henshaw to assess lake response.

In accordance with the State Water Board approved *Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch* (Marine Biochemists 2021), the District applied 40,000 pounds of SePRO PAK 27 (active ingredient sodium carbonate peroxyhydrate 85%) to Lake Henshaw on March 14 and 15, 2022. The majority of the lake was treated on March 14, with one application boat launching at approximately 8:30 AM and heading toward the southern portion of the lake, and a second boat launching at 9:00 AM and heading toward northern portion of the lake. Treatment ended on March 14 at approximately 6:00 PM. The northernmost and southernmost portions of the lake were treated on March 15 between the hours of 8:30 AM and 1:30 PM. Over the course of two days, the entire lake surface area (approximately 728 acres) was treated with SePRO PAK 27.

The Lake Henshaw boat applicator tracks were spaced approximately 100 feet apart and the treatment boom width was approximately 40 feet across, such that approximately 40% of the lake surface area was treated on March 14 and 15, 2022. The resulting dose of hydrogen peroxide in the 40% of lake surface area that was treated, assuming an average 5-foot water depth, was approximately 2.9 mg/L (ppm). Averaged across the entire lake surface, the hydrogen peroxide dose was 1.1 mg/L, although the latter estimate assumes complete mixing immediately following dosing, which is unlikely to have occurred.

This technical memorandum provides an assessment of the effectiveness of this first algaecide treatment in Lake Henshaw. The methodology, results, and conclusions of the water quality monitoring effort associated with the algaecide treatment are described below.

2 METHODS

To inform the assessment of algaecide treatment effectiveness, the District expanded routine (i.e., weekly) water quality monitoring at four Lake Henshaw sites (H-S, H-FD, H-BL, H-BLS) to include eight additional open water and shoreline sites (Table 1), as well as *in situ* water quality monitoring parameters and additional analytical constituents before and after the treatment event.

In situ water quality parameters included water temperature, dissolved oxygen (DO), conductivity, total dissolved solids, pH, oxidation reduction potential (ORP), and turbidity. *In situ* measurements were taken in the morning (between approximately 9 am and 11 am) and the afternoon (between approximately 12 pm and 4 pm) at all sites on 3/14, 3/15, 3/16, 3/17, 3/18, and 3/22/2022 and were made with a calibrated YSI DSS multiprobe.

Chlorophyll-a, pheophytin-a, and nutrients (total nitrogen, nitrate, ammonia, total phosphorus, orthophosphate) were sampled in the morning (between approximately 7 am and 11 am) at all sites on 2/28, 3/7, 3/14, 3/16, 3/18, and 3/22, except sites H-FD, H-FDD, and H-MLD on 2/28 and 3/14/2022. In addition, chlorophyll-a, pheophytin-a, and nutrients were sampled on the morning of 4/4/2022 at H-S, H-FD, H-BL, and H-BLS. Grab samples were shipped overnight to the analytical laboratory (Bend Genetics, Sacramento, California) and analyzed using the fluorometric (acidification) method (EPA 445) for chlorophyll-a and pheophytin-a; persulfate digestion and spectrophotometric methods 10208 (total nitrogen) and 10210 (total phosphorus); and spectrophotometric methods 10209 (orthophosphate), 10205 (ammonia), and 10206 (nitrate).

Cyanobacterial counts by genus were sampled in the morning (between approximately 7 am and 11 am) at all sites on 2/28, 3/7, 3/14, 3/16, 3/18, and 3/22 at all sites except H-FDD on 2/28/2022. Microcystin and anatoxin-a were sampled on the same dates at all sites except H-NS, H-ES, and H-SS on 2/28 and 3/14/2022. In addition, microcystin and anatoxin-a were sampled on 4/4/2022 at H-S, H-FD, H-BL, and H-BLS. Grab samples were shipped overnight to the analytical laboratory (Bend Genetics, Sacramento, California) and analyzed using microscopy for identification of potentially toxigenic cyanobacteria (PTOX) and enzyme linked immunosorbent assay (ELISA) for total anatoxin-a and total microcystin/nodularin concentrations.

Table 1. Lake Henshaw water quality monitoring sites for algaecide effectiveness monitoring in February, March, and April 2022.

Site ID	Location	Latitude	Longitude
H-S	Southwestern shoreline at beach adjacent to fishing dock	33.23496°N	116.75617°W
H-FD	Southwestern shoreline at the in-water end of the fishing dock in surface waters	33.23544°N	116.75568°W
H-FDD	Southwestern shoreline at the in-water end of the fishing dock in bottom waters	33.23544°N	116.75568°W
H-BLS	Buoy line at dam in surface waters	33.23963°N	116.76174°W
H-BL	Buoy line at dam in bottom waters	33.23963°N	116.76174°W
H-NL	Northern portion of lake in surface waters	33.24600°N	116.75300°W
H-ML	Mid-lake in surface waters	33.23890°N	116.75275°W
H-MLD	Mid-lake in bottom waters	33.23890°N	116.75275°W
H-SL	Southern portion of lake in surface waters	33.23000°N	116.74400°W
H-NS	Northern shoreline at beach	33.24729°N	116.75414°W
H-ES	Eastern shoreline at beach	33.23546°N	116.73801°W
H-SS	Southern shoreline at beach	33.22659°N	116.74316°W

Figure 1. Lake Henshaw water quality monitoring sites for algaecide effectiveness monitoring in February, March, and April 2022.

3 RESULTS

3.1 *In situ* Water Quality

Water quality results for *in situ* measurements are summarized below and shown in Figures A-1 through A-9 (Appendix A).

3.1.1 Water temperature

Across all sampling dates and for both morning and afternoon *in situ* measurements, water temperatures at the deeper open water sites (i.e., H-BL, H-FD, H-ML) remained relatively stable throughout the water column, ranging 12–15°C, with some slight increases in the upper water column, particularly during the afternoon. Water temperatures at the shallow open water sites (i.e., H-NL, H-SL) tended to be 1–2°C warmer at the surface, particularly during the afternoon, but were also generally consistent through the water column. Overall, there was no evidence of thermal stratification at the open water sites. Water temperatures tended to be warmest at the shoreline sites (i.e., H-S, H-NS, H-ES, H-SS), particularly during afternoon measurements.

3.1.2 Dissolved oxygen

Across all sampling dates, DO readings were generally high (9–13 mg/L) in surface waters and at mid-depths at deeper open water sites (i.e., H-BL, H-FD, H-ML), as well as in shallow open water sites (i.e., H-NL, H-SL). Concentrations in surface waters in the afternoon often exceeded 10 mg/L and 100% saturation, with some measurements at 13 mg/L and over 120% saturation, indicating high levels of photosynthesis, both prior to and following algaecide treatment. DO declined to concentrations between 4–6 mg/L in bottom waters of deeper water sites (i.e., H-BL, H-FD). The lowest concentrations (approximately 4 mg/L) occurred at H-BL in bottom waters on the afternoon of 3/16 and the morning of 3/17, approximately 48–60 hours following algaecide application.

3.1.3 Turbidity

Across all sampling dates, turbidity readings were variable throughout the water column at open water and shoreline sites and generally remained between 50–100 Formazin Nephelometric Units (FNU). There was no pattern with water depth or across sites that suggested pronounced accumulation of algae, either before (3/14 morning), during (3/14 afternoon and 3/15 morning), or after (3/16 – 3/22) algaecide treatment. There was no consistent vertical correspondence between turbidity readings and DO, where high turbidity and high DO might indicate an accumulation of photosynthesizing algae, and high turbidity and low DO might indicate an accumulation of dying algae following algaecide treatment.

3.1.4 pH/ORP

pH readings were stable throughout the water column on all sampling dates, remaining near 9.0 (s.u.) at all sites. Elevated pH in eutrophic lakes like Lake Henshaw is typically indicative of high rates of photosynthesis. ORP ranged from approximately +50 to +235 mV on all sampling dates and was generally consistent with depth. Positive ORP indicates oxidizing conditions, which is consistent with the generally high DO concentrations measured on all sampling dates. Relatively lower DO (4–6 mg/L) in bottom waters of deeper water sites (i.e., H-BL, H-FD) did not correspond to a decrease in ORP in bottom waters, suggesting that the lower DO was somewhat ephemeral. ORP values supporting denitrification (i.e., reduction of nitrate [NO_3^-] to nitrogen gas [N_2]) tend to be in the range -50 to +50 mV, such that the Lake Henshaw ORP values were likely too high to support water column denitrification.

3.1.5 Conductivity

Conductivity readings were generally stable throughout the water column on all sampling dates and at all sites, with a range of 452–770 µS/cm and an average reading of 519 µS/cm. Conductivity values less than 1,000 µS/cm for lakes and reservoirs are generally considered to be moderate. There was no pattern with water depth or across sites, either before (3/14 morning), during (3/14 afternoon and 3/15 morning), or after (3/16 – 3/22) algaecide treatment.

3.1.6 Chlorophyll-*a* and pheophytin-*a*

Chlorophyll-*a* (chl-*a*) concentrations (a measure of algal biomass) varied by sampling site, ranging 25–153 µg/L throughout the sampling period (Table 2). The lowest chl-*a* concentration was measured in bottom waters at H-MLD on 3/14, just prior to algaecide treatment, and at H-FDD on 3/18, four days following algaecide treatment. The highest chl-*a* concentration was measured in surface waters at H-ML on 2/28, three weeks prior to algaecide treatment (Table 2). Comparisons between chl-*a* concentrations at H-FD and H-FDD, H-BLS and HBL, and H-ML and H-MLD indicate no consistent pattern between surface and depth samples at open water sites. Chl-*a* was generally lower at H-MLD across all sampling dates, whereas concentrations measured at H-BL were among the highest. Almost all chl-*a* concentrations were greater than 50 µg/L in Lake Henshaw during the sampling period, indicating eutrophic conditions in late winter and spring 2022.

Overall, chl-*a* concentrations decreased modestly (i.e., by 10%–40%) following algaecide treatment at the deeper open water sites (H-FD, H-FDD, H-BLS, H-BL, H-NL, H-ML [surface]) and two shoreline sites (H-S, H-SS) (Figure 2 and Table 2). Chl-*a* concentrations increased modestly at one shallow open water site (H-SL) and two shoreline sites (H-NS, H-ES) following algaecide treatment, although only one pre-treatment data point is available for comparison. Chl-*a* concentrations increased by over 200% at H-MLD (bottom water) following algaecide treatment (Figure 2, Table 2), but only one pre-treatment data point is available for comparison and it was a particularly low value.

Pheophytin-*a* (phe-*a*) concentrations were generally lower than chl-*a* samples that were collected at the same location. Phe-*a* ranged 40–129 µg/L throughout the sampling period, except for eight samples collected on 3/18 (Table 2).

The ratio of phe-*a* to chl-*a* was similar across sampling sites and across all sampling dates, generally in the range 0.5–0.8. Ratios were generally higher following algaecide treatment, which is to be expected from senescing (dying) algae; the higher ratios were mainly due to samples in which phe-*a* was greater than chl-*a* (i.e., the ratio was greater than 1.0).

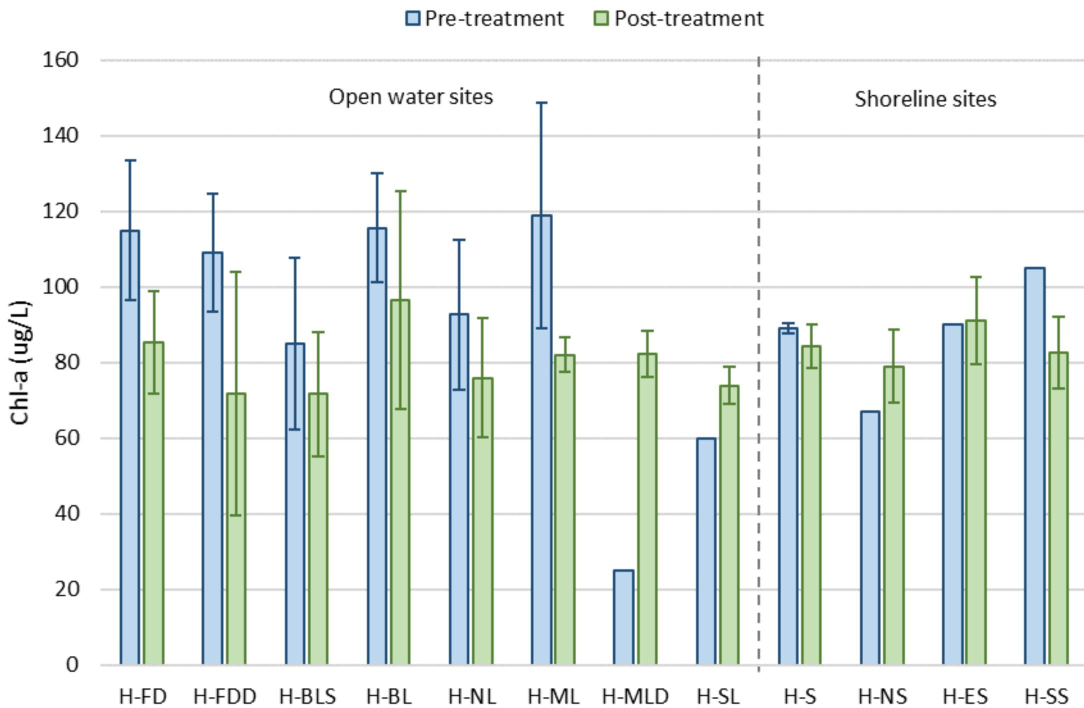

Figure 2. Chlorophyll-a (chl-a) concentrations at Lake Henshaw open water sites (to the left of the vertical dashed line) and shoreline sites (to the right of vertical dashed line) before and after peroxide-based algaecide treatment. Data are presented as average ±1 standard deviation, with number of samples per site and sampling dates presented in Table 2.

Table 2. Chlorophyll-*a* and pheophytin-*a* concentrations measured in Lake Henshaw before and after peroxide-based algaecide treatment.

Date	Site ID											
	H-S	**H-FD**	**H-FDD**	**H-BLS**	**H-BL**	**H-NL**	**H-ML**	**H-MLD**	**H-SL**	**H-NS**	**H-ES**	**H-SS**
Chlorophyll-a (µg/L)												
2/28/2022	90	102	120	101	130	97	153	-	-	-	-	-
3/7/2022	88	128	98	59	116	71	107	-	-	-	-	-
3/14/2022	-	-	-	95	101	110	97	25	60	67	90	105
Average Pre-treatment	*89*	*115*	*109*	*85*	*116*	*93*	*119*	*25*	*60*	*67*	*90*	*105*
3/14/2022	93	97	99	-	-	-	-	-	-	-	-	-
3/16/2022	85	75	79	83	94	58	86	83	74	68	95	92
3/18/2022	85	77	25	52	65	82	77	76	69	86	100	73
3/22/2022	77	75	84	87	135	88	83	88	79	83	78	83
4/4/2022	82	103	-	65	92	-	-	-	-	-	-	-
Average Post-treatment	*84*	*85*	*72*	*72*	*97*	*76*	*82*	*82*	*74*	*79*	*91*	*83*
Average % Difference	-5%	-26%	-34%	-16%	-17%	-18%	-31%	229%	23%	18%	1%	-21%
Pheophytin-a (µg/L)												
2/28/2022	48	58	68	56	73	59	83	-	-	-	-	-
3/7/2022	70	79	86	129	71	51	63	-	-	-	-	-
3/14/2022	-	-	-	61	79	60	63	19	41	42	61	62
Average Pre-treatment	59	69	77	82	74	57	70	19	41	42	61	62
3/14/2022	61	59	63	-	-	-	-	-	-	-	-	-
3/16/2022	57	56	58	58	65	51	60	61	58	53	66	63
3/18/2022	68	83	40	74	88	105	89	91	109	73	64	52
3/22/2022	43	47	50	50	84	54	48	53	49	52	45	56
4/4/2022	48	60	-	40	57	-	-	-	-	-	-	-
Average Post-treatment	*55*	*61*	*53*	*56*	*73*	*70*	*66*	*68*	*72*	*59*	*58*	*57*
Average % Difference	-6%	-11%	-31%	-32%	-1%	24%	-6%	260%	76%	41%	-4%	-8%

Date	Site ID											
	H-S	**H-FD**	**H-FDD**	**H-BLS**	**H-BL**	**H-NL**	**H-ML**	**H-MLD**	**H-SL**	**H-NS**	**H-ES**	**H-SS**
Ratio (Phe-a:Chl-a)												
2/28/2022	0.5	0.6	0.6	0.6	0.6	0.6	0.5	-	-	-	-	-
3/7/2022	0.8	0.6	0.9	2.2	0.6	0.7	0.6	-	-	-	-	-
3/14/2022	-	-	-	0.6	0.8	0.5	0.7	0.7	0.7	0.6	0.7	0.6
Average Pre-treatment	0.7	0.6	0.8	1.1	0.7	0.6	0.6	0.7	0.7	0.6	0.7	0.6
3/14/2022	0.7	0.6	0.6	-	-	-	-	-	-	-	-	-
3/16/2022	0.7	0.7	0.7	0.7	0.7	0.9	0.7	0.7	0.8	0.8	0.7	0.7
3/18/2022	0.8	1.1	1.6	1.4	1.3	1.3	1.2	1.2	1.6	0.8	0.6	0.7
3/22/2022	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.7
4/4/2022	0.6	0.6	-	0.6	0.6	-	-	-	-	-	-	-
Average Post-treatment	*0.7*	*0.7*	*0.9*	*0.8*	*0.8*	*0.9*	*0.8*	*0.8*	*1.0*	*0.7*	*0.6*	*0.7*
Average % Difference	4%	20%	17%	-27%	20%	56%	39%	19%	43%	22%	-10%	17%

Shaded cells indicate results from samples collected following algaecide treatment.
\- Indicate no sampling occurred.

3.1.7 Cyanobacterial cell counts

The genera represented in cyanobacterial cell counts were *Planktothrix*, *Microcystis*, *Snowella*, *Aphanocapsa*, and *Dolichospermum*. The first four genera were present in all samples, with *Planktothrix* as the dominant genus and *Dolichospermum* as relatively uncommon. Generally, cell densities (cells/mL) were similar (i.e., the same order of magnitude) for each genus across sampling sites on a given sampling date (Figure 3, Table 3), indicating little spatial variation in the relative dominance of the different cyanobacteria either before or after algaecide treatment. Cell densities tended to be lowest on 3/16 (24–48 hours after algaecide treatment) but returned to pre-treatment levels by 3/22. At sites H-BLS and H-ML, the highest cell counts for *Planktothrix* (and by extension total cyanobacteria) were reported four days following algaecide treatment, and on average *Planktothrix* cell counts increased following treatment at sites H-S, H-BLS, and H-ML (Figure 3, Table 3). On average *Snowella* and *Dolichospermum* cell counts decreased following treatment at sites H-S, H-BLS, and H-ML (Figure 3, Table 3). Changes in cell counts were variable before and after treatment for *Microcystis* and *Aphanocapsa* (Figure 3, Table 3).

Patterns in cell biovolume were dominated by *Planktothrix* (14.1 μm^3) and *Microcystis* (22.4 μm^3) due to their relatively large size and high abundance.

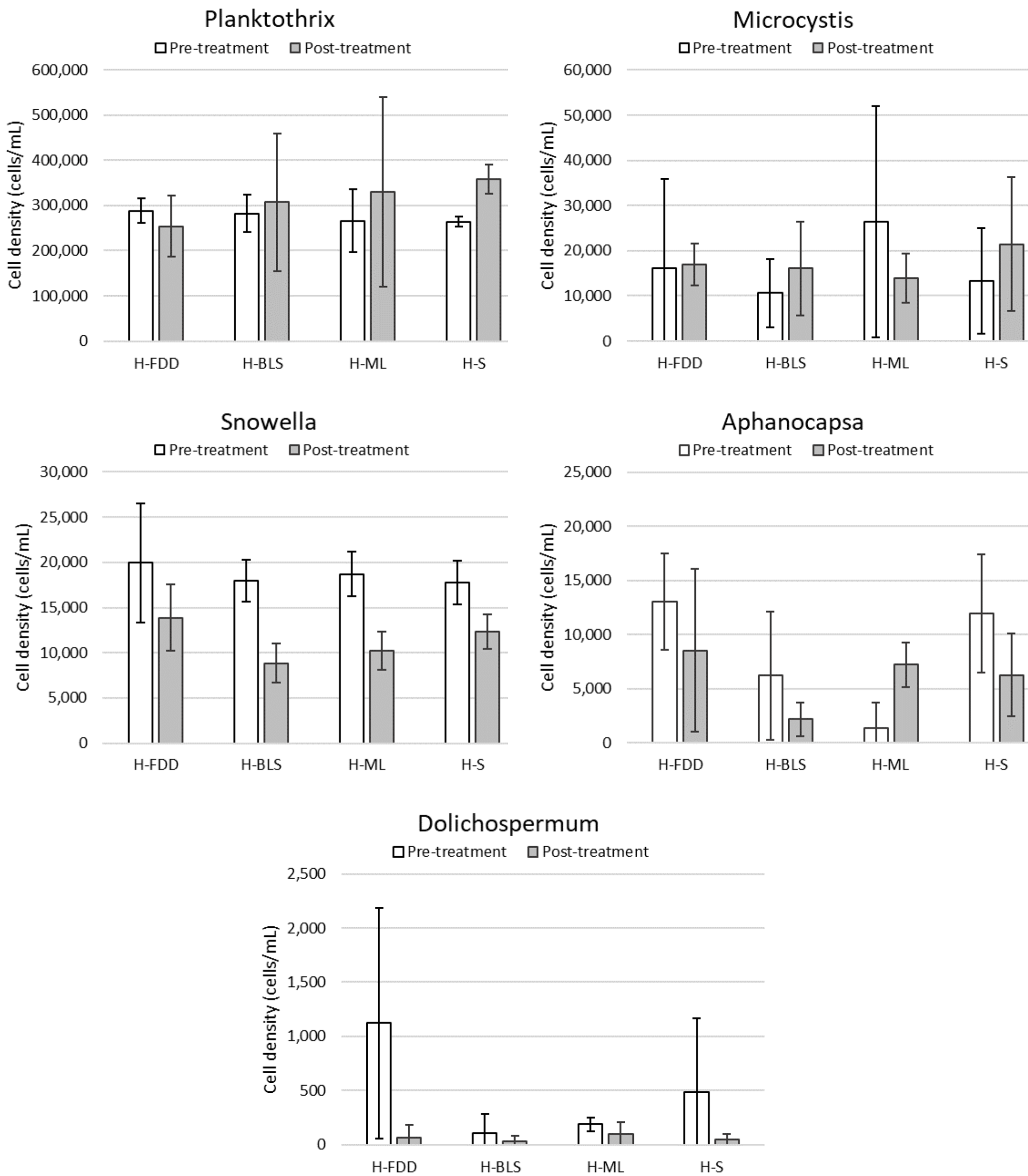

Figure 3. Cyanobacterial cell densities measured in Lake Henshaw before and after peroxide-based algaecide treatment. Data are presented as average ±1 standard deviation, with number of samples per site and sampling dates presented in Table 3.

Table 3. Cyanobacterial cell densities measured in Lake Henshaw before and after peroxide-based algaecide treatment.

Date	Site ID			
	H-S	H-FD/FDD [1]	H-BLS	H-ML
Planktothrix (cells/mL)				
2/28/2022	256,100	269,100	240,240	192,400
3/7/2922	272,250	308,000	283,250	276,375
3/14/2022	-	-	322,718	329,548
Average Pre-treatment	*264,175*	*288,550*	*282,069*	*266,108*
3/14/2022	368,945	353,453	-	-
3/16/2022	314,746	205,695	189,370	186,105
3/18/2022	359,755	230,743	479,673	570,607
3/22/2022	392,150	226,972	253,115	232,913
Average Post-treatment	*358,899*	*254,216*	*307,386*	*329,875*
Average % Difference	36%	-12%	9%	24%
Microcystis (cells/mL)				
2/28/2022	5,006	2,225	5,340	8,900
3/7/2922	21,500	30,100	19,350	55,900
3/14/2022	-	-	7,252	14,503
Average Pre-treatment	*13,253*	*16,163*	*10,647*	*26,434*
3/14/2022	9,173	13,650	-	-
3/16/2022	9,784	20,383	8,736	12,230
3/18/2022	26,910	12,159	27,907	9,369
3/22/2022	39,840	21,580	11,620	19,920
Average Post-treatment	*21,427*	*16,943*	*16,088*	*13,840*
Average % Difference	62%	5%	51%	-48%
Snowella (cells/mL)				
2/28/2022	16,100	15,295	17,388	17,710
3/7/2922	19,475	24,600	20,500	21,525
3/14/2022	-	-	16,050	16,853
Average Pre-treatment	*17,788*	*19,948*	*17,979*	*18,696*
3/14/2022	9,695	17,655	-	-
3/16/2022	12,188	14,773	7,519	7,848
3/18/2022	13,335	8,890	7,620	11,007
3/22/2022	14,150	14,150	11,320	11,792
Average Post-treatment	*12,342*	*13,867*	*8,820*	*10,216*
Average % Difference	-31%	-30%	-51%	-45%

Date	Site ID			
	H-S	**H-FD/FDD [1]**	**H-BLS**	**H-ML**
Aphanocapsa (cells/mL)				
2/28/2022	8,110	16,220	0	4,055
3/7/2922	15,800	9,875	11,850	0
3/14/2022	-	-	6,791	0
Average Pre-treatment	*11,955*	*13,048*	*6,214*	*1,352*
3/14/2022	11,900	19,403	-	-
3/16/2022	4,760	7,933	1,700	6,545
3/18/2022	5,308	4,069	3,892	9,554
3/22/2022	3,174	2,822	945	5,643
Average Post-treatment	*6,286*	*8,557*	*2,179*	*7,247*
Average % Difference	-47%	-34%	-65%	436%
Dolichospermum (cells/mL)				
2/28/2022	0	370	0	185
3/7/2922	969	1875	313	250
3/14/2022	-	-	0	129
Average Pre-treatment	*485*	*1,123*	*104*	*188*
3/14/2022	78	0	-	-
3/16/2022	0	0	89	208
3/18/2022	0	235	0	0
3/22/2022	99	26	0	92
Average Post-treatment	*44*	*65*	*30*	*100*
Average % Difference	-91%	-94%	-72%	-47%
Total cyanobacteria (cells/mL)				
2/28/2022	285,316	303,210	262,968	223,250
3/7/2922	329,994	374,450	335,263	354,050
3/14/2022	-	-	352,811	361,033
Average Pre-treatment	*307,655*	*338,830*	*317,014*	*312,778*
3/14/2022	399,791	404,161	-	-
3/16/2022	341,478	248,784	207,414	212,936
3/18/2022	405,308	256,096	519,092	600,537
3/22/2022	449,413	265,550	277,000	270,360
Average Post-treatment	*398,998*	*293,648*	*334,502*	*361,278*
Average % Difference	30%	-13%	6%	16%

Shaded cells indicate results from samples collected following algaecide treatment.
- Indicates no sampling occurred.
[1] H-FD (surface) samples were collected pre-treatment and H-FDD (bottom) samples were collected post-treatment. Since the water column at this site was generally well-mixed, the pre- and post-treatment comparisons combine the data from the surface and bottom water sites.

3.1.8 Microcystin and anatoxin-a

Anatoxin-a was not detected at any sampling site before, during, or after algaecide treatment (Table 4). Microcystin concentrations were similar throughout the lake during each sampling event. Microcystin concentrations ranged 0.14–0.53 µg/L prior to algaecide treatment and 0.16–0.71 µg/L following treatment. At sites where sampling occurred on 2/28 and 3/4/2022, microcystin concentrations generally decreased prior to algaecide treatment and generally increased following treatment (Figure 4). At sites where sampling first occurred on the morning of algaecide application, microcystin concentrations generally increased through the remainder of the sampling period. Since the ELISA method used to analyze cyanotoxin concentrations includes a cell lysing step, reported concentrations should represent both microcystin within the cyanobacterial cells and dissolved microcystin in the water column, and any increases following algaecide treatment are expected to be the result of additional cellular production rather than simply cell wall lysing during senescence. Additionally, since peroxide has the potential to chemically break down microcystin during treatment, the higher concentrations post-treatment suggest that either the anticipated breakdown did not occur, or more cyanotoxin was produced by the cyanobacteria during or immediately following treatment such that the net amount measured in the lake post-treatment was still generally greater than concentrations measured prior to treatment.

Table 4. Cyanotoxin concentrations in Lake Henshaw before and after peroxide-based algaecide treatment.

Date	Site ID											
	H-S	**H-FD**	**H-FDD**	**H-BL**	**H-BLS**	**H-NL**	**H-ML**	**H-MLD**	**H-SL**	**H-NS**	**H-ES**	**H-SS**
Microcystin (µg/L)												
2/28/2022	0.50	0.39	-	0.53	-	0.37	0.40	-	0.47	-	-	-
3/7/2022	0.43	0.42	-	0.47	-	0.31	0.48	-	0.42	-	-	-
3/14/2022	-	-	-	0.28	-	0.15	0.17	-	0.19	0.17	0.14	0.20
Average Pre-treatment	*0.47*	*0.41*	-	*0.43*	-	*0.28*	*0.35*	-	*0.36*	*0.17*	*0.14*	*0.20*
3/14/2022	0.18	0.18	0.15	-	-	-	-	-	-	-	-	-
3/16/2022	0.16	0.16	0.19	0.23	0.24	0.25	0.21	0.29	0.29	0.22	0.24	0.28
3/18/2022	0.38	0.32	0.42	0.64	0.37	0.37	0.38	-	0.47	0.45	0.40	0.48
3/22/2022	0.48	0.71	0.52	0.61	0.51	0.45	0.39	-	0.36	0.37	0.39	0.62
4/4/2022	0.30	0.29	-	0.34	-	-	-	-	-	-	-	-
Average post-treatment	*0.30*	*0.33*	*0.32*	*0.46*	*0.37*	*0.36*	*0.33*	*0.29*	*0.37*	*0.35*	*0.34*	*0.46*
Average % Difference	-35%	-18%	-	7%	-	29%	-7%	-	4%	104%	145%	130%
Anatoxin-a (µg/L)												
2/28/2022	-	<0.015	-	<0.015	-	<0.015	<0.015	-	<0.015	-	-	-
3/7/2022	-	<0.015	-	<0.015	-	<0.015	<0.015	-	<0.015	-	-	-
3/14/2022	<0.015	<0.015	<0.015	<0.015	-	<0.015	<0.015	-	<0.015	<0.015	<0.015	<0.015
3/16//2022	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015
3/18/2022	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	-	<0.015	<0.015	<0.015	<0.015
3/22/2022	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	<0.015	-	<0.015	<0.015	<0.015	<0.015
4/4/2022	<0.015	<0.015	-	<0.015	-	-	-	-	-	-	-	-

Shaded cells indicate results from samples collected following algaecide treatment.
- Indicates no sampling occurred.

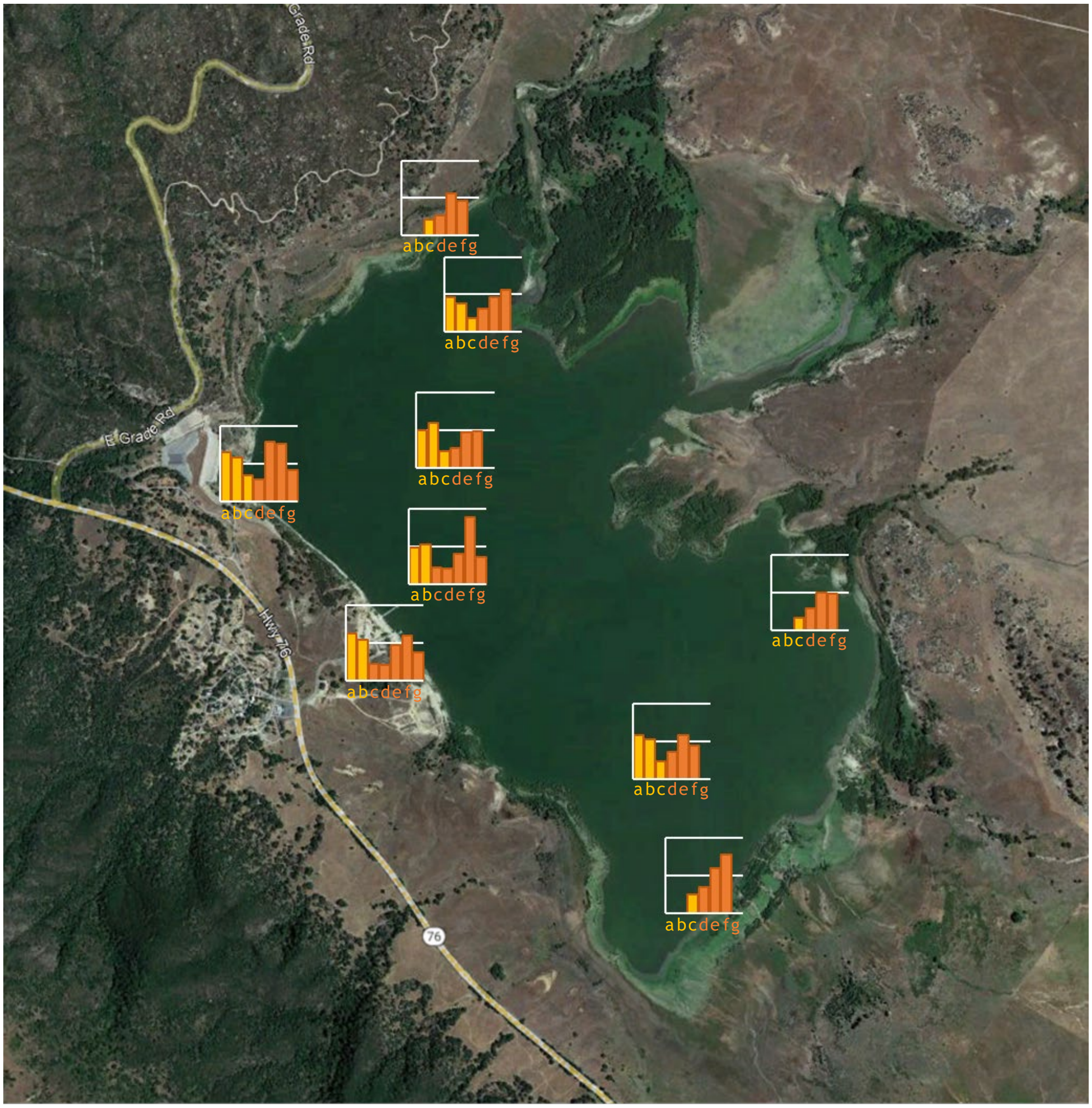

Figure 4. Microcystin concentrations in Lake Henshaw before and after algaecide treatment. Yellow bars and lower case letters represent microcystin concentrations from samples collected prior to algaecide treatment. Orange bars and lower case letters represent microcystin concentrations from samples collected after algaecide treatment. Sampling dates are as follows: a = 2/28/22; b = 3/7/22; c = 3/14/22; d = 3/16/22; e = 3/18/22; f = 3/22/22; and g = 4/4/22. White horizontal lines indicate 0.0, 0.4, and 0.8 ug/L microcystin. Missing bars indicate that no sampling occurred at a given sampling site on a given date.

3.1.9 Nutrients

Fewer analytical results are available for nutrients sampled prior to algaecide treatment than for the post-treatment sampling period, except for those collected at H-BL and H-BLS. In general, most nutrient species were slightly higher prior to algaecide treatment than after, though differences were small. Nutrient concentrations were generally similar across sites within given sampling dates.

Total nitrogen concentrations ranged 3.96–5.66 mg/L prior to algaecide treatment and 3.17–4.78 mg/L following treatment (Table 5). Nitrate concentrations were generally similar throughout the sampling period, ranging 0.10–0.21 mg/L and 0.12–0.21 mg/L prior to and following treatment, respectively. Ammonia concentrations ranged 0.01–0.11 mg/L prior to treatment and 0.01–0.05 mg/L following treatment.

Total phosphorous concentrations ranged 0.30–0.40 mg/L prior to algaecide treatment and 0.24–0.35 mg/L following treatment. Orthophosphate concentrations were generally low and were similar prior to and following treatment, ranging 0.01–0.03 mg/L for both sampling periods.

Table 5. Nutrients in Lake Henshaw before and after peroxide-based algaecide treatment.

Date	Site ID				
	H-FD	H-FDD	H-BLS	H-BL	H-MLD
Total nitrogen (mg/L)					
2/28/2022	-	-	5.66	4.82	-
3/7/2022	-	-	4.93	4.77	-
3/14/2022	4.28	3.96	4.56	4.28	4.09
3/16//2022	4.00	3.76	3.89	4.78	3.46
3/18/2022	3.17	4.45	4.30	4.04	3.46
3/22/2022	3.23	3.55	3.28	3.36	3.23
4/4/2022	3.89	-	3.63	3.54	-
Nitrate (mg/L)					
2/28/2022	-	-	0.21	0.12	-
3/7/2022	-	-	0.11	0.10	-
3/14/2022	0.16	0.16	0.17	0.12	0.20
3/16//2022	0.13	0.15	0.15	0.17	0.17
3/18/2022	0.12	0.16	0.13	0.13	0.15
3/22/2022	0.16	0.13	0.21	0.16	0.14
4/4/2022	0.14	-	0.12	0.13	-
Ammonia (mg/L)					
2/28/2022	-	-	0.09	0.03	-
3/7/2022	-	-	0.07	0.01	-
3/14/2022	0.11	0.01	0.05	0.01	0.01
3/16//2022	0.02	0.01	0.02	0.02	0.01
3/18/2022	0.02	0.01	0.05	0.01	0.01
3/22/2022	0.02	0.01	0.03	0.01	0.01
4/4/2022	0.06	-	<MRL	0.06	-

Date	Site ID				
	H-FD	**H-FDD**	**H-BLS**	**H-BL**	**H-MLD**
Total Phosphorous (mg/L)					
2/28/2022	-	-	0.39	0.40	-
3/7/2022	-	-	0.35	0.37	-
3/14/2022	0.31	0.31	0.30	0.32	0.30
3/16//2022	0.30	0.33	0.28	0.34	0.29
3/18/2022	0.25	0.30	0.31	0.35	0.29
3/22/2022	0.24	0.27	0.26	0.28	0.25
4/4/2022	0.22	-	0.25	0.20	-
Ortho-P (mg/L)					
2/28/2022	-	-	0.03 [C1,J]	0.01[C1,J]	-
3/7/2022	-	-	0.01[C1,J]	<0.015	-
3/14/2022	0.01 [C1,J]	0.01 [C1,J]	<0.015	<0.015	<0.015
3/16//2022	<0.015	0.01 [C1,J]	<0.015	0.03 [C1,J]	<0.015
3/18/2022	<0.015	<0.015	<0.015	<0.015	<0.015
3/22/2022	0.02 [C1,J]	0.02 [C1,J]	0.01 [C1,J]	0.01 [C1,J]	0.02 [C1,J]
4/4/2022	0.02	-	0.02	0.02	-

Shaded cells indicate results from samples collected following algaecide treatment.
[C1,J] Indicates the value of the result is below the MRL but above the threshold of sensitivity for the analytical instrument.

4 CONCLUSIONS

The application of 40,000 pounds (2.9 mg/L [ppm] on 40% of the lake surface area) of a peroxide-based (SePRO PAK 27) algaecide to Lake Henshaw on March 14–15, 2022, appears to have had a minor effect on HABs as evidenced by the following:

- Modest decreases in chl-*a* concentrations (i.e., 10%–40% post-treatment decrease) at the deeper open water sites and two shallow shoreline sites, and modest increases at most other sites;
- Modest increases in the ratio of phe-*a* to chl-*a* following treatment at most sites, suggesting a limited degree of senescing (dying) algae;
- Decreased cyanobacteria cell densities within 24–48 hours following treatment at most sites but a return to pre-treatment levels at 7–8 days following algaecide (and some increases in cell counts above pre-treatment concentrations four days post-treatment);
- Generally increased microcystin concentrations following treatment. Note that since peroxide has the potential to chemically break down microcystin during treatment, the higher concentrations post-treatment suggest that either this did not occur or more cyanotoxin was produced by the cyanobacteria during or immediately following treatment such that the net amount measured in the lake post-treatment was still generally greater than concentrations measured prior to treatment.

Conditions in Lake Henshaw during the peroxide-based algaecide application event were characterized by a generally well-mixed water column at both deep (> 10 feet) and moderate depth (< 10 feet) open water sites, and across open water and shoreline locations alike. Algal activity was high at all sites, as evidenced by supersaturated DO in surface waters, pH > 8.5 throughout the lake, elevated turbidity, and chl-*a* concentrations ranging 25–153 µg/L. Microcystin (< 0.8 ug/L) and anatoxin-a (<0.015 ug/L) concentrations were low at all sites. Low DO concentrations between 4–6 mg/L in bottom waters of deeper water sites (i.e., H-BL, H-FD) approximately 48–60 hours following algaecide application may indicate decomposition of senescing algae due to algaecide treatment. Nutrients were relatively low in surface and bottom waters compared with summer and fall 2021 concentrations, although in general nutrients do not appear to be limiting cyanobacteria growth in Lake Henshaw, and they did not appear to be affected by any cell lysing that may have occurred during algaecide application.

A higher dose of peroxide-based algaecide or a copper-based algaecide may be required to have a meaningful effect on HABs in Lake Henshaw under well-mixed conditions. A combination of algaecides may also provide a better lake response. The potential for benthic cyanotoxin production should be investigated in Lake Henshaw.

5 REFERENCES

Marine Biochemists. 2021. Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch. Prepared by Marine Biochemists, Anaheim, California for Vista Irrigation District, Vista, California.

Appendices

Appendix A

In situ Figures

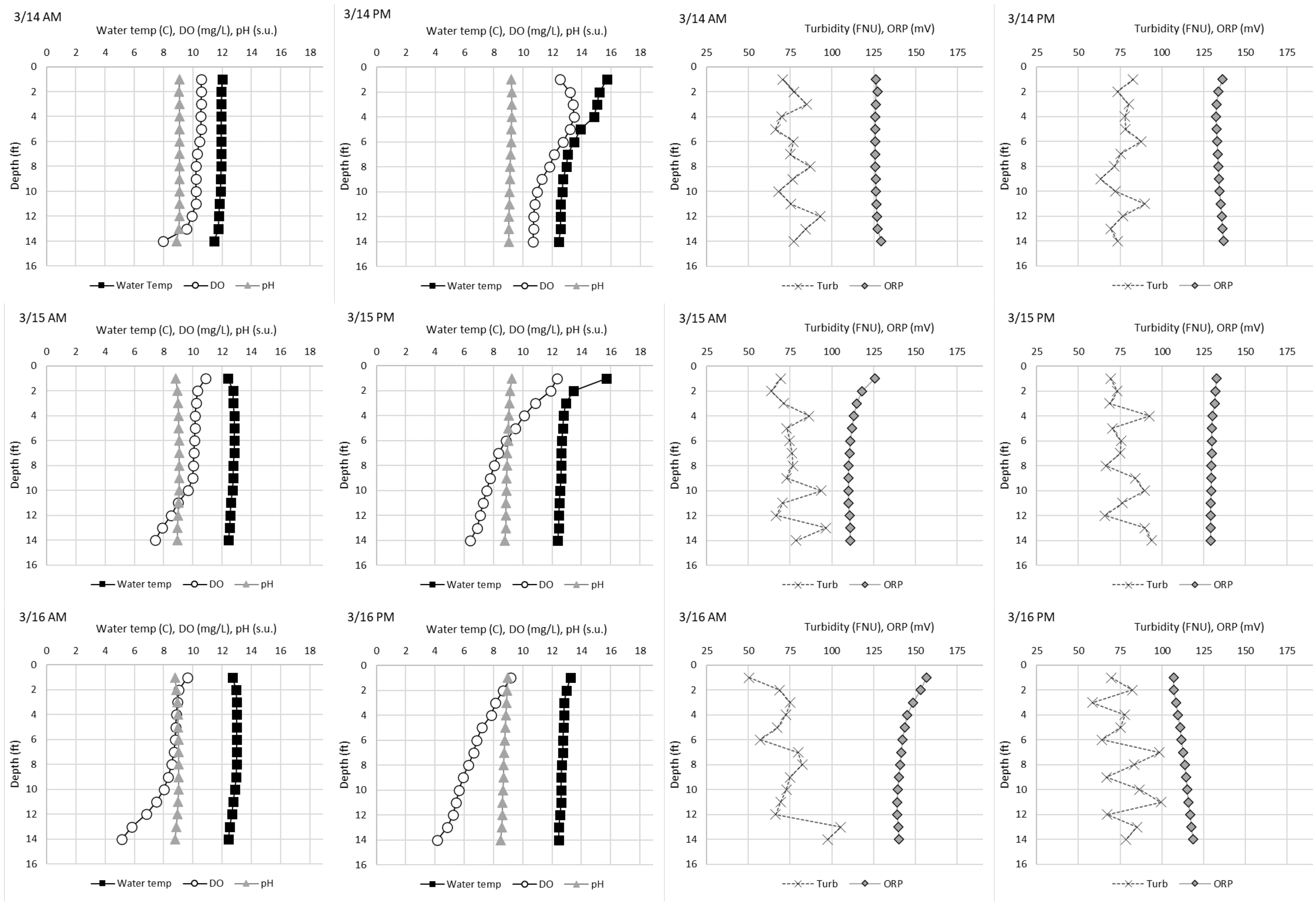

Figure A-1a. Buoy line (H-BL) *in situ* water quality morning and afternoon results for dates indicated.

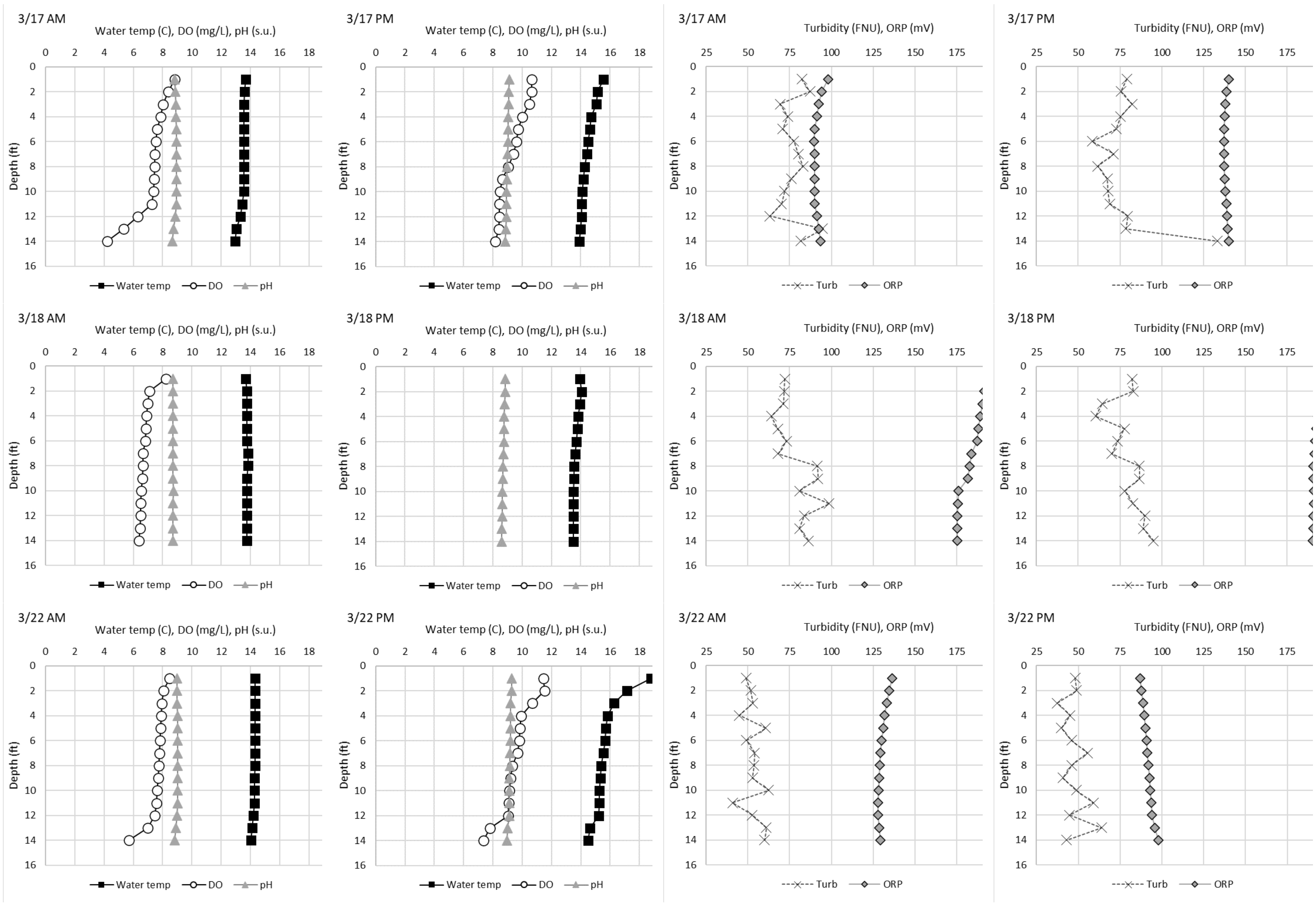

Figure A-1b. Buoy line (H-BL) *in situ* water quality morning and afternoon results for dates indicated. Note: (1) Negative DO readings during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced.

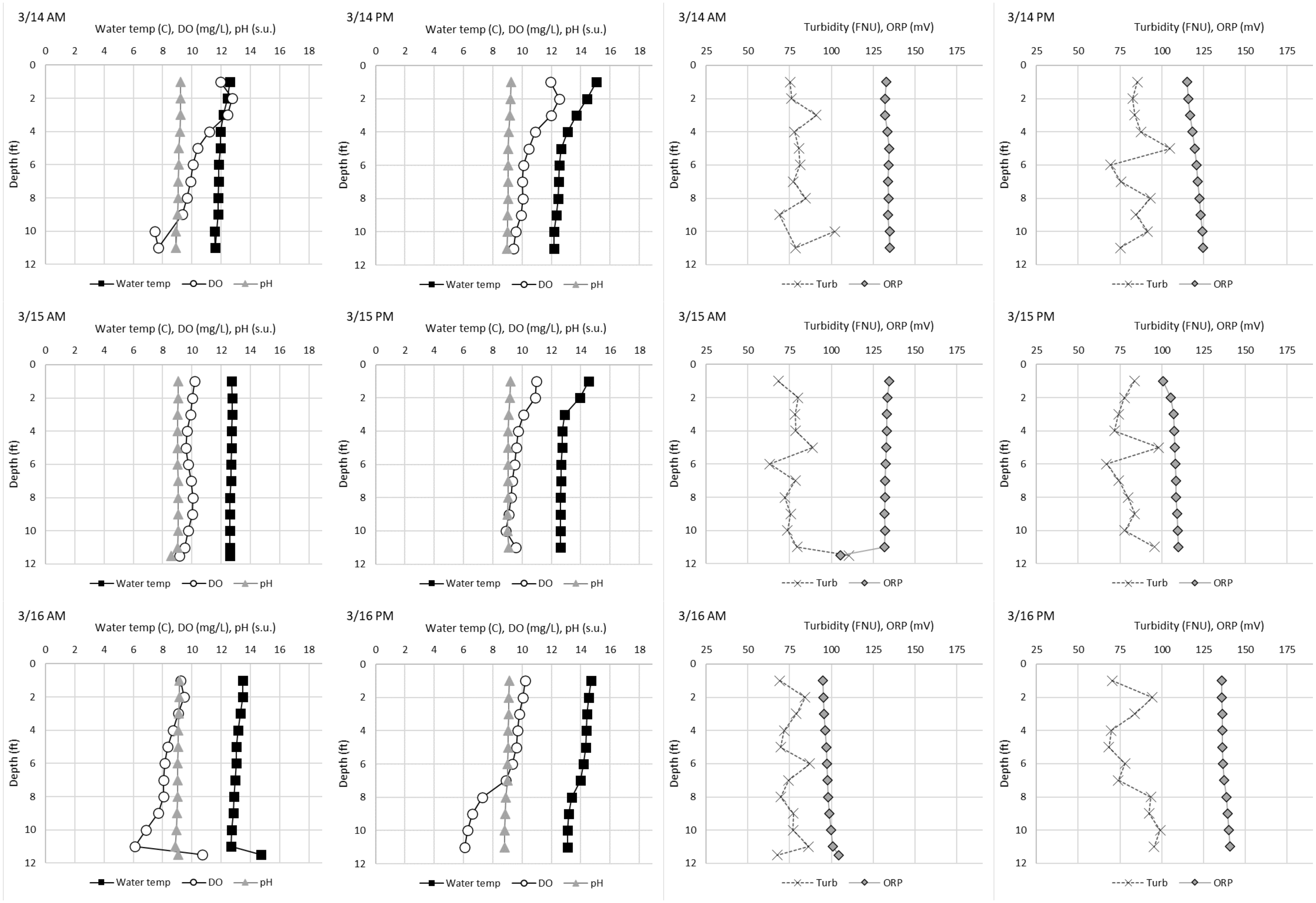

Figure A-2a. Fishing dock (H-FD) *in situ* water quality morning and afternoon results for dates indicated.

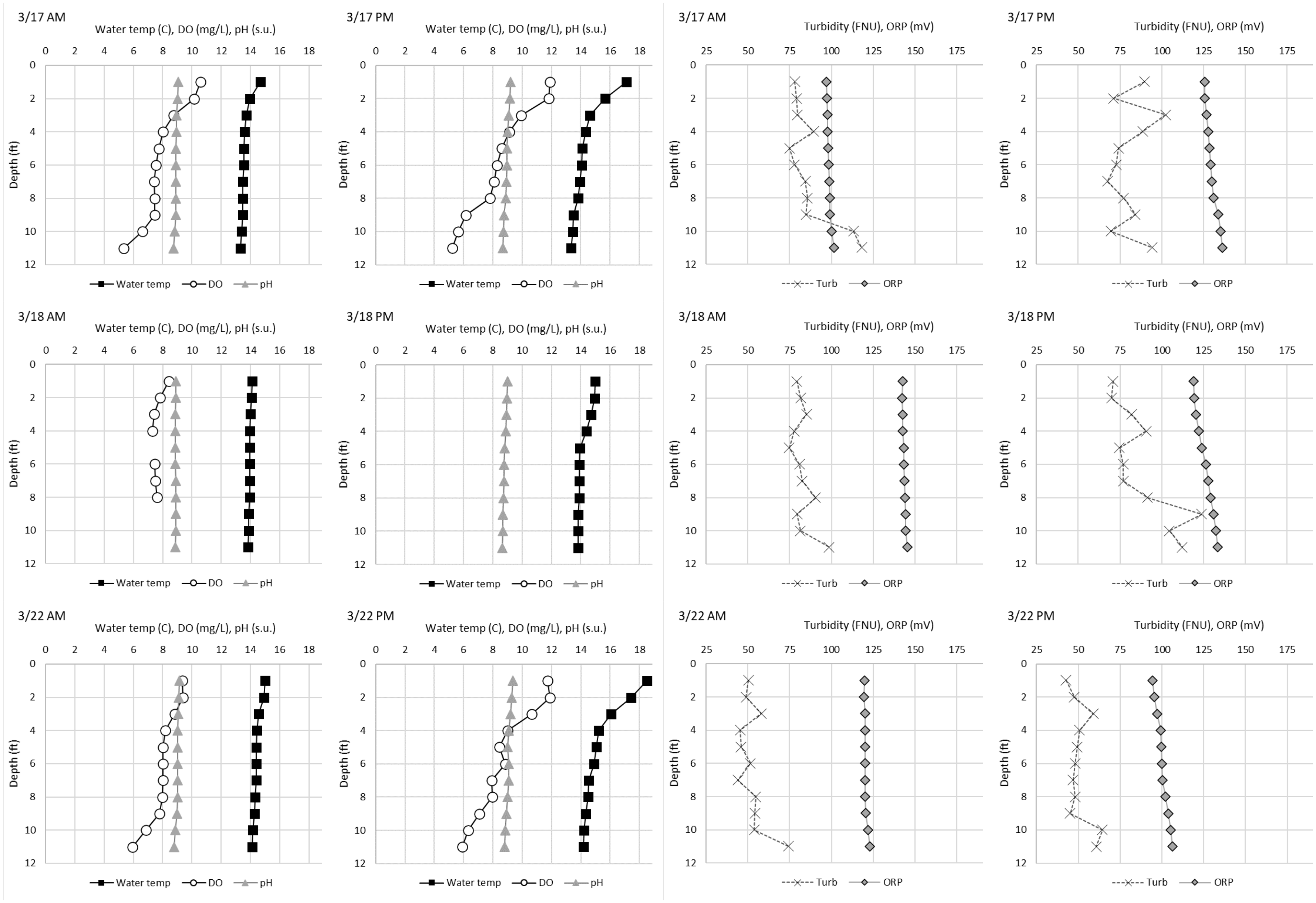

Figure A-2b. Fishing dock (H-FD) *in situ* water quality morning and afternoon results for dates indicated. Notes: (1) Negative DO readings at various depths during morning and at all depths during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced.

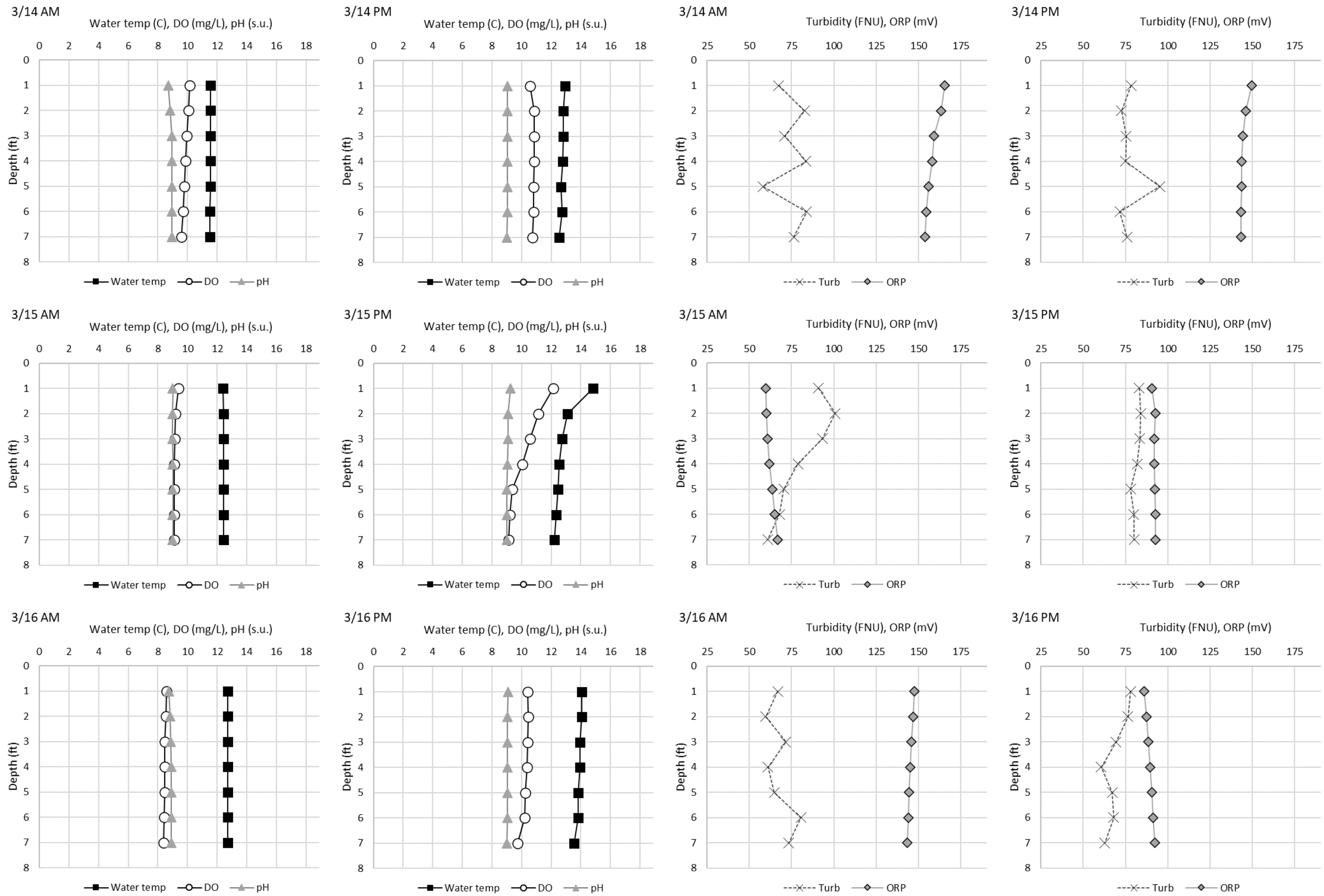

Figure A-3a. Middle lake (H-ML) *in situ* water quality morning and afternoon results for dates indicated.

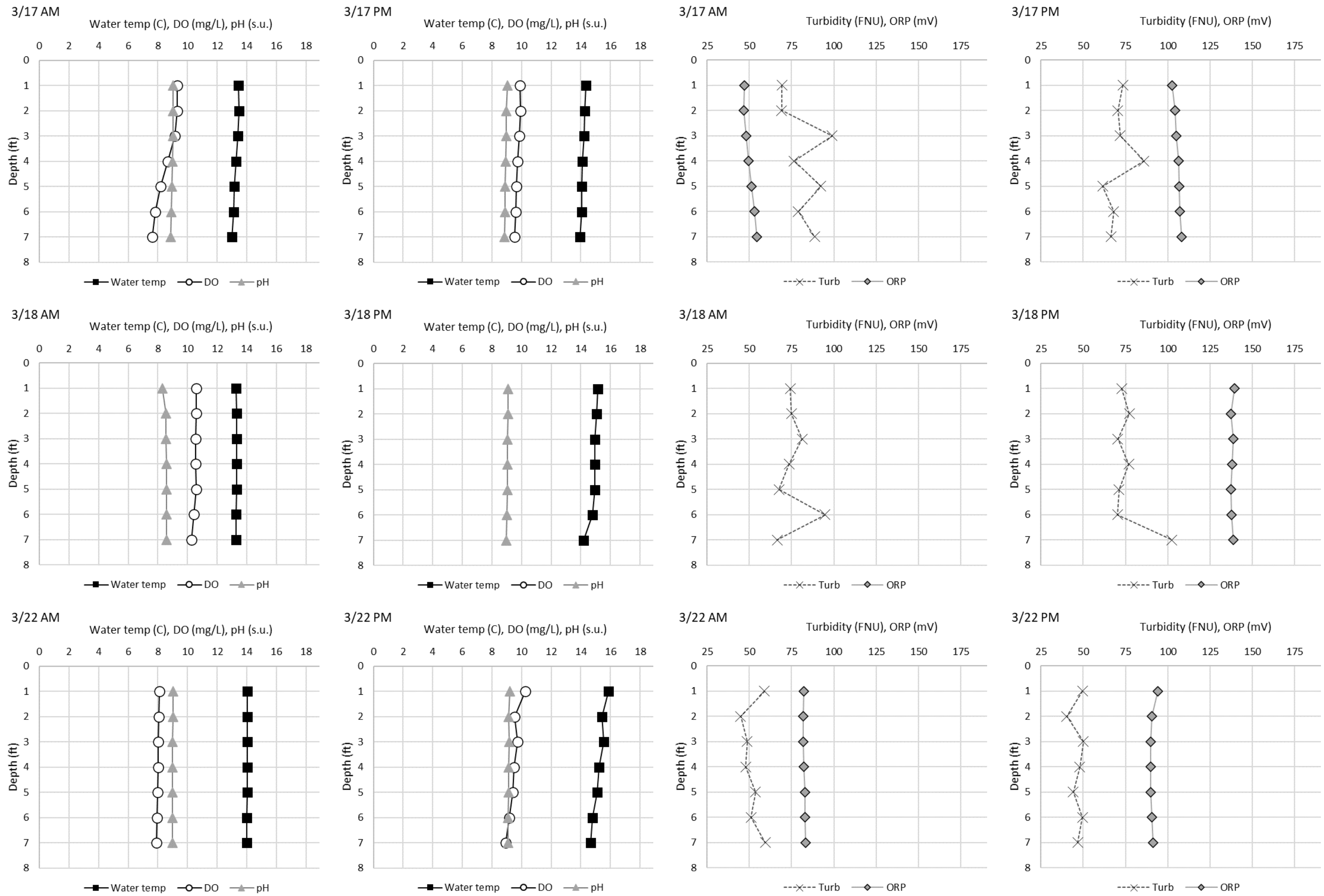

Figure A-3b. Middle lake (H-ML) *in situ* water quality morning and afternoon results for dates indicated. Notes: (1) Negative DO readings during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced; (2) ORP values greater than 200 mV during afternoon on 3/18 are not shown.

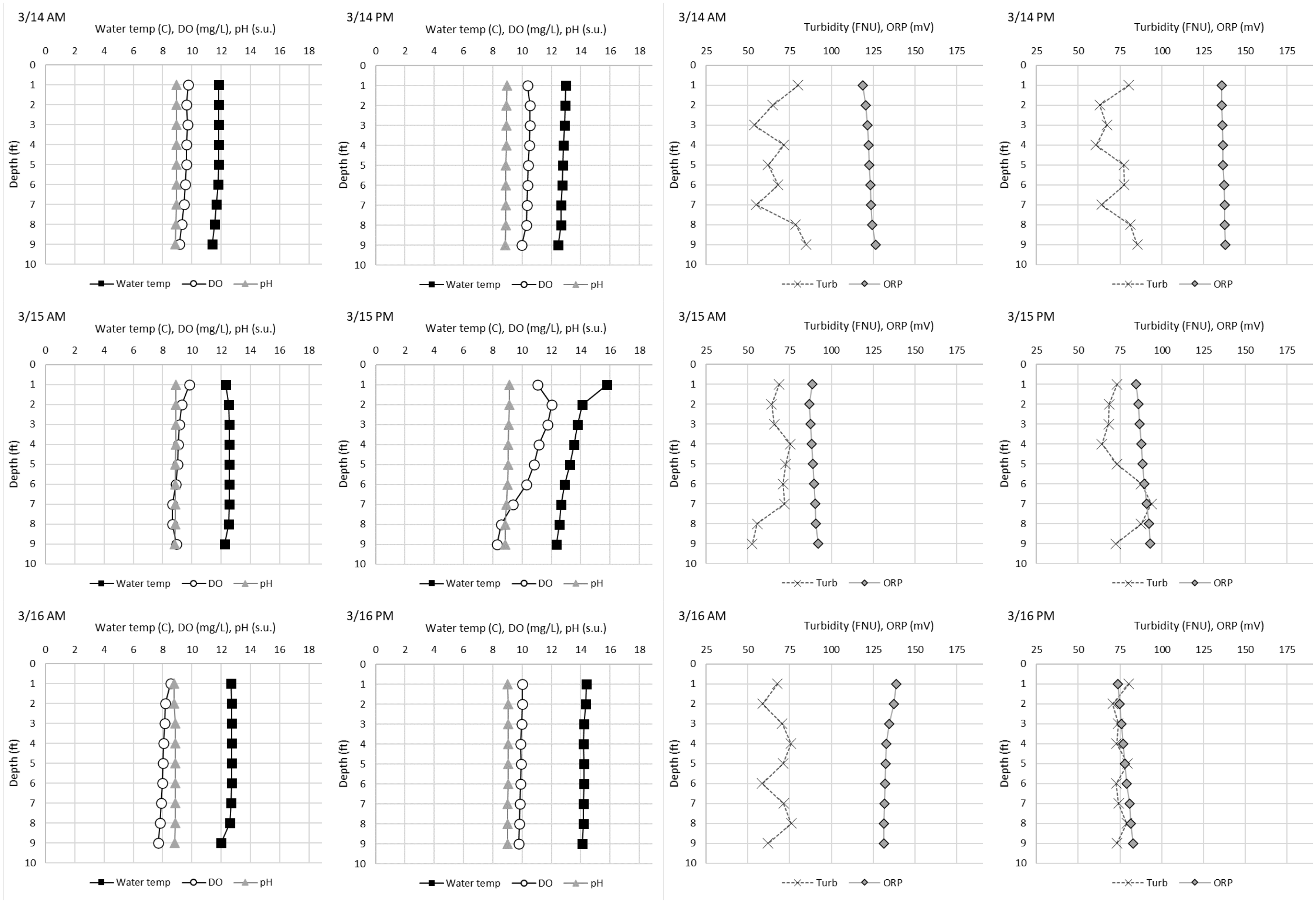

Figure A-4a. North lake (H-NL) *in situ* water quality morning and afternoon results for dates indicated.

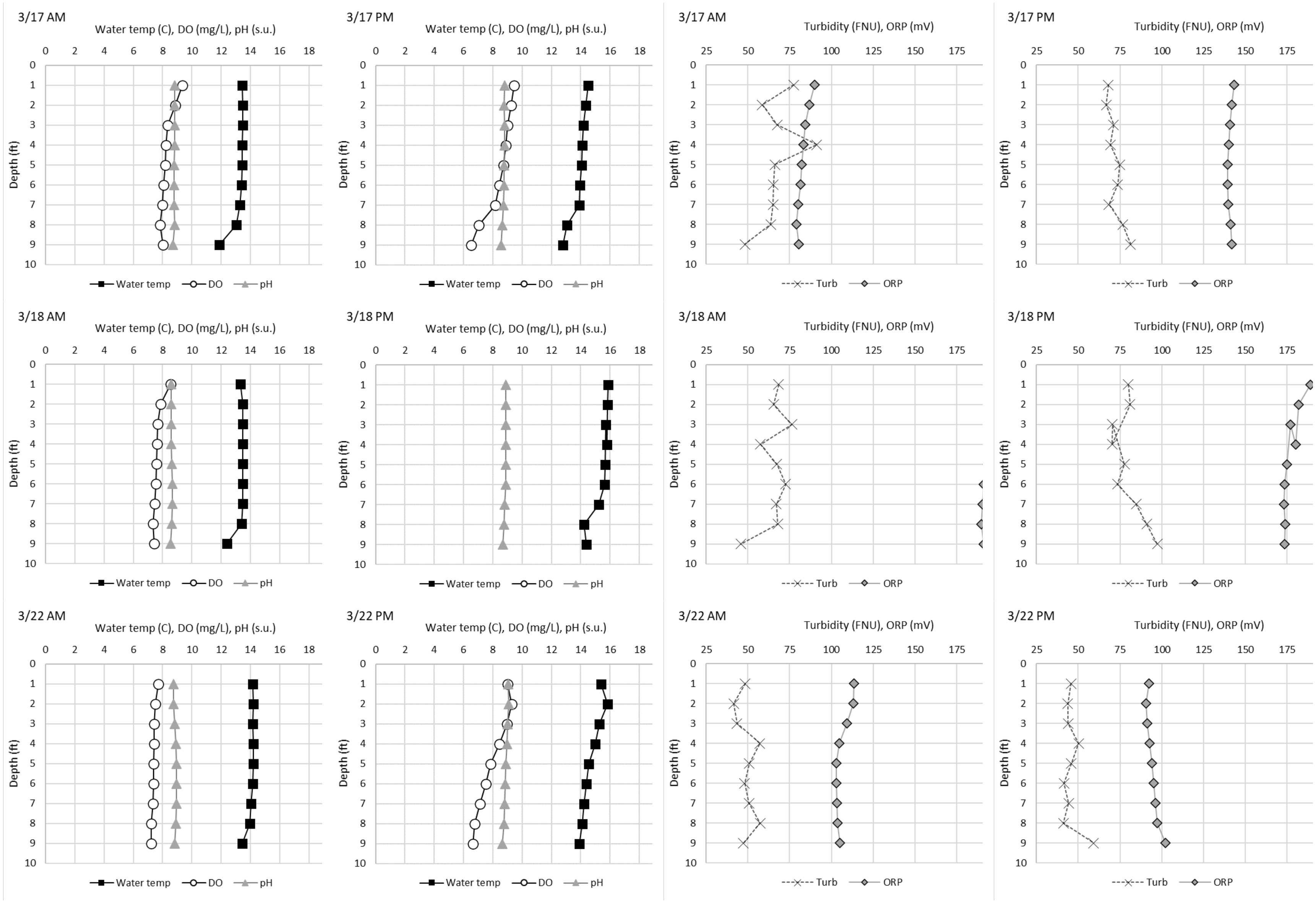

Figure A-4b. North lake (H-NL) *in situ* water quality morning and afternoon results for dates indicated. Notes: 1) Negative DO readings during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced; 2) Partial ORP values greater than 200 mV during morning on 3/18 are not shown.

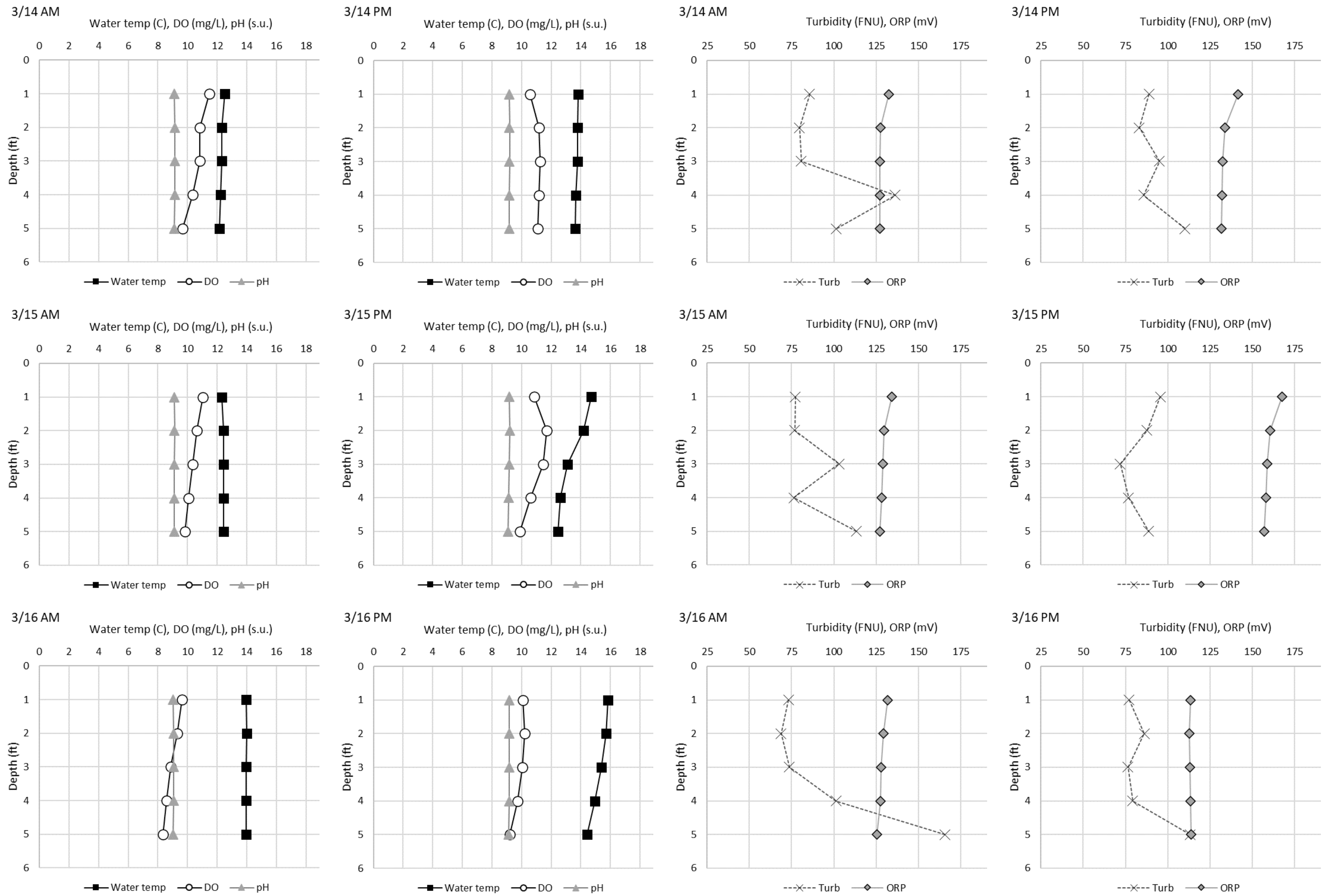

Figure A-5a. South lake (H-SL) *in situ* water quality morning and afternoon results for dates indicated.

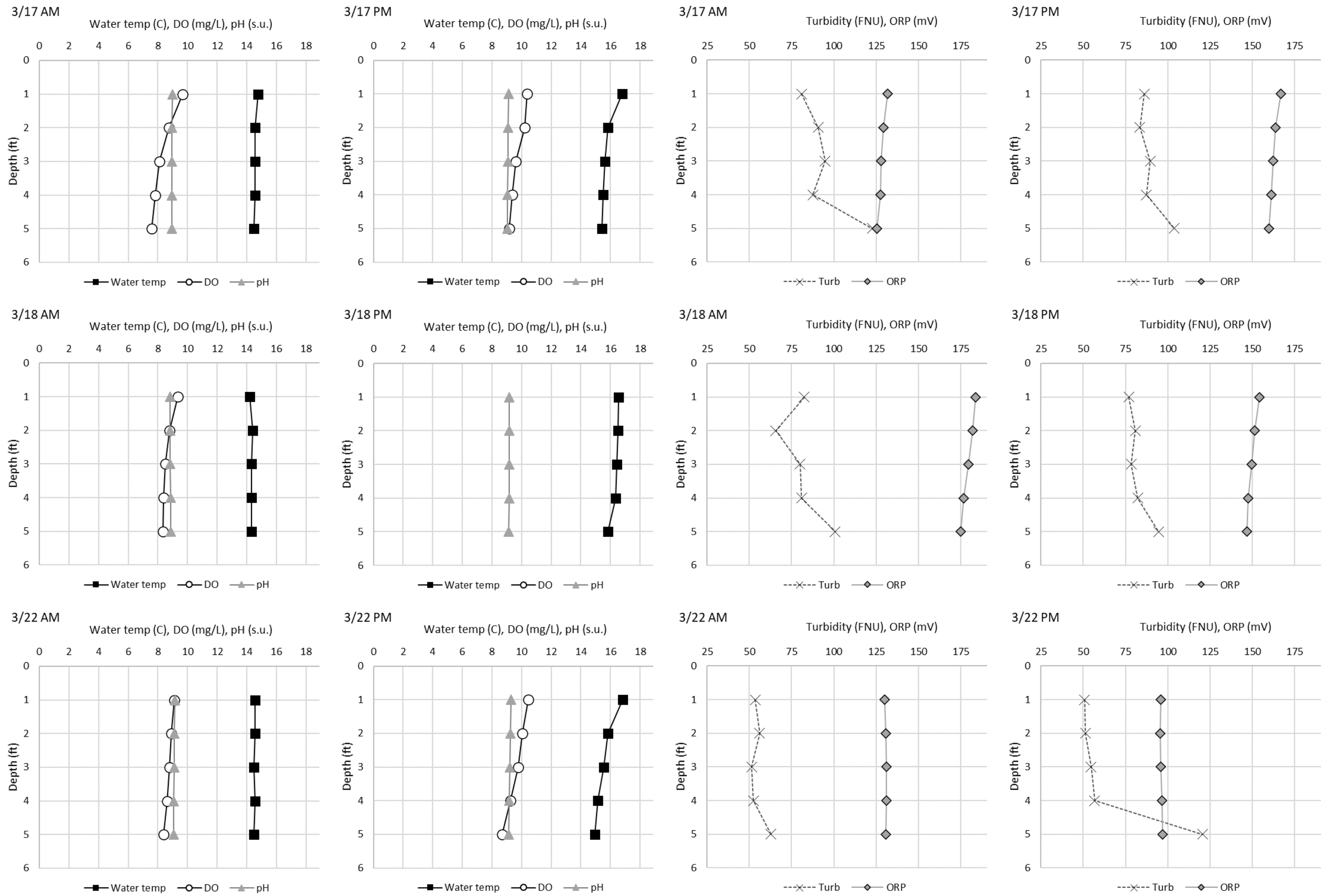

Figure A-5b. South lake (H-SL) *in situ* water quality morning and afternoon results for dates indicated. Note: 1) Negative DO readings during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced.

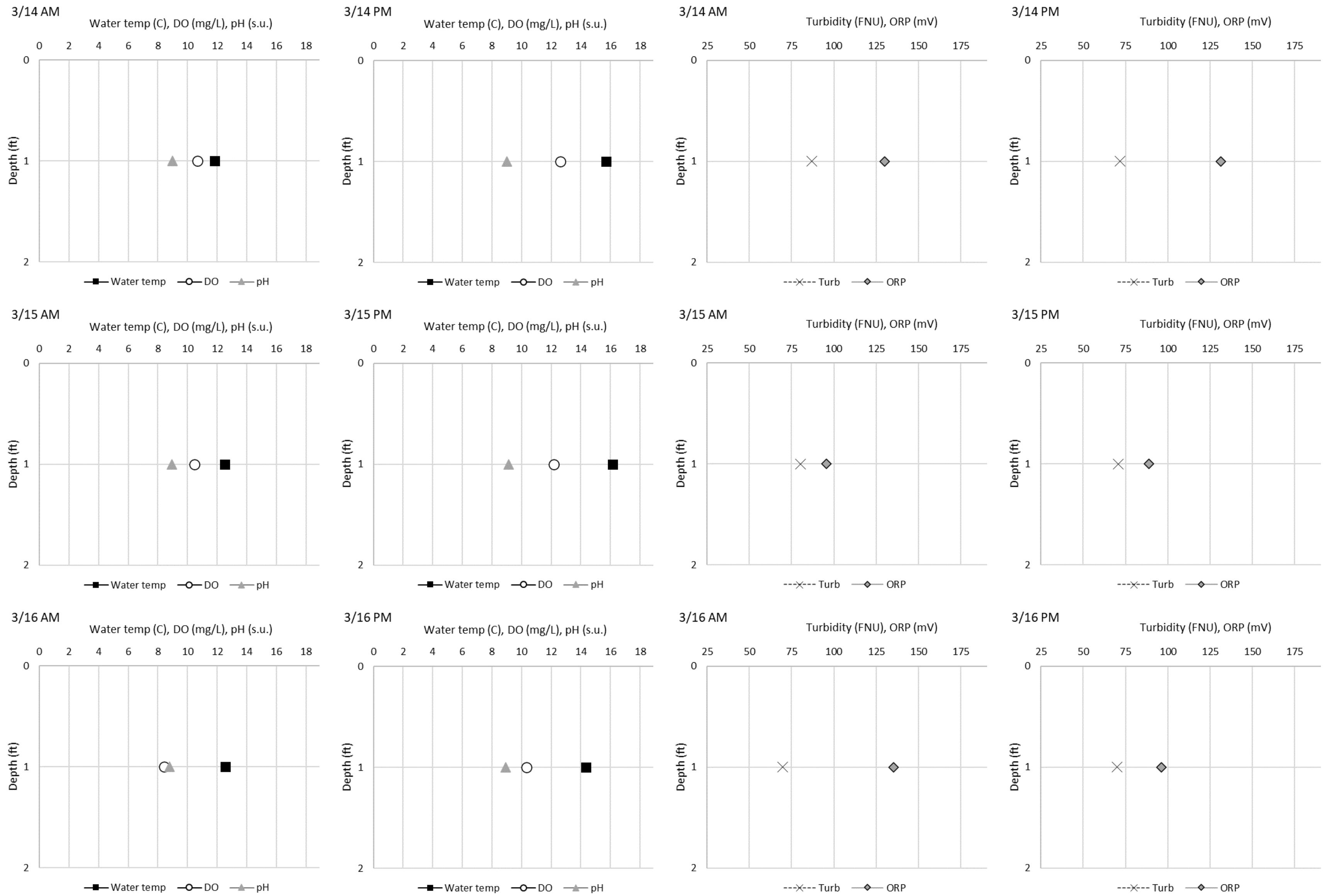

Figure A-6a. North shore (H-NS) *in situ* water quality morning and afternoon results for dates indicated.

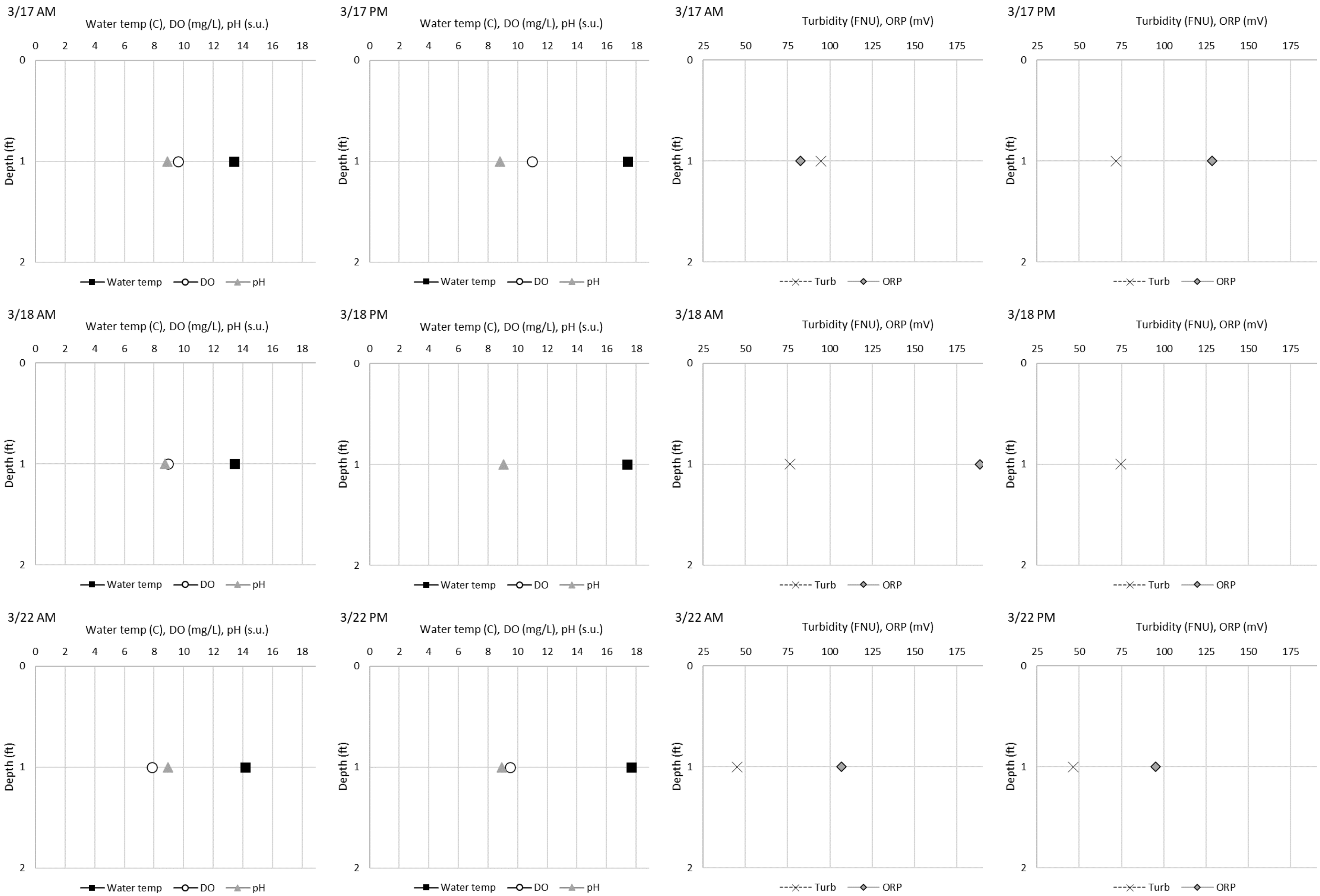

Figure A-6b. North shore (H-NS) *in situ* water quality morning and afternoon results for dates indicated. Note: 1) Negative DO readings during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced.

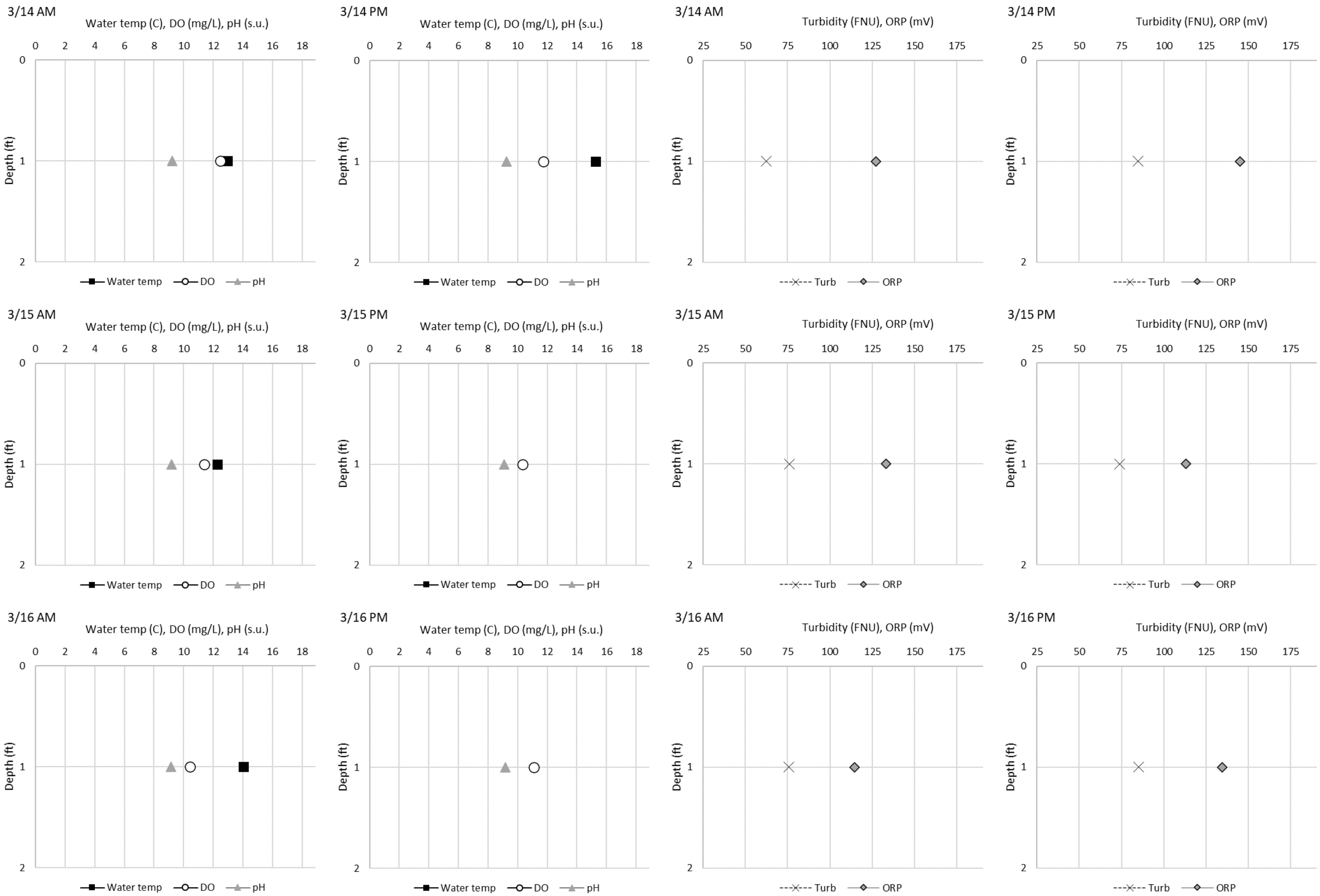

Figure A-7a. East shore (H-ES) *in situ* water quality morning and afternoon results for dates indicated. Note: Water temperature >20ºC during afternoon on 3/15 and 3/16 are not shown.

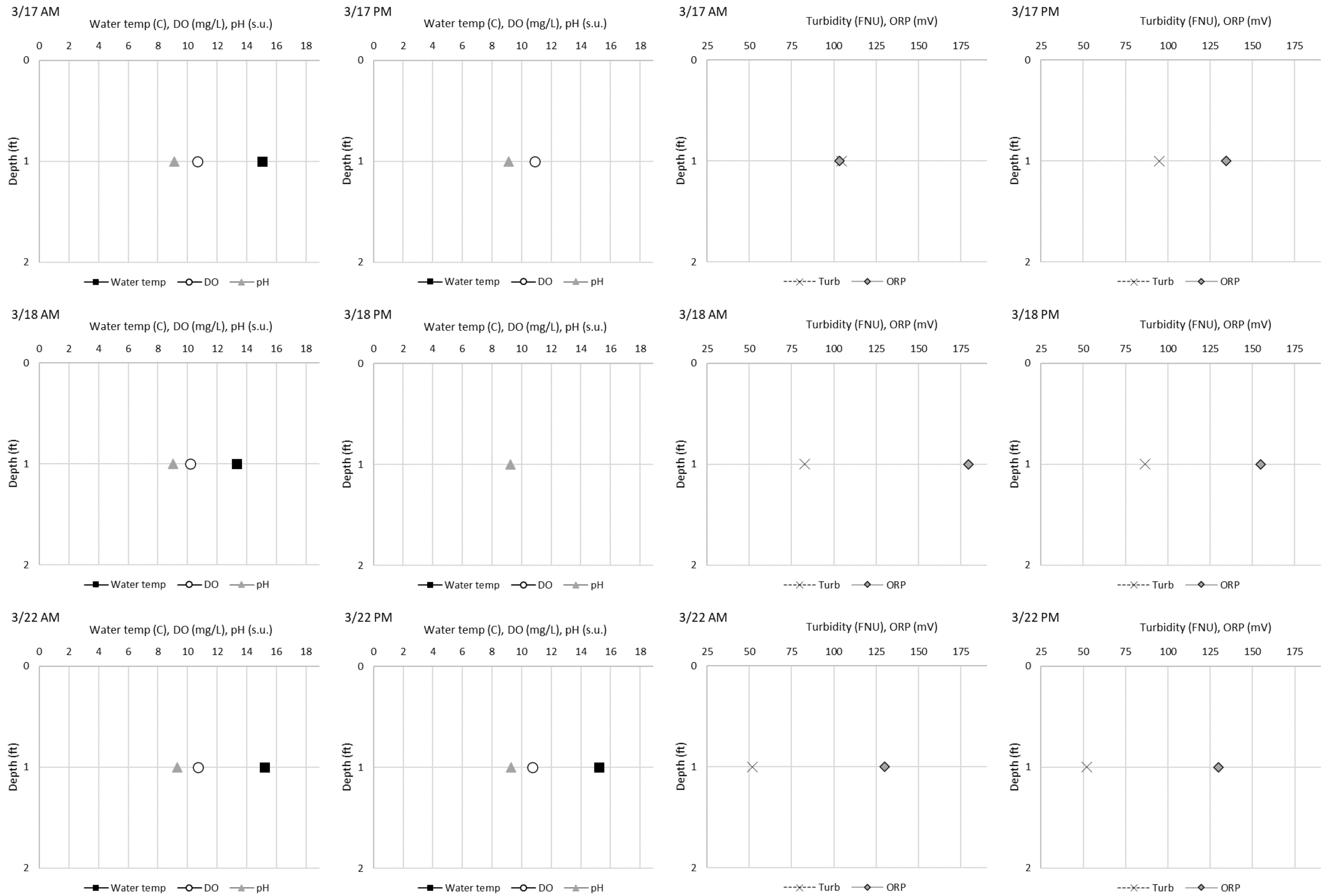

Figure A-7b. East shore (H-ES) *in situ* water quality morning and afternoon results for dates indicated. Notes: 1) Water temperature >20ºC during afternoon on 3/17 and 3/18 are not shown; 2) Negative DO readings during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced; 3) No afternoon readings were taken on 3/22.

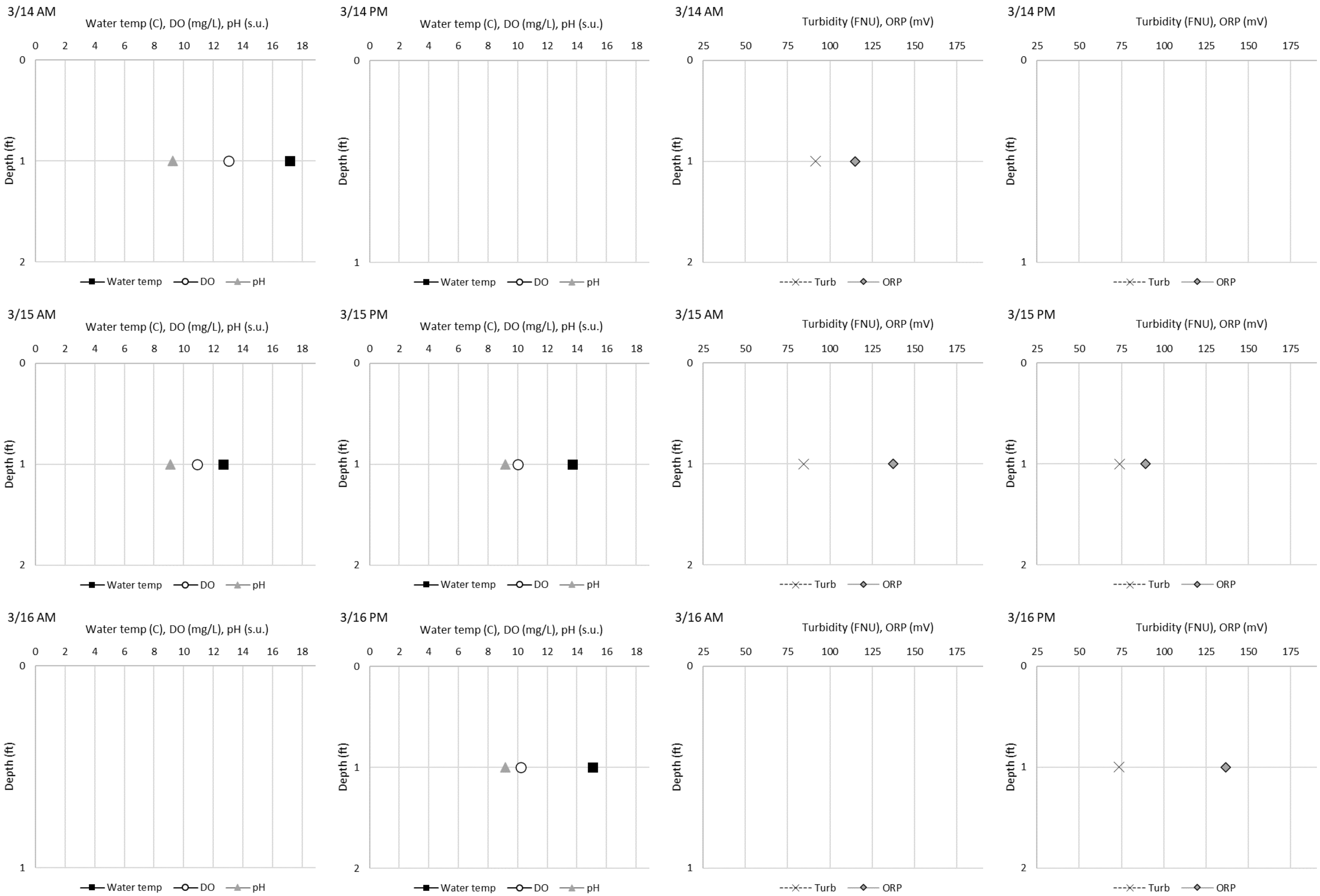

Figure A-8a. West shore (H-S) *in situ* water quality morning and afternoon results for dates indicated. Notes: 1) Not sampled during afternoon on 3/14; 2) Not sampled during morning on 3/16.

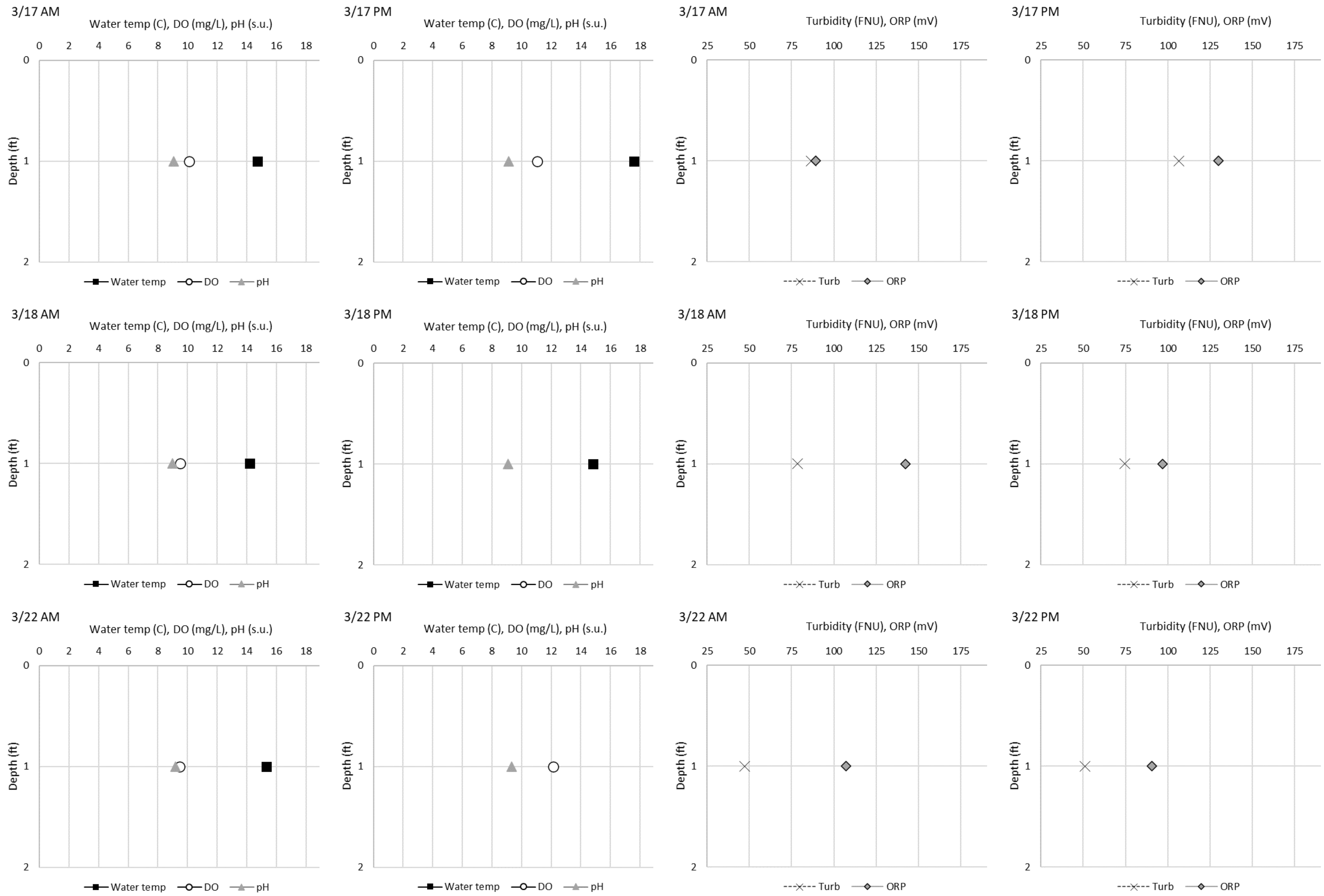

Figure A-8b. West shore (H-S) *in situ* water quality morning and afternoon results for dates indicated. Notes: 1) Negative DO readings during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced.

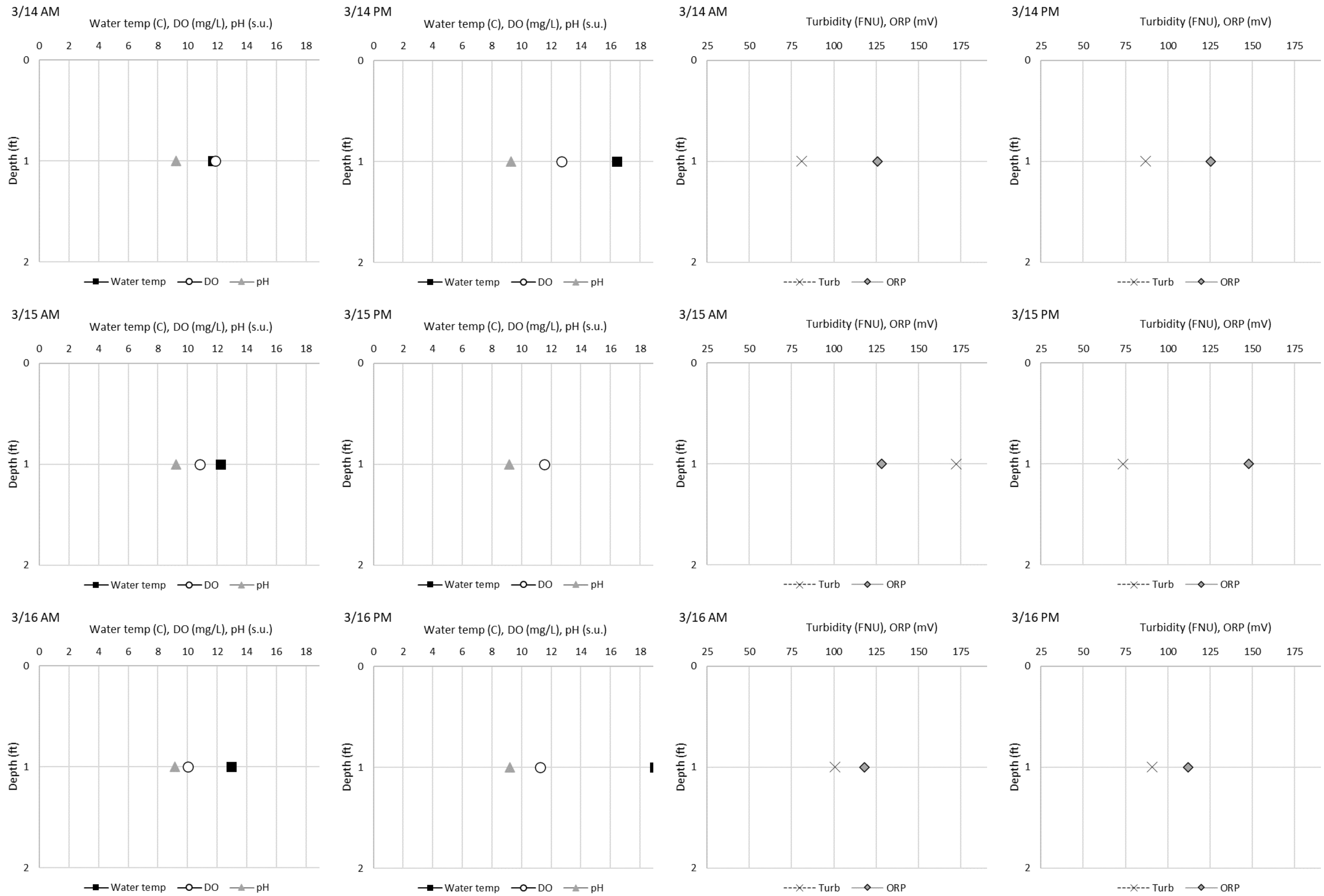

Figure A-9a. South shore (H-SS) *in situ* water quality morning and afternoon results for dates indicated.

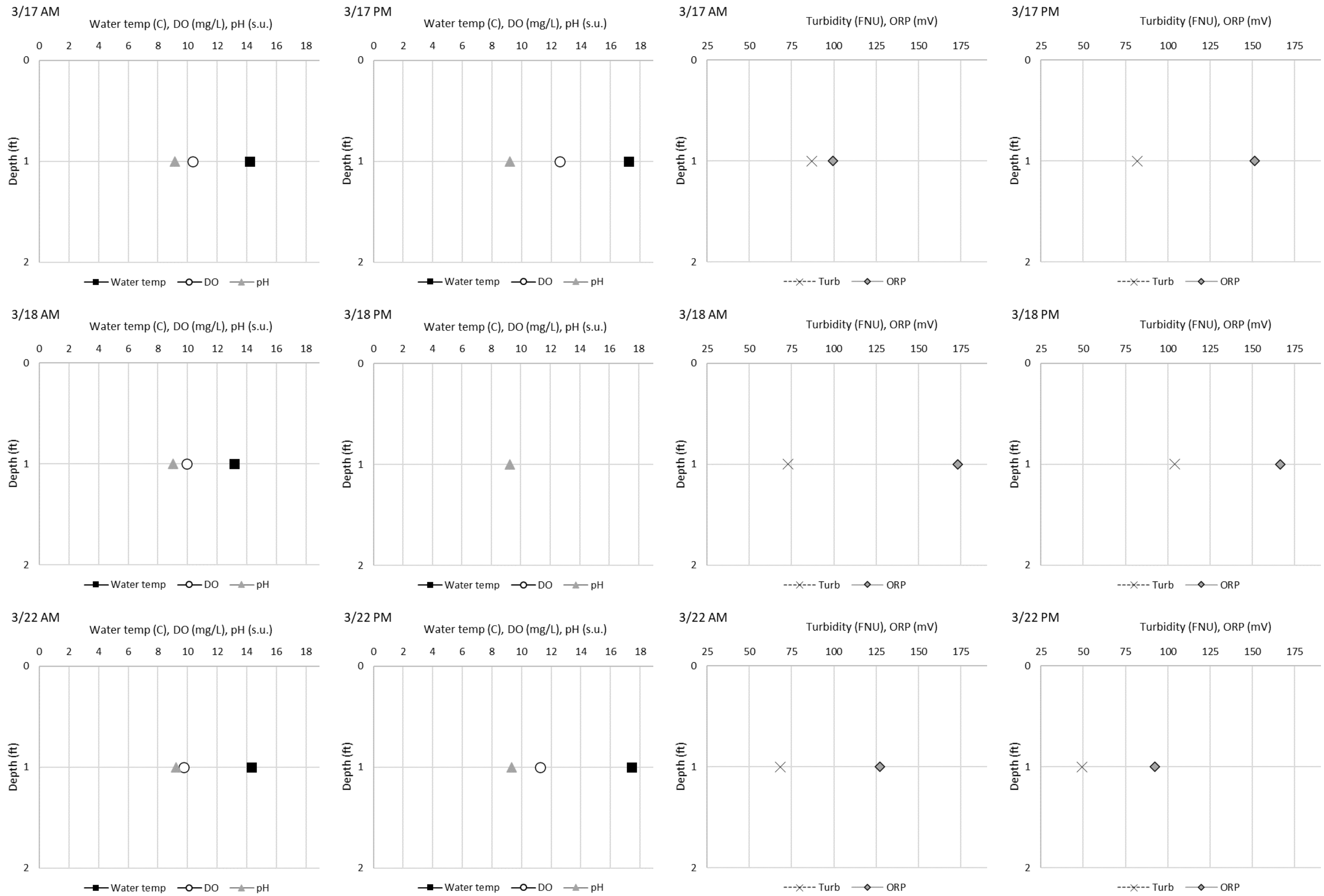

Figure A-9b. South shore (H-SS) *in situ* water quality morning and afternoon results for dates indicated. Note: 1) Negative DO readings during afternoon on 3/18 are not shown; these readings were due to a malfunctioning DO sensor that was subsequently replaced.

Appendix C

Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch

Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch

Vista Irrigation District
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INTRODUCTION

The Vista Irrigation District (District) located in northern San Diego County, California, occasionally requires the use of aquatic algaecides and/or aquatic herbicides as part of a larger program for managing water resources, maintaining designated beneficial uses, and controlling nuisance growths of algae and aquatic vegetation within the water system (Figure 1). Management of water resources via the occasional use of chemicals must be undertaken carefully so that their use does not impair the resources they strive to protect.

Regulatory Background

In March 2001, the State Water Resources Control Board (SWRCB) prepared Water Quality Order # 2001-12-DWQ which created Statewide General National Pollutant Discharge Elimination System (NPDES) Permit # CAG990003 for the discharges of aquatic herbicides to waters of the United States. The purpose of Order # 2001-12-DWQ was to minimize the areal extent and duration of adverse impacts to beneficial uses of water bodies treated with aquatic herbicides. The purpose of the general permit was to substantially reduce the potential discharger liability incurred for releasing water treated with aquatic herbicides into waters of the United States. The general permit expired January 31, 2004.

On May 20, 2004, the SWRCB adopted the statewide general NPDES Permit for Discharge of Aquatic Pesticides for Aquatic Weed Control in Waters of the United States #CAG 990005. Dischargers were required to have the general permit to perform aquatic herbicide applications. In May 2009, the general permit expired, but was administratively continued until November 30, 2013.

The Statewide General NPDES Permit (Permit) for Residual Aquatic Pesticide Discharges to Waters of the United States from Algae and Aquatic Weed Control Applications #CAG 990005 was adopted on March 5, 2013 and became available on December 1, 2013. The Permit requires compliance with the following:

- Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries in California, a.k.a. the State Implementation Plan, or SIP (2005);
- California Toxics Rule (CTR); and,
- Applicable Regional Water Quality Control Board (RWQCB) Basin Plan Water Quality Objectives (WQO's) San Diego Regional Board Basin Plan (SDRBBP 1994).

Coverage under the Permit is available to single dischargers and potentially to regional dischargers for releases of potential and/or actual pollutants to waters of the United States. Dischargers eligible for coverage under the Permit are public entities that conduct resource or pest management control measures, including local, state, and federal agencies responsible for control of algae, aquatic weeds, and other organisms that adversely impact operation and use of drinking water reservoirs, water conveyance facilities, irrigation canals, flood control channels, detention basins and/or natural water bodies.

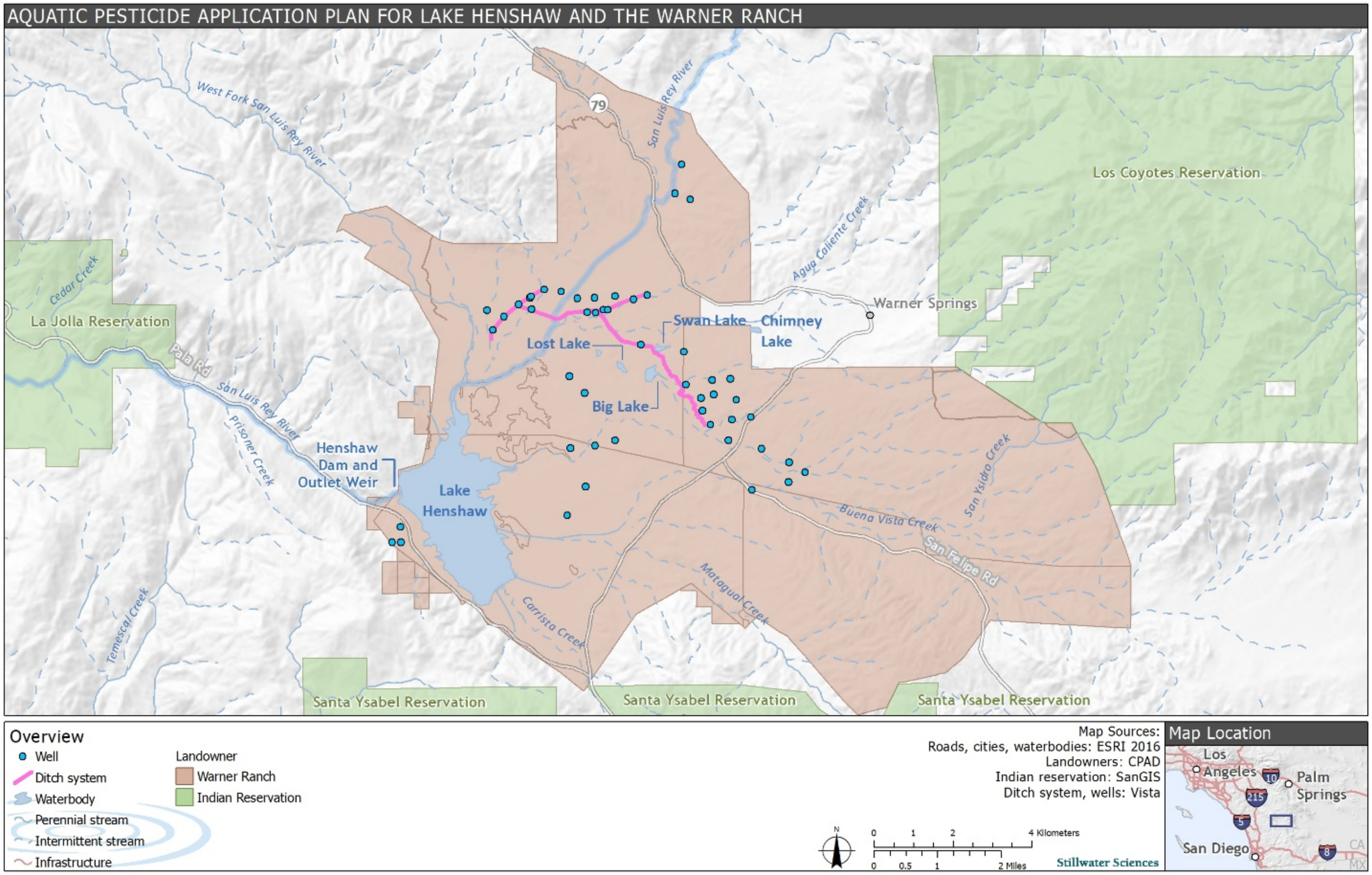


Figure 1. Vista Irrigation District water system, including Warner Ranch groundwater wells and ditches, Lake Henshaw, and Henshaw Dam.

The Permit does not cover indirect or non-point source discharges, whether from agricultural or other applications of pesticides to land, that may be conveyed in storm water or irrigation runoff. The Permit only covers algaecides and aquatic herbicides that are applied according to label directions and that are registered for use on aquatic sites by the California Department of Pesticide Regulation (DPR).

State Water Board Water Quality Order 2013-0002-DWQ (as amended by Orders 2014-0078-DWQ, 2015-0029-DWQ, and 2016-0073-EXEC) is the most up-to-date Water Quality Order for NPDES Permit # CAG990005 covering algae and aquatic weed control via aquatic pesticides in California. The Order expired in 2018, however it remains in effect until the State Water Board updates the Order.

Scope

This Aquatic Pesticides Application Plan (APAP) describes the best management practices (BMPs) and precautions that will be implemented to protect surface waters within the Warner Ranch, and Lake Henshaw, while maintaining sufficient storage in the lake to meet water delivery demands. This APAP addresses the application of algaecides/herbicides for controlling algae and aquatic weeds in the District's water supply system.

Specifically, this APAP contains the following eleven (11) elements:

1. Description of the water system to which algaecides and aquatic herbicides will be applied.
2. Description of the treatment area in the water system.
3. Description of types of weed(s) and algae that will be controlled and why.
4. Algaecide and aquatic herbicide products or types of algaecides and aquatic herbicides expected to be used and if known their degradation byproducts, the method in which they will be applied, and if applicable, the adjuvants and surfactants that will be used.
5. Discussion of the factors influencing the decision to select algaecide and aquatic herbicide applications for algae and weed control.
6. A listing of the gates or control structures to be used to control the extent of receiving waters potentially affected by algaecide and aquatic herbicide application and an inspection schedule of those gates or control structures to ensure they are not leaking.
7. Description of any applicable Short Term Seasonal Exceptions.
8. Description of procedures used to prevent sample contamination from persons, equipment, and vehicles associated with algaecide and aquatic herbicide application.
9. Description of the BMPs to be implemented:

a. Measures to prevent algaecide and aquatic herbicide spill and for spill containment during the event of a spill.
b. Measures to ensure that only an appropriate rate of application consistent with product label requirements is applied for the targeted weeds or algae.
c. The District's plan to educate its staff and algaecide and aquatic herbicide applicators on how to avoid any potential adverse effects from the algaecide and aquatic herbicide applications.
d. Discussion on planning and coordination so that designated beneficial uses of the water are not impacted during the treatment period; and,
e. A description of measures that will be used for preventing fish kill when algaecides and aquatic herbicides will be used for algae and aquatic weed controls.

10. An examination of possible alternatives to algaecide and aquatic herbicide use to reduce the need for applying algaecides and herbicides.
    a. An evaluation of the following management options, in which the impact to water quality, impact to non-target organisms including plants, algaecide and aquatic herbicide resistance, feasibility, and cost effectiveness is considered:
        i. No action
        ii. Prevention
        iii. Native species establishment
        iv. Mechanical or physical methods
        v. Cultural methods
        vi. Biological control agents
        vii. Algaecides and aquatic herbicides

    If there are no alternatives to algaecides and aquatic herbicides, dischargers shall use the minimum amount of algaecides and aquatic herbicides that is necessary to have an effective control program and is consistent with the algaecide and aquatic herbicide product label requirements.

    b. Using the least intrusive method of algaecide and aquatic herbicide application; and
    c. Applying a decision matrix concept to the choice of the most appropriate formulation.

11. Monitoring provisions including sampling procedures, record retention, device calibration, mapping, certification, and reporting schedules.

This APAP is organized to address the aforementioned elements.

1.0 WATER SYSTEM

The District owns and operates the 43,000-acre Warner Ranch in the northern portion of San Diego County (Figure 1). Its primary interest in Warner Ranch is water production, and it operates wells, ditches, and the 52,000-acre-foot Lake Henshaw to produce an average annual yield of 13,500 acre-feet of water for use by the District, the City of Escondido, and the Rincon Band of Indians for the period 1953 through 2020. During years of inadequate surface runoff, the District pumps water from the Warner Basin aquifer into 6.2 miles of ditches for delivery to Lake Henshaw (Figure 2). Water in the ditch system flows by gravity into an approximately 1,800-foot unnamed stream reach (Figure 3) before entering the ephemeral San Luis Rey River upstream of Lake Henshaw, and then flows for approximately one mile before entering the lake.

Figure 2. Groundwater Wells Discharging to Ditch at Location 1 on the Warner Ranch.

Figure 3. Terminus of the Warner Wellfield ditch system (left), which flows downstream into the San Luis Rey River and then Lake Henshaw (right).

Lake Henshaw was artificially created in 1923 with the building of the Henshaw Dam, an earth dam 123 ft (37m) tall, and 650 (200m) long. The original capacity of Lake Henshaw was approximately 200,000 acre-feet. The spillway was lowered in 1981 to address seismic concerns, which brought the reservoir capacity to its current 52,000 acre-feet and 2,256 lake surface acres (at full capacity).

Lake Henshaw designated beneficial uses include the following (San Diego Regional Water Quality Control Board 2012):

- municipal and domestic water supply
- agricultural water supply
- industrial process and service supply
- freshwater replenishment
- rare, threatened, or endangered species habitat
- hydropower generation
- warm freshwater habitat
- contact[1] and noncontact recreation
- wildlife habitat

Lake Henshaw water supply includes tribal water rights for the Indian Water Authority.

Water is released from the Lake Henshaw outlet at the base of Henshaw Dam through valves and into an open concrete channel that flows for approximately 125 feet and across an outlet weir before entering the San Luis Rey River (Figure 4).

[1] Fishing from shore or boat is permitted in Lake Henshaw, but other water contact recreational (REC-1) uses are prohibited.


Figure 4. Henshaw Dam outlet channel to the San Luis Rey River.

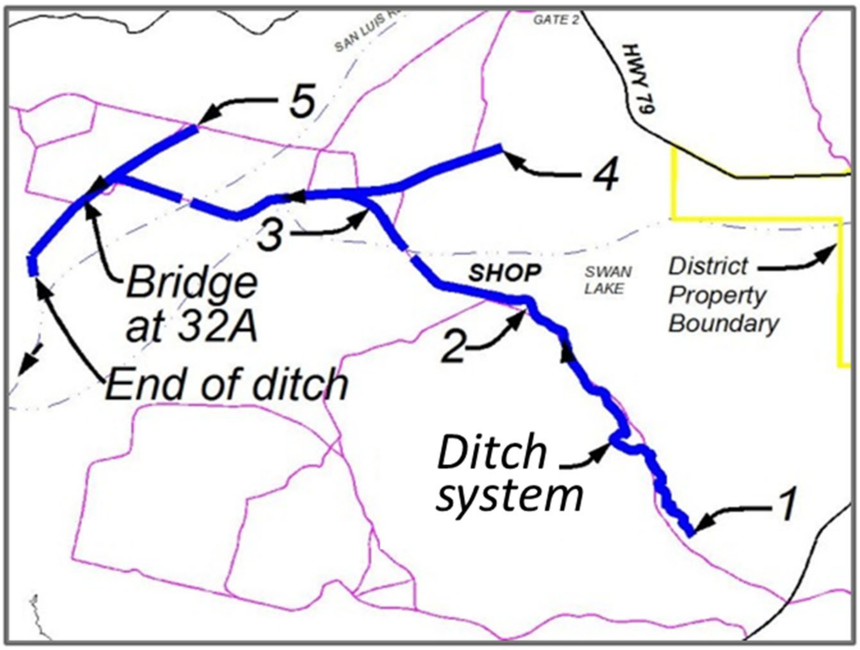
2.0 TREATMENT AREA

2.1 Warner Ranch

The Warner Ranch *treatment area* is represented by the 6.2-mile length of the ditch system. Aquatic algaecides/herbicides will be applied to the Warner Ranch ditches at strategic treatment locations. *Treatment locations* are defined as the specific sites at which aquatic pesticides will be directly applied. For the Warner Ranch, up to five tentative treatment locations are shown in Figure 5, located at the start of each ditch and at two locations along the main ditch length. However, spot treatments just upstream of areas where aquatic macrophytes and/or filamentous algae have established are likely to be most effective for maintaining the ditch segments and thus the ultimate treatment locations will depend on growth of aquatic macrophytes and/or filamentous algae in the ditch system. While there are multiple small off-channel watering ponds (e.g., see Swan Lake, Big Lake, Lost Lake in Figure 1) for cattle within the Warner Ranch, one of these ponds (Swan Lake) is supplied by the ditch system and once in the pond, surface water containing macrophytes or algae does not move back into the ditches. Thus, the off-channel watering ponds are not part of the treatment area for this APAP.

2.2 Lake Henshaw

Treatments to Lake Henshaw will be applied to the lake surface or sub-surface at various locations to be determined on an as-needed basis, with the treatment area extending 2,256 acres at full capacity. Treatment locations in Lake Henshaw are defined as the specific sites at which aquatic pesticides will be directly applied. Note that for some aquatic algaecide/herbicide products, as specified on the label, treatment locations in the lake cannot extend beyond one-third of the lake surface area during a single treatment event.



Tentative Treatment Locations	Position (Latitude / Longitude)
Location 1	N 33° 15' 26.1" W 116° 40' 43.9"
Location 2	N 33° 16' 0.56" W 116° 41' 11.5"
Location 3	N 33° 16' 55.9" W 116° 42' 33.8"
Location 4	N 33° 17' 13.7" W 116° 41' 44.6"
Location 5	N 33° 17' 18.9" W 116° 43' 21.5"

Figure 5. Warner Ranch ditch system treatment area with five tentative treatment locations for aquatic macrophytes and/or filamentous algae. Blue line indicates the 6.2-mile length of the ditch system.

3.0 WEED AND ALGAE GROWTH IN WARNER RANCH DITCHES AND LAKE HENSHAW

3.1 Warner Ranch Ditches

The majority of the water conveyance ditches on the Warner Ranch are not covered, such that aquatic weeds and algae grow throughout the year under direct sunlight. A 4,500-foot length of ditch was recently covered by installing reinforced concrete pipe within a section of open ditch that had been damaged during heavy precipitation and flooding in February of 2019. Aquatic macrophyte and filamentous algae growth limits the hydraulic capacity of open ditches delivering water to Lake Henshaw and results in the need for periodic mechanical removal that is costly, is not consistently successful at restoring unimpeded flows, and can exacerbate root damage to ditch linings through physical disturbance.

Figure 6: Weeds growing on the ditch downstream of the trash rack.

Although not taxonomically identified, there are aquatic macrophytes and filamentous algae that grow in the Warner Ranch ditch system and require control. Filamentous algae masses and uprooted aquatic weeds that break off and float down the ditches (either open or covered lengths) are caught by one of the eight trash racks on the Warner Ranch (Figure 6). Some portions of the open ditches are temporarily covered by chain link fencing that is laid across the ditch to keep tumbleweed from becoming lodged in the ditches. The lengths of fencing and captured tumbleweed are periodically removed as part of normal maintenance operations. Trash racks are distributed throughout the ditch system to prevent the obstruction of ditch siphons. Maintenance requires that all trash racks routinely be cleaned; otherwise, algae and weeds would block ditches and impede flow.

Aquatic macrophyte and filamentous algae growth have not been observed at the terminus of the Warner Ranch ditch system (Figure 3), which flows into the San Luis Rey River approximately one mile upstream of Lake Henshaw.

3.2 Lake Henshaw

The main impacts to Lake Henshaw designated beneficial uses from nuisance growths of filamentous, benthic, and/or planktonic algae are related to drinking water quality (decaying organic matter, algal toxins), drinking water supply (clogging of screens and contamination of water treatment basins), aesthetics, and recreational fishing (low dissolved oxygen, high pH, algal toxins). Algae control on an as-needed basis along the lake shoreline and within the main body of the lake is intended to reduce or eliminate the occurrence of harmful algal blooms (HABs) that produce cyanotoxins at levels that exceed California voluntary posting guidance for planktonic sources of microcystin and anatoxin-a in recreational inland surface waters. Cyanobacteria, a type of photosynthetic bacteria also known as blue-green algae, are often the cause of algal blooms in freshwater and occasionally in marine water. Cyanotoxins can have harmful effects on people, fish, birds and livestock. The human illnesses caused by HABs though rare, can be debilitating, or even fatal. The most common cyanobacterial HAB toxins in the U.S. are microcystins, a group of liver toxins that can cause gastrointestinal illness in humans, and mortality in pets, livestock, and wildlife.

The volume of water in the area targeted for cyanobacteria control will vary based on water levels in the lake, which in drier years range approximately 2,500 to 6,000 acre-feet and in wetter years range approximately 25,000 to 52,000 acre-feet. The types of cyanobacteria to be controlled in the District's system may include, but are not limited to the following:

- *Microcystis sp.*
- *Planktothrix sp.*
- *Snowella sp.*
- *Aphanizomenon sp.*
- *Woronichinia sp.*
- *Dolichospermum sp.*

The presence of algae, cyanobacteria, and other aquatic weeds reduces the water quality and clarity.

4.0 AQUATIC ALGAECIDES AND HERBICIDES EXPECTED TO BE USED AND APPLICATION METHODS

If needed, the District proposes to apply various forms/formulations of aquatic herbicides and algaecides (see Appendix B) of various products with active ingredients such as copper, Endothall, Diaquat, Imazamox, and Peroxyhydrate. When the rate of aquatic macrophyte and/or filamentous algae in the Warner Ranch ditch system and/or filamentous, planktonic and/or benthic algae growth in Lake Henshaw indicates the need for an aquatic herbicide or algaecide (see also Section 5.1), the District will begin a program of dosing the ditches and/or lake with one or more of these chemicals, as needed. The aquatic herbicides and algaecides will be applied at the strategic points within the treatment area (see also Section 2.0) and the frequency of dosing may be adjusted throughout the season based on weather conditions, actual algae growth patterns, and the treatment area calculations. Applications will be made by a drip system, and/or surface and subsurface application methods, dependent upon product type and nuisance aquatic macrophyte or algal/cyanobacteria species at a rate consistent with the label requirements and/or California Department of Pesticide Regulation licensed Pest Control Adviser (PCA) recommendations. As noted in Section 2, label requirements for some aquatic algaecide/herbicide products (e.g., Endothall) include limitations on the extent of treatment locations in a lake to no more than one-third of the lake surface area for a single treatment event. The District and/or its agent may use a variety of application vehicles or vessels including boats to apply algaecides and herbicides. Application techniques may include injection, granular spreaders or liquid sprays. Combined with the need to hold, safely transport and properly apply algaecides and aquatic herbicides, the District and/or its agent will utilize techniques that are the least intrusive as possible.

As required, aquatic-labeled adjuvants may be used to enhance the efficacy of an herbicide. All herbicide applications will be made in accordance with the product label. Table 1 summarizes the algaecides and aquatic herbicides that may be used by the Vista Irrigation District.

Table 1: Algaecides/Aquatic Herbicides That May be Used Within the Treatment Area

Herbicide	Application Method	Adjuvant	Primary Degradation Products
Copper – Chelated	Sprayer, injection boom, granular spreader	Not Applicable	None
Copper Sulfate	Sprayer or injection boom	Not Applicable	None
Diquat Dibromide	Sprayer or injection boom	Aquatic labeled adjuvants	Teir 2 organic products
Endothall	Sprayer, injection boom or granular spreader	Not Applicable	Glutamic Acid
Glyphosate	Power or backpack sprayer	Aquatic labeled adjuvants	Aminomethyl-phosphonic acid
Sodium Carbonate Peroxyhydrate	Boom injector or spreader	Not Applicable	Water, bicarbonate
Ammonium Salt of Imazamox	Sprayer, injection boom, power or backpack sprayer	Aquatic Labeled Adjuvants	Nicotinic acid and di- and tricarboxylic acids

5.0 DECISION TO USE AQUATIC PESTICIDES

The decision to treat aquatic vegetation and algae using aquatic pesticides is best made within the framework of Integrated Pest Management (IPM) techniques. One of the primary operational goals of an IPM program is to establish a general and reasonable set of control measures that not only aid in managing aquatic vegetation populations, but also address public health and safety, economic, legal, and aesthetic requirements. An IPM control threshold level is the point at which action should be taken to control aquatic vegetation before the water body is significantly impacted; moreover, established threshold levels for implementing selected control measures may change based on public expectations. A central feature of IPM is to determine when control measures are absolutely necessary and when they are not, for the presence of some aquatic vegetation species may be a sign of a well-balanced, flourishing ecosystem. Examples of when or how thresholds are met are when algae or aquatic vegetation causes complaints with odor or creates a nuisance or safety concerns with water contact activities. Typical problems associated with aquatic vegetation or algae blooms are adverse impacts to water quality and nuisance odors. If vegetation or algae equals or exceeds a threshold, a control method is implemented. Control methods may include mechanical, cultural controls, biological, and/or chemical, consistent with the IPM techniques. Algaecide and aquatic herbicide use may or may not be employed as a last resort control method, and is considered a critical part of the IPM program. For some aquatic weed varieties, herbicides offer the most effective control; sometimes, they may be the only control available. The District's the decision to use an algaecide or aquatic herbicide may be informed by the recommendation of a California Department of Pesticide Regulation licensed Pest Control Adviser (PCA). The District and/or PCA considers a variety of control options that may include mechanical and/or cultural techniques that alone or in combination with algaecide or aquatic herbicide use are the most efficacious and protective of the environment.

Algaecide and aquatic herbicide applications may be made prior to IPM control threshold exceedance. For example, based on predicted growth rate and density, historical algae and aquatic weed trends, weather, water flow, and experience, aquatic weeds or algae may reasonably be predicted to cause future problems. Accordingly, they may be treated soon after emergence or when appropriate based on the algaecide and aquatic herbicide to be used. Even though algae and aquatic weeds may not be an immediate problem at this phase, treating them before they mature reduces the total amount of algaecide and aquatic herbicide needed because the younger aquatic weeds and more susceptible and there is less biomass to target. Furthermore, treating aquatic weeds and algae within the ideal time frame of its growth cycle ensures that the selected control measures will be most effective. Managing aquatic weed populations before they produce seeds, tubers or other reproductive organs is an important step in a comprehensive aquatic weed control program. Generally, treating algae or aquatic weeds earlier in the growth cycle results in fewer controls needed and less total herbicide use. Selection of appropriate algaecide and aquatic herbicide(s) and rate of application is done based on the identification of the algae and aquatic weed, its growth state and the appearance of that algae or aquatic weed on the product label. Further, the quantity of algaecide and aquatic herbicide required for an application is determined by the District or PCA in accordance with label directions. The rate at which an algaecide and aquatic herbicide is used is highly variable and depends on the type, time of year, location, and density and type of aquatic weeds, water presence, and goal of the application. All these factors are considered by the District and/or PCA prior to making a decision regarding application.

Consistent with general IPM practices, the District's use of aquatic herbicides and/or algaecides to control nuisance aquatic macrophytes, filamentous algae, planktonic algae, benthic algae, and/or cyanobacteria in the Warner Ranch and Lake Henshaw water system will be reserved for conditions when this control measure is necessary. If algaecides and aquatic herbicides are used, the District will use the minimum amount of algaecides and/or aquatic herbicides necessary to have an effective control program and that is consistent with the algaecide and aquatic herbicide product label requirements. During the summer and fall months, generally between May and November, the growth rate of aquatic weeds and/or filamentous algae can outpace the District's ability to effectively remove them from by physical means, and in the case of planktonic and benthic algae, the natural growth forms of these organisms can preclude physical removal and subsequent disposal of large amounts of biomass, such that there are no feasible alternatives to algaecides and aquatic herbicides for effective management of HABs. During high growth rate periods, the District proposes to apply the aquatic herbicides and/or algaecides listed in Table 1 at up to five locations along the Warner Ranch ditch system (see Figure 5) as well as in Lake Henshaw. As treatment of algae, cyanobacteria, and/or aquatic weeds earlier in the growth cycle is expected to result in fewer controls needed and less total herbicide use, the District may also apply the aquatic herbicides and/or algaecides listed in Table 1 during spring months (March and April).

5.1 Thresholds for Using Pesticides as a Control Measure

As seasonal insolation (i.e., the amount of solar radiation reaching a given area) affects filamentous algal and aquatic weed growth throughout the year, the District maintains a year-round monitoring program for nuisance algal and aquatic weed growth at the Warner Ranch to ensure sufficient water supply to Lake Henshaw. Especially during summer months, the increased insolation, extended daylight hours, and warmer temperatures accelerate the growth of filamentous algae and aquatic weeds in the District's ditch system. The District tolerates moderate growth of filamentous algae and aquatic weed patches until the number of patches and/or the extent of the patches threaten to break off and float down the ditches (either open or covered lengths) where they will be caught by one of the eight trash racks and impede flow behind the racks. The District will apply chemical treatment when this condition appears imminent, and when alternatives to chemical control would not be reasonably effective. Limiting growth to a minimal size helps maintain wellfield production and keep maintenance down.

With regard to Lake Henshaw, the District also monitors algae, cyanobacteria, and aquatic weeds year-round to maintain adequate water quality. The District tolerates moderate filamentous algae and aquatic weed growth at levels that do not create nuisance conditions in the lake, including but limited to impeding fishing boat access, fouling fishing gear, and degrading water quality (e.g., dissolved oxygen, pH). The District will apply chemical treatment when filamentous algae or aquatic weed growth creates nuisance conditions, and when alternatives to chemical control would not be reasonably effective. The District will apply algaecides to control HABs in Lake Henshaw as needed and when alternatives to chemical control would not be reasonably effective at reducing algal toxin concentrations below California Tier 1 voluntary posting guidance for planktonic sources in recreational inland surface waters (i.e., total microcystins = 0.8 ug/L and anatoxin detection).

Alternatives to chemical control are discussed in Section 10.

6.0 CONTROL STRUCTURES

The District has no gates or control structures within the Warner Ranch ditch system that supplies groundwater to Lake Henshaw. The ditches remain free-flowing throughout the year, except during periods when groundwater pumping is not occurring.

As applicable or necessary, the controllable valves at the Henshaw Dam intake tower will be closed during an algaecide or aquatic herbicide application to control the extent, if any, that the receiving waters of the San Luis Rey River would be affected by residual algaecides or aquatic herbicides.

7.0 SHORT TERM OR SEASONAL EXCEPTION

The District has not applied for a short term or seasonal exception.

8.0 PROCEDURES USED TO PREVENT SAMPLE CONTAMINATION

Collection of samples for the monitoring program described in Section 11 will not be undertaken in close proximity to algaecide or aquatic herbicide application equipment and will be preferably upwind of the application point. Sampling will be done in a manner that prevents contact with algaecide or aquatic herbicide application equipment, containers, or personal protective equipment (PPE). Care will be taken by samplers to minimize contact with any treated water, vegetation, or application equipment.

It is possible that actual field conditions may require a modification of the procedures outlined herein. Specifically, water levels, weather, other environmental parameters and hazards including stream flow, rainfall, and wave action may pose access and/or sampling problems. In such instances, variations from standard procedures and planned sampling locations and frequencies will be documented by means of appropriate entry into the field logbook.

In the event that sampling equipment will be used in more than one location, the equipment will be thoroughly cleaned with a non-phosphate cleaner, triple-rinsed with distilled water, and then rinsed once with the water being sampled prior to its first use at a new sample collection location.

9.0 DESCRIPTIONS OF ALGAECIDE AND AQUATIC HERBICIDE BMPs

The District or its agent will utilize the following BMPs when applying aquatic herbicides and/or algaecides:

- Structural BMPs will prevent accidental spillage and contain any chemicals from coming into contact with the surrounding environment. Structural BMPs will include:
    - Making spill kits available in the copper sulfate storage locker.
    - Providing spill containment precautions in the copper sulfate storage locker.
    - Implementing spill prevention during transportation of aquatic herbicides and algaecides to the point(s) of application (e.g., spill tray, restraining system, no open containers).
    - Applying aquatic herbicides and/or algaecides under favorable weather conditions (e.g., calm-to-light wind conditions and no precipitation) to reduce exposure to the surrounding habitat in accordance with label recommendations.
    - Utilizing closed system application equipment when possible.

- To ensure that the rate of algaecide/herbicide application is consistent with product label requirements for the targeted weeds or algae, total ditch flow will be assessed before each application to determine proper dosing, and ditch conditions will be assessed to verify the need for treatment. Only areas deemed in need of aquatic plant maintenance will be treated.
- Provide periodic training sessions (as necessary) for District staff conducting treatment.
- District/Contractor Staff will be required to use proper protection from algaecide/herbicide exposure, including the use of gloves, aprons, and ventilation masks (if needed). The District/agent will take all precautions necessary to ensure employees are suitably protected from exposure.
- Before each algaecide/herbicide treatment application, the District, City of Escondido, San Luis Rey Indian Water Authority and other interested parties that utilize the watershed and discharge waters from Lake Henshaw will be notified.
- Mitigating measures to prevent potential fish kills will include:
    - Application rates consistent with label requirements and/or PCA recommendations.

- Pre-application water sampling to verify pH and DO, to determine if an algaecide/herbicide application could adversely affect oxygen levels.
- Application of products that have been approved by the DPR or an exemption has been granted.

10.0 POSSIBLE ALTERNATIVE CONTROL METHODS

10.1 Warner Ranch

The various alternatives to using aquatic algaecides/herbicides for controlling of algae and weed growth in the Warner Ranch ditches include the following:

- **No Action —** As feasible, no action is used as a control measure prior to reaching a threshold for using aquatic pesticides as a control measure (see also Section 5.1).
- **Physical Removal** — District staff manually clean ditches and trash racks to remove aquatic weed and filamentous algae year-round when the well field is in use. Physical removal is more effective in the winter months and may not keep up with plant and algae growth from May to November each year (see also Section 3.0).
- **Alternative Chemical Controls —** The only algaecides registered for use in ditches are endothall, 2-4D, Diaquat, peroxyhydrate, copper, copper sulfate, and chelated copper compounds, thus there are no currently available alternative chemical controls.
- **Constructing Covered Ditches** — Wholesale replacement of the existing 6.2 miles of open water supply ditches at the Warner Ranch with covered ditches or buried piping would not be economically feasible due to the significant cost of construction and environmental permitting associated with the project scale. The projected cost of this alternative is several million dollars, not including environmental mitigation. Due to the potential presence of endangered species on the Warner Ranch (i.e., Stephens kangaroo rat [*Dipodomys stephensi*] and the arroyo toad [*Anaxyrus californicus*]), it is likely that such a project would require extensive environmental mitigation at significant additional cost. While sections of the existing supply ditches on the Warner Ranch may be replaced over time, aquatic herbicide and/or algaecide applications may be needed in sections that remain open for the foreseeable future.
- **Cease Pumping Operations** — Wellfield operations are suspended when practical, typically following normal or above-normal rainfall/runoff seasons resulting in above normal reservoir storage. When natural watershed runoff provides sufficient storage volume in Lake Henshaw to meet annual delivery objectives of local water, the District

minimizes or otherwise halts groundwater pumping into the ditch system. This strategy allows for periods when maintenance in the ditch system is greatly reduced, and algaecide use is not needed. This strategy is not suitable for periods of low runoff production, however, due contractual obligations to produce water for the Rincon Band of Indians and the cost of replacement water for the District and the City of Escondido.

- **Biological Control Agents —** Not considered practical due to an inability to hold the waters so that biological agents can be effective (i.e., microbes/enzymes, zooplankton, and macrophytes).

10.2 Lake Henshaw

The various alternatives to using aquatic algaecides/herbicides for controlling algae and weed growth in Lake Henshaw and the outlet weir are listed below.

- **No Action —** As feasible, no action is used as a control measure prior to reaching a threshold for using aquatic pesticides as a control measure (see also Section 5.1).
- **Prevention: Habitat Modification —** While aquatic macrophytes are currently not a problem in Lake Henshaw, this alternative may become necessary if they were to establish at nuisance levels in the future. After the removal of non-native nuisance or invasive species, the introduction and re-establishment of native species may be successful. This technique is intended to provide competition for non-desirable species and reduce the need for aquatic weed abatement only around the perimeter. This approach would not directly affect algae populations.

  The District may also consider other habitat modifying techniques appropriate for the individual target areas: for example, dredging, oxygenation or aeration, shading with dyes, and bio-manipulation. In areas where sedimentation has significantly impacted the capacity of the water body, dredging can increase the water volume; reduce organic matter generated in the water body; and remove nutrient-containing sediment. Aeration, oxygenation and mixing are methods that can mechanically add oxygen directly to the water, and they can result in the reduction of nuisance algae growth.

  Shading the water column using non-toxic, inert dyes can reduce unwanted submerged plants and algae. Use of dyes works on algae and submerged vegetation by limiting their ability to photosynthesize when the dye is present, but is not a long-term solution and is generally not applicable for drinking water sources.

  Bio-manipulation utilizes various natural mechanisms that can reduce suspended algae, and this method involves increasing biological controls in the habitat. The biological controls are typically done by top-down or bottom-up changes to the food-web structure aimed at increasing populations of algae-consuming zooplankton. Bio-manipulation may be more efficient when used in conjunction of other habitat modification methods.

- **Native Species Establishment —** While aquatic macrophytes are currently not a problem in Lake Henshaw, this alternative may become necessary if they were to establish at nuisance levels in the future. No appropriate submersed aquatic native plants have been found to establish with the lake to out compete aquatic weed species present and not create similar or other operational problems. As such, aquatic vegetation in the lake must be controlled to maintain the aquatic weed density tolerances established by Vista Irrigation District and/or its agent.

- **Mechanical or Physical Methods —** While aquatic macrophytes are currently not a problem in Lake Henshaw, this alternative may become necessary if they were to establish at nuisance levels in the future. Mechanical removal in the lake would require various methods including hand cutting from shore or while wading, hand-pulling aquatic weeds, use of motor-driven aquatic weed harvesters to cut and harvest vegetation, aquatic weed-whacking, or mowing.

  Generally, these techniques are very labor intensive per unit acre of water treated. Mechanical removal places personnel at risk of general water, boating, slip, trip and fall hazards, drowning, risks of spilling of motor oil and fuel, and can increase air pollution. The cost per area of mechanical removal is significantly higher than the cost of labor, product and equipment of the application of aquatic herbicides.

  In some instances, the use of mechanical techniques may be necessary when the use of algaecides or aquatic herbicides is not practical, or vegetation is not at an appropriate growth state. In general, mechanical removal and disposal of cut vegetation is significantly higher than chemical control of the same area desired for control.

- **Cultural Methods —** Cultural methods used to reduce the amount of aquatic herbicides used include modifying the timing of algaecide and aquatic herbicide and non-herbicide controls. The District and/or its agent may make algaecide and aquatic herbicide applications before the density of algae or aquatic vegetation is high enough to require higher algaecide or aquatic herbicide application rates or additional applications to maintain algae or aquatic weed populations below threshold levels.

Further, evaluating alternative control techniques is part of the District's overarching IPM approach, and additional alternatives to chemical treatment may be selected as part of the IPM program. Alternative control techniques may include mechanical removal, native species establishment, introduction of microbes/enzymes, zooplankton, and macrophytes, or enhancement of aeration/circulation methods to help improve water quality.

11.0 AQUATIC PESTICIDE MONITORING AND REPORTING PROGRAM

The Aquatic Pesticide Monitoring and Reporting Program (APMRP) described below has been developed in compliance with the requirements of Attachment C of Order 2013-0002-DWQ (as amended by Orders 2014-0078-DWQ, 2015-0029-DWQ, and 2016-0073-EXEC) for NPDES Permit # CAG990005.

The APMRP addresses the following two key questions:

> *Does the residual algaecides and aquatic herbicides discharge cause an exceedance of the receiving water limitations?*
>
> *Does the discharge of residual algaecides and aquatic herbicides, including active ingredients, inert ingredients, and degradation byproducts, in any combination cause or contribute to an exceedance of the "no toxics in toxic amount" narrative toxicity objective?*

Under the APMRP, visual, physical, and chemical monitoring will be performed in association with all algaecide and/or aquatic herbicide treatment events undertaken by the District at the Warner Ranch and Lake Henshaw. Results of the monitoring will be recorded by qualified personnel. Additional details regarding District APMRP procedures and policies are presented below. An example Aquatic Pesticide Treatment Log is provided in Appendix C.

11.1 Monitoring Types

A *treatment event* is defined as a discrete event involving the application of algaecide and/or aquatic herbicide to control nuisance growths of filamentous algae and/or weeds within the treatment area for Warner Ranch and/or filamentous, benthic, and/or planktonic algae and/or aquatic macrophytes within the treatment area for Lake Henshaw (see also Section 2). During treatment event monitoring, water samples will be collected at all applicable *treatment locations* (i.e., the specific sites at which aquatic pesticides will be directly applied) and analyzed for the parameters and constituents described in Section 11.3. If algaecides or aquatic herbicides are applied at only one treatment location per treatment event, then at least one set of treatment location samples must be collected. If multiple locations are treated per treatment event, then separate sets of treatment location samples must be collected for the event.

At a **minimum**, three types of monitoring are required for each treatment event:

Background Monitoring — Background monitoring (BG) samples must be collected upstream of the treatment location at the time of the treatment event, or they may be collected in the treatment area just prior to the treatment event (i.e., up to 24 hours in advance of treatment).

Event Monitoring — Event monitoring (EM) samples must be collected immediately downstream of the treatment location in flowing waters, or proximally adjacent to the treatment location in non-flowing (static) waters, in the portion of the treatment area that is exposed to the algaecide plume immediately or shortly after the treatment event.

The location and timing of the EM sample may be based on a number of factors including, but not limited to, algae and aquatic weed density and type, flow rates, size of the treatment area and duration of treatment.

The lake EM sample for non-flowing (static) waters must be collected immediately outside the treatment area immediately after the treatment event, but after sufficient time has elapsed such that treated water would have exited the treatment area.

Post-Event Monitoring — Post-event monitoring (PE) samples must be collected within the treatment area, or depending on where treatment locations occur, immediately downstream of the treatment area in flowing waters or adjacent to the treatment area in non-flowing waters, within one week after the treatment event.

One full set of three samples (i.e., BG, EM and PE samples) will be collected during each treatment event according to the monitoring frequency and locations described in Section 11.2.

Additionally, one Field Duplicate (FD) and one Field Blank (FB) will be collected and submitted for analysis for each analyte, once per year. The FD and FB samples will most likely be collected during Event Monitoring.

For products that contain sodium carbonate peroxyhydrate, no treatment event monitoring for residuals is required before or after treatment because peroxyhydrate breakdown products are water and bicarbonate. For application of all other algaecides and aquatic herbicides listed on the

APAP, monitoring will be conducted for each active ingredient utilized at the time of treatment event.

11.2 Monitoring Locations and Frequency

In general, monitoring locations for treatment event monitoring will include one background monitoring (BG) location, one event monitoring (EM) location, and one post-event monitoring (PE) location for each treatment event. Monitoring frequency will be based on the number of treatment locations and will address the two questions stated in Section 11.0. (Refer to Section 11.1 for a description of BG, EM and PE monitoring types.)

11.2.1 Warner Ranch

For the Warner Ranch treatment area, placement of the BG monitoring locations will be immediately upstream of the treatment location (Figure 7). If the treatment location is Location 1, 4, or 5 in Figure 7, then the BG monitoring location will be at the groundwater pump before water enters the ditch. The EM location will be directly downstream of the treatment location in the flowing waters outside of the treatment location itself. The PE monitoring location will be at the terminus of the ditch system before the water is released to the San Luis Rey River leading into Lake Henshaw (Figure 7). The EM sample will be taken within one week after the treatment event. Monitoring frequency will be based on the number of treatments, treatment areas, and duration of the application, but at a minimum will be one per treatment event.

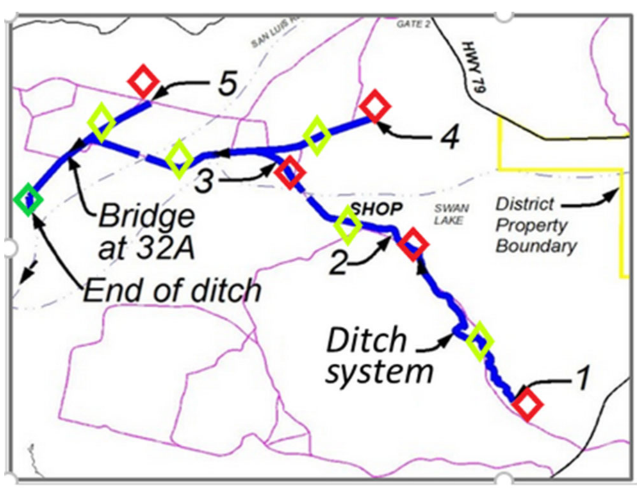


Figure 7: Warner Ranch Treatment Event Monitoring Locations. Red Diamonds represent background (BG) monitoring locations directly upstream of treatment location/s (numbered); Yellow Diamonds represent event monitoring (EM) location/s; Green Diamond represents post-event monitoring (PE) location at the terminus of the ditch system.

11.2.2 Lake Henshaw

For Lake Henshaw, the BG monitoring location will be at the treatment location itself prior to any treatment (i.e., pre-treatment event) (Figure 8). The EM monitoring will be immediately outside of the treatment location immediately after the application of algaecides and aquatic herbicides and downwind of the prevailing wind direction. The PE location will be within the treatment area; however, in some instances, the PE location will be at the Henshaw Dam outlet channel (Figure 4) when the treatment location is within the vicinity of the intake tower. PE receiving water sampling will be collected within one week after treatment event. In the event of a subsurface application in the lake, the EM and PE locations may be dependent upon hydraulic currents as well as prevailing wind direction. Monitoring frequency will be based on the number of treatments and treatment areas, but at a minimum will be one per treatment event.

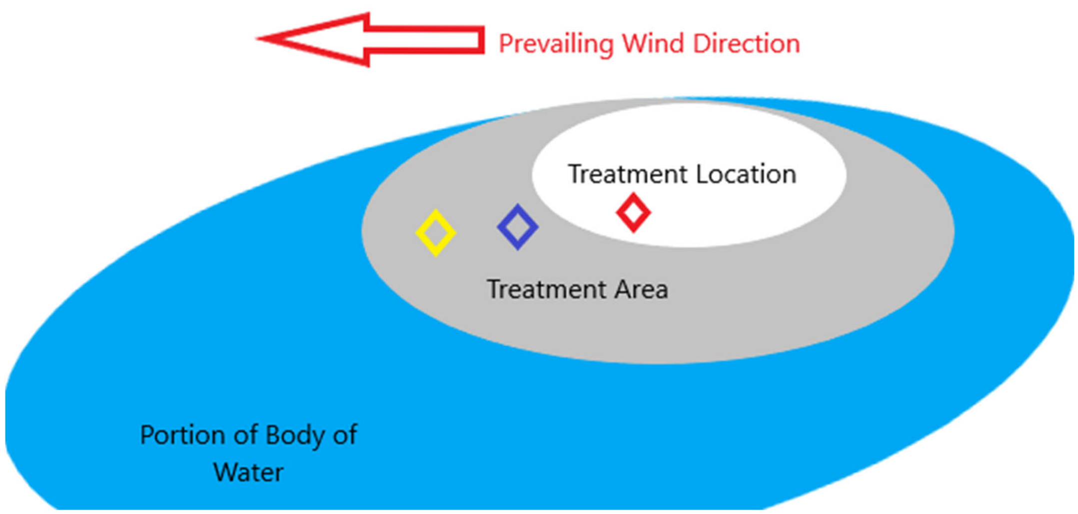


Figure 8: Lake Henshaw Treatment Event Monitoring Locations. Red Diamond represents background (BG) sampling location; Blue Diamond represents event monitoring (EM) sampling location; Yellow Diamond represents post-event (PE) monitoring location. Note that the sampling locations are dependent on prevailing wind direction and/or current direction for subsurface applications.

11.3 Monitoring Procedures

A recording logbook must be maintained by members of the monitoring team to provide a record of monitoring location, including a map showing the location of each treatment area and treatment location); and significant events, observations, and measurements taken during monitoring. Monitoring records are intended to provide sufficient data and observations to enable project team members to reconstruct events that occurred during the monitoring and must be legible, factual, detailed, and objective. As appropriate, and at the discretion of Vista Irrigation District and/or it's agent staff, observations and measurements can be supplemented with pictures of site conditions at the time of sampling.

When recording observations in the field book, the monitoring team will note the presence or absence of the following:

- Floating or suspended debris;
- Discoloration;
- Bottom deposits;
- Aquatic life;
- Visible films, sheens, or coatings;
- Fungi, slimes, or objectionable growths; and
- Potential nuisance conditions.

The District will collect field measurements and grab samples for laboratory analysis for the set of monitoring constituents and parameters listed in Table 2. Monitoring procedures for field measurements and grab samples are discussed in Section 11.3.1 and 11.3.2 below.

Table 2. Monitoring Constituents and Parameters.

Sample Type	Constituent/ Parameter	Sample Method	Laboratory Method	Frequency
Visual	Site description (lake, open waterway, channel, estimate of percent covered by vegetation, etc.) Appearance of waterway (sheen, color, clarity, etc.) Weather conditions (fog, rain, wind, etc.)	Visual Observation	Not Applicable	All applications at all sites treated
Physical	Temperature[1] Turbidity[2] Conductivity[2]	Grab[3]	See EPA Guidelines	All applications at all sites treated
Chemical	Active Ingredient[4] pH[2] Dissolved Oxygen[2] Hardness (CaCO3)	Grab[3]	See EPA Guidelines	All applications at all sites treated

[1] Field measurements with electronic instrumentation, probes, or mercury thermometers.
[2] Field measurement or sample collection for analytical laboratory testing.
[3] Samples shall be collected at 3 feet below the surface, or mid-depth if water body is less than 6 feet deep.
[4] For products that contain sodium carbonate peroxyhydrate, no residual sampling is required before or after treatment because the peroxyhydrate breakdown products are water and bicarbonate.

11.3.1 Field Sampling

In conjunction with treatment event monitoring, water temperature will be measured in the field. Turbidity, electrical conductivity, pH, and dissolved oxygen may be measured in the field using field meters as available, or these constituents may be collected as grab samples and analyzed in the laboratory. Turbidity, pH, and dissolved oxygen meters are calibrated according to manufacturer's specifications at the recommended frequency and will be checked with a standard prior to each use. Conductivity meters are calibrated by the manufacturer and will be checked according to manufacturer's specifications with standards throughout the year to evaluate instrument performance. If the calibration is outside the manufacturer's specifications, the probe will be recalibrated. Calibration logs will be maintained for all instruments to document calibration.

- **Water Temperature** — A standard direct-reading, instrument-grade thermometer shall be partially submerged in the treatment location flow until it reaches equilibrium before being read.

- **pH** — A Hach "pH Pocket Pal Tester" or equivalent will be used for field measurement of pH. It shall be calibrated with at least two prepared buffer solutions (e.g., pH 4 and pH 8) before each day of testing (if applicable). Calibrations and field testing shall be performed in adherence with the manufacturer's instructions.
- **Turbidity** — A Hach 2100P Portable Turbidimeter or equivalent shall be utilized for field measurement of turbidity. The instrument shall be calibrated before each analysis (if applicable), and the calibrations and tests shall be performed in adherence with the manufacturer's instructions.
- **Dissolved Oxygen** — A Hach Portable Oxygen Meter or equivalent shall be utilized for determining dissolved oxygen concentrations in the field. The meter shall be calibrated before each analysis (if applicable), and the calibrations and tests shall be performed in adherence with the manufacturer's instructions. The probe's membrane shall be inspected before each sampling event and replaced per manufacturer's recommendation, with membrane replacement occurring at the onset of each new testing season (i.e., spring).
- **Electrical Conductivity** — A Hach Conductivity/TDS Meter (Model 44600) or equivalent shall be utilized for field measurement of electrical conductivity in water. The instrument shall be calibrated before each analysis (if applicable), and the calibrations and tests shall be performed in adherence with the manufacturer's instructions.

11.3.2 Grab Sampling and Laboratory Analyses

If the water depth is 6 feet or greater, the grab sample will be collected at a depth of 3 feet. If the water depth is less than 6 feet the sample will be collected at the approximate mid-depth. As necessary, an intermediary sampling device will be used for locations that are difficult to access. Sampling containers will be inverted before being lowered into the water to the desired sample depth, where it will be turned upright to collect the sample (if applicable). An alternative testing method to the bottle inversion method it to utilize a Van Dorn bottle or similar device.

Grab samples will be collected using sampling procedures that minimize the loss of the constituents sampled for and that maintain sample integrity. Clean, empty sample containers with caps will be supplied in protective cartons or ice chests by the primary laboratory. The containers will be certified clean by either the laboratory or the container supplier. To ensure data quality control, the sampler will utilize the appropriate sample container as specified by the laboratory for each sample type. Sample container type, holding time, and appropriate preservatives are listed in

Table 3. Each container will be affixed with a label indicating a discrete sample number for each sample location. The label will also indicate the date and time of sampling and the sampler's name.

Samples may either be collected with bottles containing the correct preservatives(s), or collected in unpreserved bottles and preserved upon receipt at the analytical lab. After collection, samples will be refrigerated at approximately four degrees Celsius, stored in a dark place, and transported to the laboratory.

All samples will be packed and transported the day the samples are collected to provide ample time for samples to be analyzed within the required holding time.

Ice will be included in coolers containing samples that require temperature control and transported to the laboratory for analysis in the following manner:

- Sample container stickers will be checked for secure attachment to each sample container.
- The sample containers will be placed in the lined cooler.
- The chain of custody (COC) will be placed inside a plastic bag and placed inside the cooler. The COC will indicate each unique sample identification name, time and place of sample collection, the sample collector, the required analysis, turn-around-time, and location to which data will be reported. An example COC is provided in Appendix C.
- The cooler will then be readied for pick-up by a courier or delivered directly to the laboratory.

Table 3 presents the set of analytes associated with each treatment event monitoring sample that is collected. All laboratory analyses will be conducted by a California Department of Health Services certified laboratory in accordance with the latest edition of *Guidelines Establishing Test Procedures for Analysis of Pollutants* (40 CFR part 136).

Table 3: Laboratory Analytical Methods for Grab Samples.

Analyte	EPA Method	Method Reporting Limit	Hold Time (Days)	Sample Container	Chemical Preservative
Water Temperature[1]	N/A	N/A	N/A	N/A	N/A
Dissolved Oxygen[1]	360.1 or 360.2	0.0 mg/L	1	1L Amber Glass	None
Turbidity[2]	180.1	0.00 NTU	2	100 mL HDPE	None
Conductivity[2]	120.1	0 µS/cm	28	100 mL HDPE	None
pH[2]	150.1 or 150.2	1-14 s.u.	Immediately	100 mL HDPE	None
Nonylphenol[3]	550.1	0.5µg/L	7	2x40 mL VOA	None
Hardness [4]	SM2340B	0.7 $CaCO_3$/L	1 day unpreserved; 180 days if preserved	250 mL HDPE	HNO_3
Copper	200.8	0.2 µg/L	180 days	250 mL polypropylene	HNO_3
*Diquat	549	40 µg/L	7	500 mL Amber HDPE	H_2SO_4
*Endothall	548.1	40 µg/L	7	100 mL Amber Glass or 2x40 mL VOA	None
*Glyphosate	547	0.5 µg/L	14	2x40 mL VOA	None
*Imazamox	**FasTEST-05.03	1.0mg/L	14	30 ml HDPE	None

Notes:

*Signifies algaecide or aquatic herbicide active ingredient. Chemical analysis is only required for the active ingredients(s) used in treatment.

**Per Sepro (manufacture) no listed EPA method was noted for Imazamox. The fasTEST method from SePro is currently available.

Analysis not required for algaecides and aquatic herbicides containing sodium carbonate peroxyhydrate.

EPA Methods are taken from NEMI 2004

1 – Field Measured

2 – May be field or laboratory measured

3 – Required only when a nonylphenol-based surfactant is used

4 – Required for copper applications only

HPLC – High Performance Liquid Chromatography

m – Modified extraction or analysis technique

11.4 Monitoring Records

Records of treatment event monitoring information will include the following:

- Date, exact place, and time of sampling or measurements.
- Individuals who performed the sampling or measurements.
- Dates analyses were performed.
- Individuals who performed the analyses.
- Analytical techniques or method used.
- Results of such analyses.

11.5 Retention of Records

The District will retain records of the following:

- Monitoring information including all calibration and maintenance records.
- Copies of all reports required by the general permit.
- Records of all data used to complete the application for this general permit.

Records must be maintained for a minimum of 3 years from the date of the sampling, measurement, or report. This period may be extended during the course of any unresolved litigation regarding this discharge or when requested by the Regional Board Executive Officer.

11.6 Device Calibration and Maintenance

All monitoring instruments and devices that are used by the District for the APMRP must be properly maintained and calibrated as necessary to ensure their continued accuracy.

11.7 Treatment Area Inspections

District staff will routinely inspect the integrity of the Warner Ranch water supply system, the Henshaw Dam, and the associated treatment areas for the ranch and Lake Henshaw, prior to every algaecide/herbicide application to ascertain that treated water is not unintentionally discharged to streams, rivers, lakes, or other natural waterways.

11.7 Maximum Copper Concentration

Note that relative to the District's use of copper sulfate as an algaecide, Section C of Order 2013-0002-DWQ specifies that "discharges shall not cause or contribute to an exceedance of the following in the receiving water."

Maximum Copper Concentration = exp[0.8545(ln(harness))-1.702]

For monitoring associated with each treatment event, the District will evaluate compliance with this copper limitation and provide the compliance assessment in the Annual Report to the Regional Board.

11.8 Reporting

All APMRP reports submitted to the San Diego Regional Board must comply with the provisions stated in "Standard Provisions and Reporting for Waste Discharge Requirements (NPDES)" (see Appendix A, Attachment D).

On an annual basis, the District will compile all monitoring forms, summarize monitoring data, and present an evaluation of the monitoring results for the calendar year in an APMRP Annual Report. APMRP Annual Reports will contain the following:

- An Executive Summary discussing compliance with Order 2013-0002-DWQ as amended by Orders 2014-0078-DWQ, 2015-0029-DWQ, and 2016-0073-EXEC) and the effectiveness of the APAP to reduce or prevent the discharge of pollutants associated with aquatic pesticide applications.

- A summary of all monitoring data, including the identification of water quality improvements or degradation, and recommendations for improvements to this APAP (including proposed BMPs) based on the monitoring results. Monitoring results will be tabulated in summary form based on analytical laboratory reports and post-event (PE) monitoring results including receiving water monitoring sample results will be compared to applicable water quality standards. Monitoring results will indicate:
    - The name of the monitoring agency or organization
    - Detailed monitoring location information (including latitude and longitude or township/range/section if available)
    - A map showing each treatment area and each treatment location
    - The amount of algaecide/herbicide used during each treatment event
    - Information on surface area and/or volume of treatment area and any other information used to calculate dosage and quantity of each pesticide used
    - Sample collection date(s)
    - Name of constituent/parameter(s) and the concentration detected
    - Minimum allowable levels for each constituent/parameter
    - Method detection limits for each constituent/parameter
    - Name or description of water body and a comparison with applicable water quality standards
    - Description of analytical quality assurance/quality control
    - Identification of BMPs and a discussion of their effectiveness in meeting APAP requirements
    - A discussion of BMP modifications addressing violations of this General Permit
    - Recommendations to improve the APMRP, BMPs, and APAP to ascertain compliance with Order 2013-0002-DWQ
    - Any proposed changes to the APAP and APMRP

11.8.2 Certification

Each Annual Report submitted to the Regional Board must be signed by a responsible officer or duly authorized representative of the District. The following certification statement must be provided with each submittal:

> "I certify under penalty of law that this document and all attachments were prepared under my direction or supervision in accordance with a system designed to assure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system or those persons directly responsible for gathering the information, the information submitted is, to the best of my knowledge and belief, true, accurate, and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations."

Signed ______Brett Hodgkiss______

Brett Hodgkiss

Title ______General Manager______

Date ______04/06/2021______

11.8.3 Reporting Schedule

Annual monitoring reports (covering the calendar year, January 1 – December 31) must be submitted to the San Diego Regional Board Executive Officer by March 1 of each year. Annual Reports should be submitted to:

Executive Officer

San Diego Regional Water Quality Control Board
9174 Sky Park Court, Suite 100
San Diego, CA 92123-4340

(858) 467-2952
(858) 571-6972 (fax)

Appendix A

Executed Notice of Intent (NOI)

Attachment E – Notice of Intent

WATER QUALITY ORDER NO. 2013-0002-DWQ
GENERAL PERMIT NO. CAG990005

STATEWIDE GENERAL NATIONAL POLLUTANT DISCHARGE ELIMINATION SYSTEM (NPDES) PERMIT FOR RESIDUAL AQUATIC PESTICIDE DISCHARGES TO WATERS OF THE UNITED STATES FROM ALGAE AND AQUATIC WEED CONTROL APPLICATIONS

I. NOTICE OF INTENT STATUS (see Instructions)

Mark only one item	A. New Applicator	B. Change of Information: WDID# ______________________	
	C. ☐ Change of ownership or responsibility: WDID# ____________________________		

II. DISCHARGER INFORMATION

A. Name Vista Irrigation District			
B. Mailing Address 1391 Engineer Street			
C. City Vista	D. County San Diego	E. State California	F. Zip 92081
G. Contact Person Mark Saltz	H. E-mail address msaltz@vidwater.org	I. Title Water Resources Specialist	J. Phone 760-597-3112

III. BILLING ADDRESS (Enter Information *only* if different from Section II above)

A. Name			
B. Mailing Address			
C. City	D. County	E. State	F. Zip
G. E-mail address	H. Title	I. Phone	

IV. RECEIVING WATER INFORMATION

A. Algaecide and aquatic herbicides are used to treat (check all that apply):

1. X Canals, ditches, or other constructed conveyance facilities owned and controlled by Discharger.
   Name of the conveyance system: Warner Ranch Ditch System
2. ☐ Canals, ditches, or other constructed conveyance facilities owned and controlled by an entity other than the Discharger.
   Owner's name: ______
   Name of the conveyance system: ______
3. X Directly to river, lake, creek, stream, bay, ocean, etc.
   Name of water body: Lake Henshaw

B. Regional Water Quality Control Board(s) where treatment areas are located (REGION 1, 2, 3, 4, 5, 6, 7, 8, or 9): Region Region 9
(List all regions where algaecide and aquatic herbicide application is proposed.)

V. ALGAECIDE AND AQUATIC HERBICIDE APPLICATION INFORMATION

A. Target Organisms: Ditch System: aquatic macrophytes and filamentous algae
Lake Henshaw: cyanobacteria, including but not limited to: *Microcystis sp., Planktothrix sp., Lyngbya/Aphanizomenon sp., Woronichinia sp., and Dolichospermum sp.*

B. Algaecide and Aquatic Herbicide Used: List Name and Active ingredients

Copper sulfate and chelated copper; diquat dibromide; endothall; glyphosate; sodium carbonate peroxyhydrate; ammonium salt of imazamox

More detail provided in Appendix C of Aquatic Pesticide Application Plan

C. Period of Application: Start Date (to be determined) End Date (to be determined)

D. Types of Adjuvants Used: Aquatic-labeled adjuvants

VI. AQUATIC PESTICIDE APPLICATION PLAN

Has an Aquatic Pesticide Application Plan been prepared and is the applicator familiar with its contents?

X Yes ☐ No

If not, when will it be prepared? ______

VII. NOTIFICATION

Have potentially affected public and governmental agencies been notified? X Yes ☐ No

VIII. FEE

Have you included payment of the filing fee (for first-time enrollees only) with this submittal?

X YES ☐ NO ☐ NA (Under separate cover)

IX. CERTIFICATION

"I certify under penalty of law that this document and all attachments were prepared under my direction and supervision in accordance with a system designed to ensure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system, or those persons directly responsible for gathering the information, the information submitted is, to the best of my knowledge and belief, true, accurate, and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine or imprisonment. Additionally, I certify that the provisions of the General Permit, including developing and implementing a monitoring program, will be complied with."

A. Printed Name: Brett Hodgkiss

B. Signature: Brett Hodgkiss **Date:** 04/06/2021

C. Title: General Manager

XI. FOR STATE WATER BOARD STAFF USE ONLY

WDID:	**Date NOI Received:**	**Date NOI Processed:**
Case Handler's Initial:	**Fee Amount Received:** **$**	**Check #:**
☐ **Lyris List Notification of Posting of APAP**	**Date** ______________________	**Confirmation Sent** _______

INSTRUCTIONS FOR COMPLETING NOI

WATER QUALITY ORDER NO. 2013-0002-DWQ
GENERAL PERMIT NO. CAG990005

STATEWIDE GENERAL NATIONAL POLLUTANT DISCHARGE ELIMINATION SYSTEM (NPDES) PERMIT FOR RESIDUAL AQUATIC PESTICIDE DISCHARGES TO WATERS OF THE UNITED STATES FROM ALGAE AND AQUATIC WEED CONTROL APPLICATIONS

These instructions are intended to help you, the Discharger, to complete the Notice of Intent (NOI) form for the Statewide General NPDES permit. **Please type or print clearly when completing the NOI form**. For any field, if more space is needed, submit a supplemental letter with the NOI.

Send the completed and signed form along with the filing fee and supporting documentation to the Division of Water Quality, State Water Resources Control Board. Please also send a copy of the form and supporting documentation to the appropriate Regional Water Quality Control Board (Regional Water Board).

<u>Section I – Notice of Intent Status</u>

Indicate whether this request is for the first time coverage under this General Permit or a change of information for the discharge already covered under this General Permit. Dischargers that are covered under Order No. 2004-0009-DWQ before effective date of this General Permit should check the box for change of information. For a change of information or ownership, please supply the eleven-digit Waste Discharge Identification (WDID) number for the discharge.

<u>Section II – Discharger Information</u>

Enter the name of the Discharger.
Enter the street number and street name where correspondence should be sent (P.O. Box is acceptable).
Enter the city that applies to the mailing address given.
Enter the county that applies to the mailing address given.
Enter the state that applies to the mailing address given.
Enter the zip code that applies to the mailing address given.
Enter the name (first and last) of the contact person.
Enter the e-mail address of the contact person.
Enter the contact person’s title.
Enter the daytime telephone number of the contact person

<u>Section III – Billing Address</u>

Enter the information **<u>only</u>** if it is different from Section II above.

A. Enter the name (first and last) of the person who will be responsible for the billing.

B. Enter the street number and street name where the billing should be sent (P.O. Box is acceptable).
C. Enter the city that applies to the billing address.
D. Enter the county that applies to the billing address.
E. Enter the state that applies to the billing address.
F. Enter the zip code that applies to the billing address.
G. Enter the e-mail address of the person responsible for billing.
H. Enter the title of the person responsible for billing.
I. Enter the daytime telephone number of the person responsible for billing.

<u>Section IV – Receiving Water Information</u>

Please be reminded that this General Permit does not authorize any act that results in the taking of a threatened or endangered species or any act that is now prohibited, or becomes prohibited in the future, under either the California Endangered Species Act (Fish and Game Code §2050 et. seq) or the Federal Endangered Species Act (16 U.S.C.A. §1531 et. seq). This General Permit requires compliance with effluent limitations, receiving water limitations, and other requirements to protect the beneficial uses of waters of the state. The Discharger is responsible for meeting all requirements of the applicable Endangered Species Act.

Additional information on federally-listed threatened or endangered species and federally-designated critical habitat is available from NMFS (www.nmfs.noaa.gov) for anadromous or marine species or FWS (www.fws.gov) for terrestrial or freshwater species.

A. Check all boxes that apply. At least one box must be checked.
1. Check this box if the treatment area is a canal, ditch, or other constructed conveyance system owned and controlled by Discharger. Print the name of the conveyance system.
2. Check this box if the treatment area is a canal, ditch, or other constructed conveyance system owned and controlled by an entity other than the Discharger. Print the owner’s name and names of the conveyance system.
3. Check this box if the treatment area is not a constructed conveyance system (including application to river, lake, creek, stream, bay, or ocean) and enter the name(s) of the water body(s).

B. List all Regional Water Board numbers where algaecide and aquatic herbicide application is proposed. Regional Water Board boundaries are defined in section 13200 of the California Water Code. The boundaries can also be found on our website at http://www.waterboards.ca.gov/waterboards_map.shtml

Regional Water Board Numbers	Regional Water Board Names
1	North Coast
2	San Francisco Bay
3	Central Coast

Regional Water Board Numbers	Regional Water Board Names
4	Los Angeles
5	Central Valley (Includes Sacramento, Fresno, Redding Offices)
6	Lahontan (South Lake Tahoe, Victorville offices)
7	Colorado River Basin
8	Santa Ana
9	San Diego

Section V – Algaecide and Aquatic Herbicide Application Information

A. List the appropriate target organism(s).
B. List the name and active ingredients of each algaecide and aquatic herbicide to be used.
C. List the start and end date of proposed aquatic algaecide and aquatic herbicide application event.
D. List the name(s) and type(s) of adjuvants that will be used.

The Discharger must submit a new NOI if any information stated in this section will be changed. If the Discharger plans to use an algaecide and aquatic herbicide product not currently covered under its Notice of Applicability (NOA), and the algaecide and aquatic herbicide product may be discharged to a water of the United States as a result of algaecide and aquatic herbicide application, the Discharger must receive a revised NOA from the State Water Board's Deputy Director of the Division of Water Quality before using that product.

Section VI – Aquatic Pesticide Application Plan

The Coalition or Discharger must prepare and complete an Aquatic Pesticide Application Plan (APAP). The minimum contents of APAP are specified in the permit under Section VIII.C, Limitations and Discharge Requirements, of the General Permit. The Discharger must ensure that its applicator is familiar with the APAP contents before algaecide and aquatic herbicide application.

If an APAP is not complete at the time of application, enter the date by which it will be completed.

Section VII – Notification

Indicate if you have notified potentially affected public and governmental agencies, as required under item VIII.B of the General Permit.

Section VIII – Fee

The amount of Annual fee shall be based on Category 3 discharge specified in section 2200(b)(9) of title 23, California Code of Regulations. Fee information can be found at http://www.waterboards.ca.gov/resources/fees/docs/fy1112fee_schdl_npdes_prmt.pdf.

Check the YES box if you have included payment of the annual fee. Check the NO box if you have not included this payment. **NOTE:** You will be billed annually and payment is required to continue coverage.

<u>Section IX– Certification</u>

A. Print the name of the appropriate official. The person who signs the NOI must meet the signatory and certification requirements stated in Attachment B Standard Provisions item V.B.
B. The person whose name is printed above must sign and date the NOI.
C. Enter the title of the person signing the NOI.

Appendix B

Approved Algaecide Products Additional Information

Approved algaecide/herbicide product labels are available at the below weblinks.

Algimycin PWF
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/ab_algimycin_info_sheet.pdf

Aquathol K
http://www.cdms.net/ldat/ld195001.pdf

Aquathol Super K
http://www.cdms.net/ldat/ld2AE011.pdf

Captain
https://www.sepro.com/Documents/Captain_Label.pdf

Captain XTR
https://www.sepro.com/Documents/Captain-XTR_Label.pdf

Chem 1 Copper Sulfate
https://brandt.co/media/1638/copper-sulfate-crystals-label.pdf

Clearcast
https://www.sepro.com/Documents/Clearcast_Label.pdf

Cleargate EC9
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/ab_clearigate_ec9_info_sheet.pdf

Cleargate
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/ab_clearigate_info_sheet.pdf

Cutrine Plus Granular
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/ab_cutrine_plus_gran_info_sheet.pdf

Cutrine Plus
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/ab_cutrine_plus_info_sheet.pdf

Cutrine Ultra
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/cutrine_ultra_specimen_label.pdf

Harpoon
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/harpoon_specimen_label.pdf

Harpoon Granular
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/harpoon_granular_specimen_label.pdf

Hydrothol 191
http://www.cdms.net/ldat/ld225004.pdf

Hydrothol Granular
http://www.cdms.net/ldat/ld8UG006.pdf

Komeen
https://www.sepro.com/Documents/Komeen_Label.pdf

Komeen Crystal
https://www.sepro.com/Documents/Komeen-Crystal_Label.pdf

K-tea
https://www.sepro.com/Documents/K-Tea_Label.pdf

Littora
https://www.sepro.com/Documents/Littora_Label.pdf

Pax 27
https://www.sepro.com/Documents/Pak-27_Label.pdf

Phycomycin SCP
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/ab_phycomycin_info_sheet.pdf

Pond-Klear
https://www.appliedbiochemists.com/uploads/7/6/9/4/76946485/ab_pond-klear_info_sheet.pdf

Round Up Custom
http://horizon.wiki/images/a/a6/Monsanto_Roundup_Custom_Herbicide_Label.pdf

Teton
http://www.cdms.net/ldat/ld9JU000.pdf

Appendix C

Example Chain-of-Custody Record Example Aquatic Pesticide Treatment and Monitoring Event Log

Enthalpy Analytical - San Diego
4340 Vandever Avenue, San Diego, CA 92120
Phone 858-587-7333
info@enthalpy.com

ENTHALPY ANALYTICAL

Chain of Custody

Date______________ Page___ of ___

Sample Collection By:

Report to:		Invoice To: Same as Report to ☐	
Company		Company	
Address		Address	
City/State/Zip		City/State/Zip	
Contact		Contact	
Phone		Phone	
Email		Email	

	SAMPLE ID	SAMPLE			MATRIX CODE	Container		COMMENTS	ANALYSES REQUIRED						Enthalpy Matrix Codes:	Receipt Temperature (°C)
		Date	Time	Type (G or C)	(FW, SW, Sed, STRM, GW, WW, O)	Type	Qty								G = Grab C = Composite FW = Freshwater SW = Seawater Sed = Sediment STRM = Stormwater GW = Groundwater WW = Wastewater O = Other (specify)	
1																
2																
3																
4																
5																
6																
7																
8																
9																
10																

PROJECT INFORMATION		SAMPLE RECEIPT	
Project Name:		Total No. of Containers	
PO No.:		Received Good Condition?	
Shipped Via:		Matches Test Schedule?	

SPECIAL INSTRUCTIONS/COMMENTS:

1) RELINQUISHED BY (CLIENT)		2) RECEIVED BY (COURIER)	
(Signature)	(Time)	(Signature)	(Time)
(Printed Name)	(Date)	(Printed Name)	(Date)
(Company)		(Company)	

3) RELINQUISHED BY (COURIER)		4) RECEIVED BY (LABORATORY)	
(Signature)	(Time)	(Signature)	(Time)
(Printed Name)	(Date)	(Printed Name)	(Date)
(Company)		(Company)	(Log-in #s)

Additional costs may be required for sample disposal or storage. Payment net 30 unless otherwise contracted.
Shaded areas are for lab use only
Report turn-around-time varies depending on length of test; please inquire with your project manager.

http://enthalpy.com/environmental-toxicology-2/

Herbicide/Algaecide Application and Monitoring Event Log

Warner Ranch Treatment Event

BG Sample Location:

Date:_____________ Time:_______________

Latitude:______________________________

Longitude:_____________________________

EM Sample Location:

Date:_____________ Time_______________

Latitude:______________________________

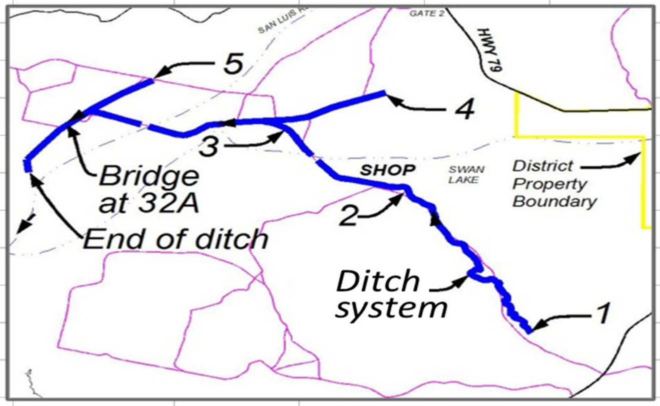
Longitude:_____________________________

PE Sample Location:

Date:_____________ Time:_______________

Latitude:______________________________

Longitude:_____________________________



Indicate sample Locations on map (BG, EM, PE)

Instructions

Background Sample (BG) for active ingredient, will be taked within 24 hours prior to application directly upstream of the treatment location; Event Monitoring (EM) of active ingredient will be taken immediately downstream of the treatment location after application in the flowing waters adjacent to the treeatment area; Post-Event Monitoring (PE) for active ingredient will be taken at the end of ditch within 1 week after the treatment event.

Additional Parameters:

(BG)	(EM)	(PE)
Temp:______________	Temp:______________	Temp:______________
pH:________________	pH:________________	pH:________________
Turbidity:___________	Turbidity:___________	Turbidity:___________
DO:________________	DO:________________	DO:________________
Conductivity:_________	Conductivity:_________	Conductivity:_________
Hardness:___________	Hardness:___________	Hardness:___________

Field Book Observations:

____

____

____

Algaeicide/Herbicide Applied:

1)______________________ Amount:______________Rate:__________________

2)______________________ Amount:______________Rate:__________________

3)______________________ Amount:______________Rate:__________________

Sample/s Taken By:___________________________________Date:_____________

Sample/s Taken By:___________________________________Date:_____________

Application Completed By:______________________________Date:_____________

Herbicide/Algaecide Application and Monitoring Event Log

Lake Henshaw Treatment Event Page 1

BG Sample Location:

Date:_____________ Time:_______________

Latitude:________________________________

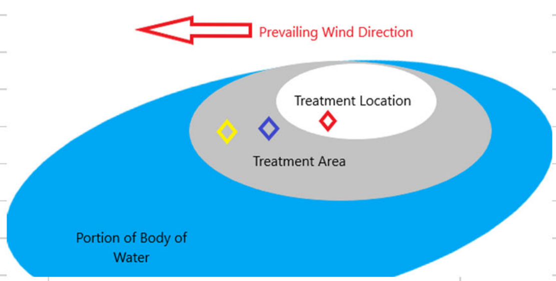
Longitude:_______________________________

EM Sample Location:

Date:_____________ Time_______________

Latitude:________________________________

Longitude:_______________________________



PE Sample Location:

Date:_____________ Time:_______________

Latitude:________________________________

Longitude:_______________________________

Red Diamond represents (BG) sample location; Blue Diamond represents (EM) location; Yellow Diamond represents (PE) location.

Instructions

Background Sample (BG) for active ingredient, will be taked within 24 hours prior to application within the treatment location; Event Monitoring (EM) of active ingredient will be taken downwind of the treatment location within the treatment area immediately after application; Post-Eventing Monitoring sample (PE) for active ingreident will be taken within 1 week in the treatment area, in some instances the PE sample location will be taken at the dam outlet channel.

Additional Parameters:

(BG)	(EM)	(PE)
Temp:_______________	Temp:_______________	Temp:_______________
pH:_________________	pH:_________________	pH:_________________
Turbidity:___________	Turbidity:___________	Turbidity:___________
DO:_________________	DO:_________________	DO:_________________
Conductivity:_________	Conductivity:_________	Conductivity:_________
Hardness:____________	Hardness:____________	Hardness:____________

Field Book Observations:

____

____

____

Algaeicide/Herbicide Applied:

1)________________________ Amount:______________Rate:___________________

2)________________________ Amount:______________Rate:___________________

3)________________________ Amount:______________Rate:___________________

Sample/s Taken By:______________________________________Date:______________

Sample/s Taken By:______________________________________Date:______________

Application Completed By:________________________________Date:______________

Herbicide/Algaecide Application and Monitoring Event Log

Lake Henshaw Treatment Event Page 2

Indicate on the provided map the Treatment Location (TL), the Treatment Area (TA)
See page 1 for example. Also indicate sample locations (BG, EM & PE) as well as relative
wind direction.

Appendix D

Reference List

WATER QUALITY ORDER NO. 2001-12-DWQ (CAG990003)
WATER QUALITY ORDER NO. 2004-0009-DWQ 9 (CAG 990005)
WATER QUALITY ORDER NO. 2013-0002-DWQ (CAG990005)
https://www.waterboards.ca.gov/board_decisions/adopted_orders/water_quality/

WATER QUALITY CONTROL PLAN FOR THE SAND DIEGO BASIN (9)
https://www.waterboards.ca.gov/sandiego/water_issues/programs/basin_plan/docs/R9_Basin_Plan.pdf

California Toxics Rule (CTR)
https://www.federalregister.gov/documents/2000/05/18/00-11106/water-quality-standards-establishment-of-numeric-criteria-for-priority-toxic-pollutants-for-the

Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries in California, a.k.a. the State Implementation Plan, or SIP (2005)
https://www.waterboards.ca.gov/water_issues/programs/state_implementation_policy/docs/sip2005.pdf

STATE WATER RESOURCES CONTROL BOARD ORDER WQ 2014-0078-DWQ AMENDING STATE WATER RESOURCES CONTROL BOARD WATER QUALITY ORDER 2013-0002-DWQGENERAL PERMIT NO. CAG 990005
https://www.waterboards.ca.gov/board_decisions/adopted_orders/water_quality/2014/wqo2014_0078_dwq.pdf

STATE WATER RESOURCES CONTROL BOARDORDERWQ 2015-0029-DWQ AMENDING STATE WATER RESOURCES CONTROL BOARD WATER QUALITY ORDER 2013-0002-DWQ (AS AMENDED BY ORDER 2014-0078-DWQ) GENERAL PERMIT NO. CAG 990005
https://www.waterboards.ca.gov/board_decisions/adopted_orders/water_quality/2015/wqo2015_0029_dwq.pdf

STATE WATER RESOURCES CONTROL BOARD ORDER 2016-0073-EXEC AMENDING WATER QUALITY ORDER 2013-0002-DWQ GENERAL PERMIT NO. CAG 990005
https://www.waterboards.ca.gov/water_issues/programs/npdes/pesticides/docs/weedcontrol/201600743exec_wcpa.pdf

National Environmental Methods Index (NEMI 2004)
https://www.nemi.gov/home/

Appendix D

Water Quality and Cyanobacteria-related Test Methods

Table D-1. Pros and cons for a variety of water quality and cyanobacteria-related test and/or measurement methods considered for use in Lake Henshaw and Lake Wohlford, California.

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Physical				
Secchi depth (Secchi disk)	USGS	0.1–10 m	Direct measure of water column turbidity Low entry cost (<$50) Low per sample cost at <$1 Instantaneous results	Indicator of phytoplankton biomass most useful when coupled with other chemical and biological measurements Non-photosynthetic suspended particles are confounding during high sediment transport events
Chlorophyll-a				
Chlorophyll-a/ phycocyanin (LG Sonic)	In vivo fluorescence sensor	Chl-a and PC: 0–500 ug/L	Direct measure of surface phytoplankton biomass along with water temperature, dissolved oxygen, pH, and turbidity, as part of treatment with ultrasonic waves to reduce cyanobacteria blooms Instantaneous, continuous results with real-time data reporting Ratio of phycocyanin to chlorophyll-a is early warning that cyanotoxins may occur by identifying when cyanobacteria are becoming more abundant relative to other types of phytoplankton – ultrasonic wave treatment is aligned with this	Experimental/unproven treatment High entry cost at $61K per MPC-Buoy Pro unit (ultrasound with probes and meters) and $56K per MPC-Buoy Lite (just ultrasound) with small (250-meter) radius (up to 50 acres), likely would require 2–3 MPC-Buoy Pro units and 20+ MPC-Buoy Lite for Henshaw, 2–3 for Wohlford; floating discs with visible buoy. Estimated costs include transportation and commissioning. Measurements only at surface (6–12 in. below the water surface)—no vertical profiles in water column Moderate level of staff training necessary for deployment, maintenance, and software interaction Periodic service needs unknown

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Chlorophyll-a/ phycocyanin (Algae Tracker)	In vivo fluorescence sensor	Chl-a: 0–2,000 RFU; 0–200 ug/L PC: 0–750 RFU; 0–1,500 ug/L	Direct measure of surface phytoplankton biomass along with solar intensity, air temperature, water temperature, and turbidity Instantaneous, continuous results with real-time data reporting Ratio of phycocyanin to chlorophyll-a is early warning that cyanotoxins may occur by identifying when cyanobacteria are becoming more abundant relative to other types of phytoplankton	High entry cost at $2,500 per unit, likely would require 4 Algae Tracker units for Henshaw, 2–3 for Wohlford; floating discs with visible buoy Does not measure dissolved oxygen or pH Measurements only at surface (6–12 in. below the water surface)—no vertical profiles in water column Chlorophyll-a/phycocyanin RFUs and turbidity NTUs not directly comparable to laboratory analyzed samples without developing lake-specific relationships Does not directly identify cyanobacteria species or the likelihood that particular species will produce cyanotoxins Moderate level of staff training necessary for deployment, maintenance, and software interaction Periodic service needs unknown

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Chlorophyll-a/ phycocyanin (PC) (YSI ProDSS or EXO)	In vivo fluore-scence sensor	Chl-a: 0.1–100 RFU; 0.1–400 ug/L PC: 0.01–100 RFU; 0.01–100 ug/L	Direct measure of phytoplankton biomass with generally understood and established relationships between chlorophyll-a and other water quality parameters affected by algal blooms, including dissolved oxygen, pH, turbidity, nutrients Instantaneous, continuous results with real-time data reporting Sensor is part of meter that typically includes one or more sensors for water temperature, conductivity, dissolved oxygen, pH, oxidation-reduction potential, and turbidity Ability to collect data throughout the water column or at a single depth	High entry cost ($5,000–$35,000) depending on what other probes the chlorophyll-a/phycocyanin probe is coupled with and length of cord for depth profiles Moderate level of staff training necessary for calibration, deployment, maintenance, and software interaction Periodic (e.g., 1x every 3-4 years) factory recalibration and service may result in instrument downtime and shipping costs to send to YSI
Chlorophyll-a	USEPA Method 445.0	0.5–2,000 ug/L	Direct measure of phytoplankton biomass with generally understood and established relationships between chlorophyll-a and other water quality parameters affected by algal blooms, including dissolved oxygen, pH, turbidity, nutrients No/low entry cost Low per sample cost at $42 per sample + $50 overnight shipping per sample = $92 per sample Limited staff training necessary No equipment necessary	3–4 d for test results Samples can be lost/broken during shipping

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Cyanotoxin concentration				
Rapid screening test strips (Eurofins)	Immuno-chromatographic	ATX : 0.4–2.5 ug/L CYN: 0.5–10.0 ug/L MC-NOD: 0–10.0 ug/L	Available for ATX, CYN, MC-NOD Test results in ~30 min No/low entry cost Low per sample cost $35–$40 (20 strips at $530 + $200 for standards) Limited staff training necessary No equipment necessary Moderate quantification limits	Not available for SAX Screening method only—not actionable/requires immediate follow up sampling with quantitative method Difficult to run more than 3 or 4 samples at a time Difficult to discern test results—optional hand-held test strip reader ($2,760 entry cost) recommended because reader better determines intensity of lines Test range likely to require in-field sample dilution during intense blooms, which would require the same sample to be analyzed more than one time Some cyanobacteria species (e.g., *Planktothrix*) are not fully lysed using chemical quick lysing step, rather than USEPA Method 546 (ELISA), which uses 3x freeze-thaw lysing ATX kit does not include chemical cell lysing reagent; must use freeze-thaw method to measure intracellular toxin concentrations Cannot distinguish between ATX and homoanatoxin, or between MC and NOD, may miss some congeners of MC

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
BlueGreenTest	Immuno-chromatographic	1 to 2 ug/L MRL; and between 1 or 2 in 20 ug/L is linear	Available for MC-NOD, including 10 congeners of MC Test results in ~15 min No/low entry cost Low per sample cost \$10–\$15 (three tests for \$30–\$45 + \$200 for standards) Limited staff training necessary Limited equipment necessary (need digital camera to use with app) Moderate quantification limits	Not available for ATX, CYN, SAX Screening method only—not quantitative and not actionable/requires immediate follow up sampling with quantitative method May be difficult to discern test results due to intensity of lines on strip Test range may require in-field sample dilution during intense blooms, which would require the same sample to be analyzed more than one time Some cyanobacteria species (e.g., *Planktothrix*) are not fully lysed using chemical quick lysing step, rather than USEPA Method 546 (ELISA), which uses 3x freeze-thaw lysing

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Rapid testing LightDeck Mini (Hach)	Fluoresence-based microarray	CYN: 0.7–3 ug/L MC-NOD: 0.5–5 ug/L	Available for CYN, MC-NOD Test results in 10–15mins No/low entry cost Low per sample cost at $30 (25 cartridges for $750) Limited staff training necessary Moderate quantification limits	Not available for ATX, SAX, but ATX test in development and will be compatible with the reader for CYN, MC-NOD Screening method only—not actionable/requires immediate follow up sampling with quantitative method High entry cost (~$6,000), requires factory-calibrated LightDeck MINI Instrument, computer with Windows 7 or higher and USB 2.0 or higher interface Time-consuming to run more than 5 samples for a given event Low ends of quantification limits are moderately high Test range likely to require in-field sample dilution during intense blooms, which would require the same sample to be analyzed more than one time Cannot distinguish between MC and NOD

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Microtiter plate kit for in-house Enzyme-Linked Immunosorbent Assays (ELISA) (Eurofins)	No specific method no.	ATX: 0.165–5.5 ug/L	Available for ATX, CYN, MC-NOD, SAX Test results in ~ 2–3 hours Moderate per sample cost at $110–$150 (assuming 5–9 sites per event) Actionable quantitative results Low quantification limits	High entry cost (~$6,500), requires ELISA plate reader and kit materials, multi-channel pipettor Requires dedicated laboratory space and moderate degree of training for 1–2 dedicated staff Test range likely to require laboratory sample dilution during intense blooms, which would require the same sample to be analyzed more than one time Cannot distinguish between ATX and homoanatoxin, or between MC and NOD
		CYN: 0.05–2.0 ug/L		
	USEPA Method 546	MC-NOD: 0.15–5.0 ug/L		
	No specific method no.	SAX: 0.22–0.44 ug/L		
Analytical laboratory ELISA	No specific method no.	ATX: 0.165–5.5 ug/L	Available for ATX, CYN, MC-NOD, SAX No/low entry cost Moderate per sample cost at $105–$130 per toxin + $50 overnight shipping per sample = $155–$180 per sample Limited staff training necessary No equipment necessary Independent quantitative cyanotoxin results Analytical laboratory handles any necessary sample dilution, and sonication and freezing/thawing to lyse cells Low quantification limits	3–4 d for test results; expedited samples (e.g., next day) possible at higher cost (~$200 per sample + $50 shipping) Samples can be lost/broken during shipping Cannot distinguish between ATX and homoanatoxin, or between MC and NOD
		CYN: 0.05–2.0 ug/L		
	USEPA Method 546	MC-NOD: 0.15–5.0 ug/L		
	No specific method no.	SAX: 0.22–0.44 ug/L		

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Liquid chromatography triple quadrupole mass spectrometry (LC/MS/MS)	USEPA Method 545	ATX: 0.04– >10,000 ug/L	Available for ATX, CYN, MC-NOD, SAX No/low entry cost Moderate per sample cost at $100 per toxin + $50 per additional toxin + $50 overnight shipping per sample = $300 per sample for all toxins or $150 per sample for a single toxin Limited staff training necessary No equipment necessary Independent quantitative cyanotoxin results Analytical laboratory handles sample dilution Lowest quantification limits Can distinguish between ATX and homoanatoxin, and between MC and NOD	3–4 d for test results Samples can be lost/broken during shipping Limited analytical standards available across the large number of toxin congeners (i.e., chemical variants), so test may underestimate total toxin concentration
		CYN: 0.04– >10,000 ug/L		
	USEPA Method 544	MC-NOD: 0.04– >10,000 ug/L		
	No specific method no.	SAX: 0.04– >10,000 ug/L		

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Molecular/cyanobacteria genetics				
Quantitative real–time PCR (qPCR)[2]	No specific method no.	100–100,000,000 copies/mL	Available for total cyanobacteria, ATX-producing cyanobacteria, CYN-producing cyanobacteria, MC-NOD producing cyanobacteria, and SAX-producing cyanobacteria No/low entry cost Low per sample cost at $45 per toxin + $50 overnight shipping per sample = $95 per sample Limited staff training necessary No equipment necessary Early warning that cyanotoxins may occur – test indicates that cyanobacteria with the genes to produce toxins are present as well as the number of genes Ratio of toxin-producing bacteria to total cyanobacteria is relatively low-cost analog for cell counts/abundance of known cyanotoxin producing bacteria, if data are collected and analyzed for long term patterns	3–4 d for test results Screening method only—requires immediate follow up sampling with quantitative method for cyanotoxin concentrations Toxin gene abundance is not correlated with cyanotoxin concentration; cyanotoxin concentrations need to be non-detectable for weeks to months to use qPCR as an early warning mechanism Samples can be lost/broken during shipping

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
DNA sequencing	No specific method no.	Non-metric scaling (0-100% of gene diversity sequenced for each sample)	Available for ATX, CYN, MC-NOD, SAX Positively identify cyanobacteria species producing each cyanotoxin	3 weeks for test results Moderate per sample cost at ~\$100, but high total cost because need to run 100 or more samples to justify \$4,000 entry cost for flow cell to run batch samples (minimum \$10,000–\$12,000 total cost) Samples can be lost/broken during shipping
Microscopy				
In-house cell counts/abundance of cyanobacteria species	Microscopy	0–10,000,000 cells/mL	Direct and quantitative measure of total phytoplankton and total cyanobacteria abundance with generally understood and established relationships between cell counts and water quality, including amongst algaecide applicators Early warning that cyanotoxins may occur by identifying whether cyanobacteria known to produce toxins are present and at what abundance relative to other types of phytoplankton Test results in ~ 2–3 hours Develops in-house expertise with reservoir and algae management	Inherent inter-sample variability Does not quantify cyanotoxins Does not indicate whether known cyanotoxin-producing species present in the sample are actually likely to produce toxins High entry cost (~\$10,000), requires microscope and in-house staff with taxonomic identification skills Substantial training necessary

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Analytical laboratory cell counts/ abundance of cyanobacteria species	Microscopy	0–10,000,000 cells/mL	Direct and quantitative measure of total phytoplankton and total cyanobacteria abundance with generally understood and established relationships between cell counts and water quality, including amongst algaecide applicators Early warning that cyanotoxins may occur by identifying whether cyanobacteria known to produce toxins are present and at what abundance relative to other types of phytoplankton No/low entry cost High per sample cost at \$225 + \$50 overnight shipping per sample = \$275 per sample Limited staff training necessary No equipment necessary	3 weeks for test results Inherent inter-sample variability Test does not quantify cyanotoxins Does not indicate whether known cyanotoxin-producing species present in the sample are actually likely to produce toxins Cyanobacteria known to produce toxins must be non-detectable for weeks to months to use this as an early warning mechanism Samples can be lost/broken during shipping
Analytical laboratory identification of potentially toxigenic cyanobacteria (PTOX)	Microscopy	N/A	Direct indication of cyanobacteria presence and identification of species that are known toxin producers No/low entry cost Low per sample cost at \$45 per toxin + \$50 overnight shipping per sample = \$95 per sample Limited staff training necessary No equipment necessary	Qualitative screening method only – requires immediate follow-up sampling with quantitative method for cell counts and/or cyanotoxin concentrations 3–4 days for test results Samples can be lost/broken during shipping

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Remote sensing				
Cell counts (US EPA CyAN)	Spectral imaging of algal pigmentation (chlorophyll-a and phycocyanin)	10,000–7,000,000 cells/mL	No cost – data are collected by US EPA for general use Limited staff training necessary No equipment necessary Cyanobacteria index used to approximate abundance and converted to cells/mL using published approximate scalar relationship 7-day maximum cyanobacteria cell counts aggregated and updated weekly, and beginning late July 2020, daily snapshots of cyanobacteria cell counts at re-occurring location in the lake Daily measurements between 11 am and 1 pm local time capture maximum or near maximum surface accumulations Historical data available for Lake Henshaw from January 2019 to current.	Data not available for Lake Wohlford save a short period in 2019 (satellites use 300m x 300m pixels [~22 acres], which requires that lakes be large enough to fit multiple pixels of this size) Surface measurements only Weekly aggregated results not actionable or early warning mechanism due to time delay Daily non-aggregated results are from a single pixel coordinate which is inherently spatially variable, thus not actionable without corroborating on-the-ground measurements Cell counts returned not easily relatable to cyanotoxin concentrations This technology is still in the development stage. In the future this may be a more useful tool.

Test/Measurement Type	Analytical Method	Test Range	Pros	Cons
Cyano Index (SFEI HAB Satellite Analysis Tool)	Spectral imaging of algal pigmentation (chlorophyll-a and phycocyanin)	10,000–7,000,000 cells/mL	No cost – data are collected by US EPA for general use Limited staff training necessary No equipment necessary 10-day maximum cyanobacteria cell counts aggregated and updated; can report for 10-day to 90-day intervals Daily measurements between 11 am and 1 pm local time capture maximum or near maximum surface accumulations Historical data available June 2002–December 2020	Data not available for Lake Wohlford (satellites use 300m x 300m pixels [~22 acres], which requires that lakes be large enough to fit multiple pixels of this size) Cyanobacteria index not easily relatable to other measures of phytoplankton biomass (e.g., chlorophyll-a, cell counts) or cyanotoxin concentrations Surface measurements only Weekly aggregated results not actionable or early warning mechanism due to time delay No daily results—satellite imagery collected every 1–3 days

ATX = anatoxin-a
CYN = cylindrospermopsin
MC = microcystin(s)
NOD = nodularin(s)
RFU = Relative Fluorescence Unit
SAX = saxitoxin

Appendix E

Assessment of May 2022 Algaecide Treatment Effectiveness for Lake Henshaw

Stillwater Sciences
2855 Telegraph Ave, Suite 400, Berkeley, CA 94705
phone 510.848.8098

TECHNICAL MEMORANDUM

DATE:	July 18, 2022
TO:	Don Smith, Vista Irrigation District
FROM:	Maia Singer, Avi Kertesz, and Wayne Swaney, Stillwater Sciences
SUBJECT:	Assessment of May 2022 algaecide treatment effectiveness for Lake Henshaw

1 INTRODUCTION

In March 2020, the Vista Irrigation District (District) began monitoring for the presence of cyanobacteria and cyanotoxins in Lake Henshaw after being alerted to the potential presence of harmful algal blooms (HABs) in the lake by remote sensing data. Since then, routine monitoring and laboratory analysis have confirmed the presence of elevated levels of the cyanotoxins microcystin and anatoxin-a at multiple sites in the lake and in water released to the downstream San Luis Rey River.

The District is currently developing a Draft HABs Management and Mitigation Plan, which outlines protocols for identifying early HAB development and actions that can be taken to minimize cyanotoxin production and associated delays to water deliveries in the short term, while longer-term alternatives are developed and implemented to prevent future blooms. As part of Draft HABs Management and Mitigation Plan development, application of copper- and/or peroxide-based algaecides has been identified as the most feasible short-term HABs control method for Lake Henshaw for the following reasons:

- Algaecide application is a well-proven mitigation method for HABs. Approved algaecide chemicals act quickly (i.e., minutes to hours) and can prevent the formation of and interrupt an ongoing HAB and to stop cyanotoxin production.
- Little to no capital investment is required for algaecide application, since licensed applicators can be hired by the District to apply the chemicals and undertake monitoring needed to meet permit requirements.
- Costs are generally predictable and there are multiple algaecide products available on the market.

In June 2021, the District obtained a Statewide Aquatic Weed Control Permit for application of copper sulfate, chelated copper, and sodium carbonate peroxyhydrate (peroxide) to control HABs in Lake Henshaw. The District desires to obtain experience with the use of both copper- and peroxide-based algaecides in the lake over time.

Throughout 2021, persistent cyanotoxin concentrations above the California Cyanobacteria Harmful Algal Bloom (CCHAB) Network “caution” thresholds (i.e., 0.8 µg/L and detection for microcystin and anatoxin-a, respectively) in Lake Henshaw hindered the District's ability to deliver water on behalf of itself, groups represented by the San Luis Rey Indian Water Authority (IWA), including the La Jolla and Rincon Bands, and the City of Escondido. Cyanotoxin concentrations in the lake dropped below the CCHAB caution thresholds in early 2022 and the District subsequently released water from Henshaw Dam. However, persistent low-level microcystin concentrations (<0.5 µg/L) and several subsequent anatoxin-a detections both in Lake Henshaw and at downstream sampling sites in the San Luis Rey River prompted the District to initiate the first algaecide treatment of Lake Henshaw to assess lake response.

In accordance with the State Water Board approved *Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch* (Marine Biochemists 2021), the District applied 40,000 pounds of SePRO PAK 27 (active ingredient sodium carbonate peroxyhydrate 85%) to Lake Henshaw on March 14 and 15, 2022. The resulting dose of hydrogen peroxide in the 40% of lake surface area that was treated, assuming an average 5-foot water depth, was approximately 2.9 mg/L (ppm). Averaged across the entire lake surface, the hydrogen peroxide dose was 1.1 mg/L, although the latter estimate assumes complete mixing immediately following dosing, which is unlikely to have occurred. The March 2022 algaecide treatment in Lake Henshaw appeared to have a minor effect on HABs, with variable and modest changes in chl-*a*, the ratio of phe-*a* to chl-*a*, total cyanobacteria cell densities, and nutrient concentrations depending on the amount of time elapsed since treatment. Microcystin concentrations doubled at shoreline sites following treatment, and either increased or decreased slightly at other sites following treatment (Stillwater Sciences 2022).

In order to support the release of recreational water to the San Luis Rey River over Memorial Day weekend, the District implemented a second algaecide treatment in Lake Henshaw beginning on May 16, 2022, with the goal of minimizing cyanotoxin concentrations in the lake leading up to the holiday weekend. Due to the current IWA preference for peroxide-based treatment products, the District applied SePRO PAK 27, consistent with the March 2022 application. The southern portion of the lake was treated with 20,000 pounds of SePRO PAK 27 on May 16 between 12 noon and 3:30 pm. The northern portion of the lake was treated with 40,000 pounds on May 17 between 7 am and 3 pm. The southern portion of the lake was treated again with 28,000 pounds on May 18 between 7 am and 2 pm. The northern portion of the lake was treated again with 32,000 pounds on May 19 between 6:45 am and 2 pm (AquaTechnex 2022). Two boats were used each day for treatment. Over the course of four days, the entire lake surface area (approximately 771 acres) was treated with 120,000 pounds of SePRO PAK 27. The Lake Henshaw boat applicator tracks were spaced approximately 100 feet apart and the treatment boom width was approximately 50 feet across, such that approximately 50% of the lake surface area was treated on May 16 and May 17. By returning to treat between the tracks made on the first two days, AquaTechnex attempted to treat the other 50% of the lake surface area on May 18 and May 19, 2022. Given likely overlap between tracks on treatment days, AquaTechnex estimates that approximately 80–90% of the lake surface area was treated from May 16 to May 19. The May treatment corresponded to a concentration of approximately 2.2 to 4.3 mg/L (ppm) on any given day, or 3.3 mg/L on average, assuming an average 5-foot water depth. Averaged across the entire lake surface, the hydrogen peroxide dose was 1.6 mg/L on any given day, although the latter estimate assumes complete mixing immediately following dosing, which is unlikely to have occurred.

This technical memorandum provides an assessment of the effectiveness of the second algaecide treatment in Lake Henshaw in May. The methodology, results, and conclusions of the water quality monitoring effort associated with the May algaecide treatment are described below, including comparisons to the March treatment results where applicable.

2 METHODS

To inform the assessment of algaecide treatment effectiveness in May, the District re-occupied water quality monitoring sites used for the March treatment effectiveness monitoring, including four routine monitoring sites (H-S, H-FD, H-BL, H-BLS) and seven additional open water and shoreline sites (Table 1). The District also included monitoring of *in situ* water quality parameters and additional analytical constituents before and after the May treatment event.

In situ water quality parameters included water temperature, dissolved oxygen (DO), conductivity, total dissolved solids, pH (standard units [s.u.]), oxidation reduction potential (ORP), and turbidity (Formazin Nephelometric Units [FNU]). *In situ* measurements were taken in the morning (between approximately 7:00 am and 10:30 am) and the afternoon (between approximately 12:00 pm and 3:00 pm) at five deep water sites (H-BL, H-FD, H-ML, H-NL, H-SL) on 5/16, 5/17, 5/18, 5/19, and 5/23/2022 and were made with a calibrated YSI DSS multiprobe.

Chlorophyll-a, pheophytin-a, and nutrients (total nitrogen, nitrate, ammonia, total phosphorus, orthophosphate), microcystin, and anatoxin-a, were sampled in the morning (between approximately 7 am and 11 am) on 5/16, 5/17, 5/18, 5/19, 5/23, 5/31, and 6/6/2022. Sampling occurred at each monitoring site according to the following schedule:

- 5/16: All sites
- 5/17: All sites except H-BL, H-BLS, H-NL, and H-NS
- 5/18: All sites except H-SL and H-SS
- 5/19: Only H-NS and H-ES
- 5/23: All sites
- 5/31: Only H-S, H-FD, H-BL, and H-MLD (except cyanotoxins)
- 6/6: Only H-FD, H-BL, and HBLS (except cyanotoxins)

Samples were shipped overnight to the analytical laboratory (Bend Genetics, Sacramento, California) and analyzed using the fluorometric (acidification) method (EPA 445) for chlorophyll-a and pheophytin-a; persulfate digestion and spectrophotometric methods 10208 (total nitrogen) and 10210 (total phosphorus); spectrophotometric methods 10209 (orthophosphate), 10205 (ammonia), and 10206 (nitrate); and enzyme linked immunosorbent assay (ELISA) for total anatoxin-a and total microcystin/nodularin concentrations.

Cyanobacterial counts by genus were sampled in the morning (between approximately 7 am and 11 am) on 5/16, 5/17, 5/18, 5/19, and 5/23/2022 at H-S, H-FD, H-BLS, and H-ML, except H-BLS on 5/17/2022. Grab samples were shipped overnight to the analytical laboratory (Bend Genetics, Sacramento, California) and analyzed using microscopy for identification of potentially toxigenic cyanobacteria (PTOX).

Table 1. Lake Henshaw water quality monitoring sites for algaecide effectiveness monitoring associated with the May 2022 treatment event.

Site ID	Location	Latitude	Longitude
H-S	Southwestern shoreline at beach adjacent to fishing dock	33.23496°N	116.75617°W
H-FD	Southwestern shoreline at the in-water end of the fishing dock in surface waters	33.23544°N	116.75568°W
H-FDD	Southwestern shoreline at the in-water end of the fishing dock in bottom waters	33.23544°N	116.75568°W
H-BLS	Buoy line at dam in surface waters	33.23963°N	116.76174°W
H-BL	Buoy line at dam in bottom waters	33.23963°N	116.76174°W
H-NL	Northern portion of lake in surface waters	33.24600°N	116.75300°W
H-ML	Mid-lake in surface waters	33.23890°N	116.75275°W
H-MLD	Mid-lake in bottom waters	33.23890°N	116.75275°W
H-SL	Southern portion of lake in surface waters	33.23000°N	116.74400°W
H-NS	Northern shoreline at beach	33.24729°N	116.75414°W
H-ES	Eastern shoreline at beach	33.23546°N	116.73801°W
H-SS	Southern shoreline at beach	33.22659°N	116.74316°W

Figure 1. Lake Henshaw water quality monitoring sites for algaecide effectiveness monitoring associated with the May 2022 treatment event.

3 RESULTS

3.1 *In situ* Water Quality

Water quality results for a subset of the *in situ* measurements are summarized below and shown in Figures A-1 through A-5 (Appendix A).

3.1.1 Water temperature

Water temperatures at the deeper open water sites remained relatively stable throughout the water column across all sampling dates and for both morning and afternoon *in situ* measurements in

May 2022, ranging 17–23°C, with some slight increases in the upper water column, particularly during the afternoon. While water temperatures measured in May 2022 were several degrees higher than values measured during March 2022 (12–15°C; Stillwater Sciences 2022), similar to the March measurements there was no evidence of a defined thermocline at the open water sites during the May sampling dates. Sites H-ML (5/18 PM, 5/23 AM), H-NL (5/18 PM), and H-SL (5/17 PM, 5/18 PM) did exhibit surface water concentrations that were 1–2°C greater than bottom water concentrations, although changes in temperature occurred gradually with depth throughout the water column (Figures A-1 to A-5).

3.1.2 Dissolved oxygen

Across all sampling dates, DO readings were variable, ranging 0.3–10.9 mg/L (3–125% saturation) with an average of 6.6 mg/L (73% saturation) and exhibiting generally lower values than were measured before and after the March 2022 algaecide treatment (Stillwater Sciences 2022). At the deepest site near the dam (H-BL), peak concentrations occurred in surface waters on the morning of 5/16 prior to algaecide treatment (8.4 mg/L and 89% saturation). Concentrations in bottom waters on the pre-treatment sampling date decreased to 3.2 mg/L (33.5% saturation), suggesting that a fair amount of water column and/or sediment oxygen demand was present prior to algaecide treatment. On 5/17, one day following the first day of treatment and throughout the second day of treatment, DO concentrations at Site H-BL were relatively unchanged. However, DO concentrations gradually decreased in both surface waters and bottom waters on 5/18 and 5/19 following the third and fourth days of treatment, respectively. The lowest DO concentrations measured in surface waters (~ 4 mg/L and ~50% saturation) and in bottom waters (< 1 mg/L and <10% saturation) occurred at this site on 5/19, the fourth day of treatment (Figure A-1b), suggesting an effect of algaecide on the bloom (e.g., oxidative stress, cell death), particularly in deeper waters where re-oxygenation from the atmosphere occurs more slowly. By 5/23, DO concentrations had largely recovered to pre-treatment concentrations throughout the water column (Figure A-1b).

DO concentrations at other sites in Lake Henshaw were less obviously affected by algaecide treatment relative to pre-treatment values, although the relatively deep Site H-FD also exhibited the lowest DO values (3 mg/L and 33% saturation; Figure A-2b) in bottom waters on 5/19 after the fourth day of treatment, as did the less deep Site H-NL in mid-water and bottom waters on 5/19 (2.2 mg/L and 44% saturation; Figure A-4b).

3.1.3 Turbidity

Across all sampling dates, turbidity readings were variable throughout the water column at open water sites, ranging 30–90 FNU. Similar to the March dataset, there was no obvious turbidity pattern with water depth or across sites that would suggest pronounced accumulation of algae, either before, during, or after algaecide treatment. There was no consistent vertical correspondence between turbidity readings and DO, where high turbidity and high DO might indicate an accumulation of photosynthesizing algae, and high turbidity and low DO might indicate an accumulation of senescing algae following algaecide treatment.

3.1.4 pH/ORP

pH readings were stable throughout the water column on all sampling dates, remaining near 9.0 at all sites. Elevated pH in eutrophic lakes like Lake Henshaw is typically indicative of high rates of photosynthesis. ORP ranged from approximately +111 to +236 mV on all sampling dates and was generally consistent with depth. Positive ORP indicates oxidizing conditions, which is consistent with the moderate to high DO concentrations measured in surface waters on most sampling dates. Relatively lower DO (< 4 mg/L) in bottom waters of deeper water sites (i.e., H-BL, H-FD) did not correspond to a decrease in ORP, with the exception of DO concentrations less than 0.5 mg/L on the afternoon of 5/19 when ORP rapidly decreased from approximately 175 mV to 120 mV at the bottom of the water column (Figure A-1b). Consistent with the March 2022 pre- and post-algaecide treatment monitoring results, the relatively higher ORP despite occasional low DO suggests that the lower DO was ephemeral. ORP values supporting denitrification (i.e., reduction of nitrate [NO_3^-] to nitrogen gas [N_2]) tend to be in the range -50 to +50 mV, such that the Lake Henshaw ORP values may have been too high to support water column denitrification.

3.1.5 Conductivity

Conductivity readings were generally stable throughout the water column on all sampling dates and at all sites, with a range of 545–685 µS/cm and an average reading of 624 µS/cm (data not shown in Appendix A). These values were generally consistent with those measured before and after the March 2022 algaecide treatment. Conductivity values less than 1,000 µS/cm for lakes and reservoirs are generally considered to be moderate. There was no pattern with water depth or across sites before, during, or after treatment.

3.1.6 Chlorophyll-*a* and pheophytin-*a*

For the May treatment event, only a single pre-treatment sampling result is available for comparison at H-FDD, H-NL, H-ML, H-MLD, H-SL, H-NS, H-ES, and H-SS, whereas a longer period of record is available for pre-treatment results at the routinely sampled synoptic monitoring sites (H-S, H-FD, H-BLS, and H-BL). Pre-and post-treatment chlorophyll-*a* (chl-*a*) concentrations (a measure of algal biomass) in Lake Henshaw ranged 16–79 µg/L and varied by sampling site (Table 2). Comparisons between chl-*a* concentrations at H-FD and H-FDD, H-BLS and H-BL, and H-ML and H-MLD reveal no consistent pattern between surface and depth samples at open water sites. Peak chl-*a* concentrations varied between surface and depth samples at H-BLS and H-BL, and H-ML and H-MLD, depending on sampling date, while concentrations were generally lower in depth samples collected at H-FDD and H-FD. Average chl-*a* concentrations were greater than 50 µg/L at all sites on all sampling dates, indicating eutrophic conditions in late spring/early summer 2022.

Chl-*a* concentrations measured before and after the May 2022 algaecide treatment were generally lower than those measured before and after the March 2022 algaecide treatment (Figure 2). The overall average pre-treatment chl-*a* concentration in March was 96±27 µg/L (n=23; Stillwater Sciences 2022), while that of May was 55±14 µg/L (n=24), indicating that algal biomass levels were roughly 60% lower in Lake Henshaw just prior to the second algaecide treatment as compared with the first. Post-treatment, overall average chl-*a* concentration in March/April was 82±16 µg/L (n=43; Stillwater Sciences 2022), while that of May/June was 48±17 µg/L (n=40), indicating that algal biomass levels were roughly 60% lower in Lake Henshaw just following the second algaecide treatment as compared with the first.

Overall, chl-*a* concentrations decreased modestly (i.e., by 10%–50%) following the second algaecide treatment at the deeper open water sites (H-FDD, H-BL, H-NL, H-ML, H-SL) and three shoreline sites (H-NS, H-ES, H-SS) (Figure 2 and Table 2). Chl-*a* concentrations increased modestly at one shoreline site (H-S) and remained relatively unchanged at two open water sites (H-FD and H-BLS) following the May algaecide treatment. The 10%–50% decrease in May/June chl-*a* at deeper open water sites following algaecide treatment was somewhat greater than the decrease in March/April at the same sites (i.e., by 10%–40%; Stillwater Sciences [2022]), where 9 of 12 sites (including both surface and bottom samples at H-FD, H-BL, and H-ML) experienced a decrease in chl-*a* concentrations after the second treatment in May and 8 of 12 sites (Table 2) experienced a decrease after the first treatment in March (Stillwater Sciences 2022).

Similar to conditions following treatment in March, the May pheophytin-*a* (phe-*a*) concentrations were generally lower than chl-*a* samples collected at the same location on the same date. Phe-*a* ranged 10–68 μg/L throughout the May pre- and post-treatment period and was lowest at all sites on 5/23/2022 (Table 2).

The ratio of phe-*a* to chl-*a* was similar across sampling sites and dates, generally between 0.5–0.8. Ratios tended to be higher following algaecide treatment, which is to be expected from senescing (dying) algae, but to a lesser degree than following the March 2022 treatment (Stillwater Sciences 2022).

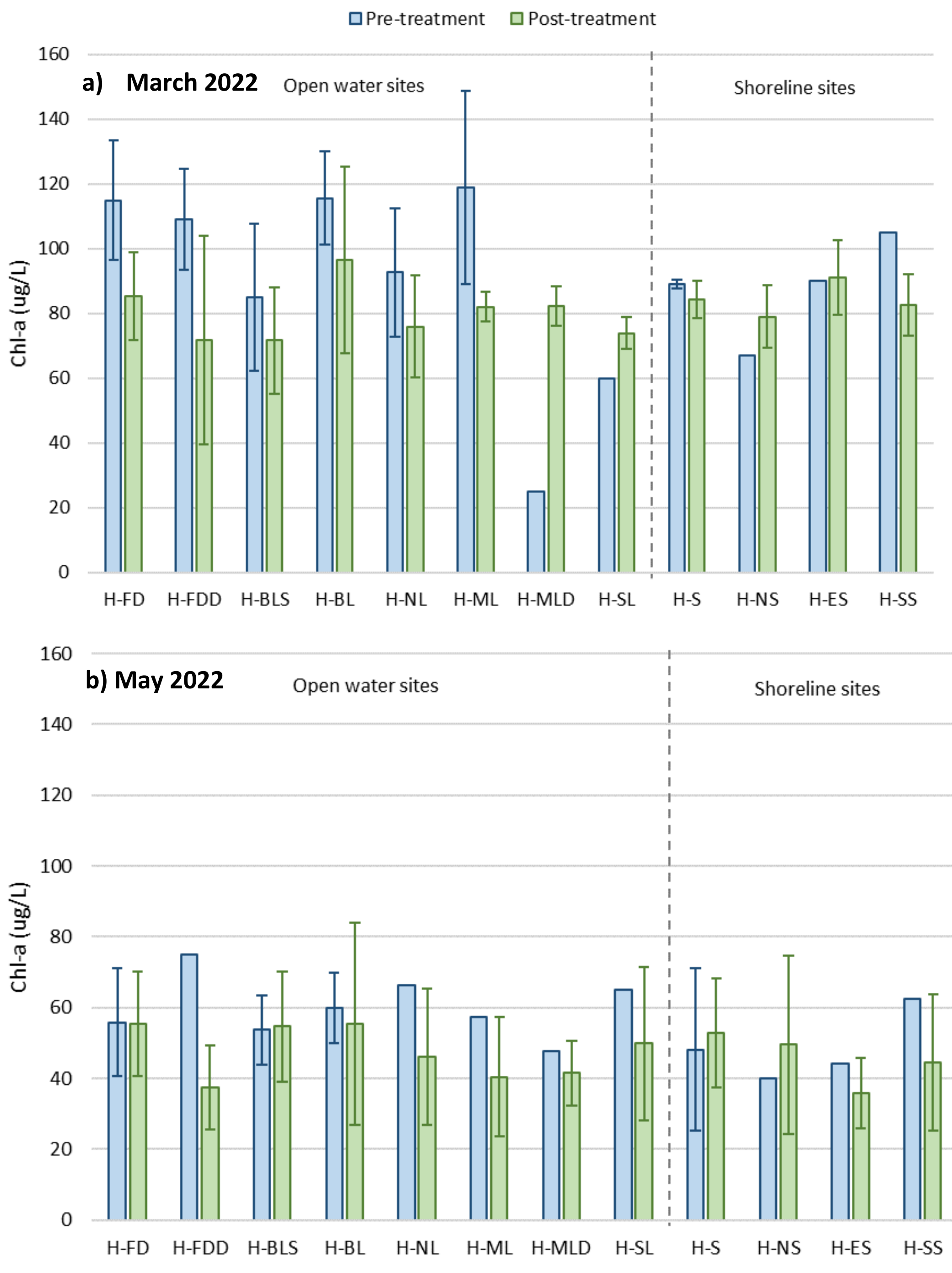


Figure 2. Chlorophyll-a (chl-a) concentrations at Lake Henshaw open water sites (to the left of the vertical dashed line) and shoreline sites (to the right of vertical dashed line) before and after the a) March 2022 and b) May 2022 peroxide-based algaecide treatments. Data are presented as average ±1 standard deviation, with number of samples per site and sampling dates for May 2022 presented in Table 2. Details for March 2022 presented in Stillwater Sciences (2022). Bars without standard deviations represent results from a single sample.

Table 2. Chlorophyll-*a* and pheophytin-*a* concentrations measured in Lake Henshaw before and after the May 2022 peroxide-based algaecide treatment.

Date	Site ID											
	H-S	H-FD	H-FDD	H-BLS	H-BL	H-NL	H-ML	H-MLD	H-SL	H-NS	H-ES	H-SS
***Chlorophyll-a* (µg/L)**												
4/25/2022	16	37	-	44	56	-	-	-	-	-	-	-
5/2/2022	54	54	-	53	58	-	-	-	-	-	-	-
5/9/2022	70	75	-	51	52	-	-	-	-	-	-	-
5/16/2022	53	58	75	67	74	66	57	48	65	40	44	62
Average Pre-treatment	*48*	*56*	*75*	*54*	*60*	*66*	*57*	*48*	*65*	*40*	*44*	*62*
5/17/2022	56	63	50	-	-	-	35	48	66	-	39	51
5/18/2022	50	43	37	67	79	60	59	45	-	67	40	-
5/19/2022	-	-	-	-	-	-	-	-	58	-	43	60
5/23/2022	34	36	26	34	18	33	27	31	25	32	21	23
5/31/2022	71	68	-	50	75	-	-	-	-	-	-	-
6/6/2022	-	66	-	67	49	-	-	-	-	-	-	-
Average Post-treatment	*53*	*55*	*38*	*55*	*55*	*46*	*40*	*41*	*50*	*50*	*36*	*44*
Average % Difference	*10%*	*-1%*	*-50%*	*2%*	*-7%*	*-31%*	*-30%*	*-13%*	*-23%*	*23%*	*-19%*	*-29%*
***Pheophytin-a* (ug/L)**												
4/25/2022	11	24	-	27	42	-	-	-	-	-	-	-
5/2/2022	32	36	-	30	34	-	-	-	-	-	-	-
5/9/2022	47	44	-	29	39	-	-	-	-	-	-	-
5/16/2022	31	36	48	37	49	39	33	30	39	26	28	39
Average Pre-treatment	*30*	*35*	*48*	*31*	*41*	*39*	*33*	*30*	*39*	*26*	*28*	*39*
5/17/2022	33	37	35	-	-	-	32	32	40	-	26	42
5/18/2022	45	45	34	55	65	54	53	41	-	68	38	-

Date	Site ID											
	H-S	**H-FD**	**H-FDD**	**H-BLS**	**H-BL**	**H-NL**	**H-ML**	**H-MLD**	**H-SL**	**H-NS**	**H-ES**	**H-SS**
5/19/2022	-	-	-	-	-	-	-	-	48	-	49	62
5/23/2022	17	18	15	18	10	16	13	18	14	18	19	14
5/31/2022	40	37	-	29	50	-	-	-	-	-	-	-
6/6/2022	-	40	-	38	32	-	-	-	-	-	-	-
Average Post-treatment	*34*	*35*	*28*	*36*	*38*	*35*	*33*	*31*	*34*	*43*	*33*	*39*
Average % Difference	*11%*	*1%*	*-41%*	*19%*	*-8%*	*-11%*	*-2%*	*1%*	*-12%*	*63%*	*17%*	*1%*
Ratio **(Phe-*a*:Chl-*a*)**												
4/25/2022	0.7	0.6	-	0.6	0.7	-	-	-	-	-	-	-
5/2/2022	0.6	0.7	-	0.6	0.6	-	-	-	-	-	-	-
5/9/2022	0.7	0.6	-	0.6	0.7	-	-	-	-	-	-	-
5/16/2022	0.6	0.6	0.6	0.6	0.7	0.6	0.6	0.6	0.6	0.7	0.6	0.6
Average Pre-treatment	*0.6*	*0.6*	*0.6*	*0.6*	*0.7*	*0.6*	*0.6*	*0.6*	*0.6*	*0.7*	*0.6*	*0.6*
5/17/2022	0.6	0.6	0.7	-	-	-	0.9	0.7	0.6	-	0.7	0.8
5/18/2022	0.9	1.0	0.9	0.8	0.8	0.9	0.9	0.9	-	1.0	0.9	-
5/19/2022	-	-	-	-	-	-	-	-	0.8	-	1.1	1.0
5/23/2022	0.5	0.5	0.6	0.5	0.6	0.5	0.5	0.6	0.6	0.6	0.9	0.6
5/31/2022	0.6	0.5	-	0.6	0.7	-	-	-	-	-	-	-
6/6/2022		0.6		0.6	0.7							
Average Post-treatment	*0.6*	*0.7*	*0.7*	*0.6*	*0.7*	*0.7*	*0.8*	*0.7*	*0.7*	*0.8*	*0.9*	*0.8*
Average % Difference	*7%*	*6%*	*19%*	*13%*	*2%*	*18%*	*31%*	*15%*	*12%*	*20%*	*44%*	*32%*

Shaded cells indicate results from samples collected following algaecide treatment.
- Indicates no sampling occurred.

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3.1.7 Cyanobacterial cell counts

The genera represented in the May/June 2022 cyanobacterial cell counts were *Planktothrix*, *Microcystis*, *Snowella*, *Dolichospermum, and Aphanocapsa*. The first four genera were present in all samples, with *Planktothrix* as the dominant genus. *Aphanocapsa* was only detected at relatively low densities (cells/mL) during the pre-treatment sampling period (i.e., this genus was not detected following algaecide treatment) (Figure 3, Table 3).

Cell densities before and after the March algaecide treatment were generally similar (i.e., the same order of magnitude) for each genus across sampling sites on a given sampling date, indicating little spatial variation in the relative dominance of the different cyanobacteria either before or after the first algaecide treatment (Stillwater Sciences 2022). In contrast, cell densities after the May algaecide treatment generally decreased compared with those measured before the treatment, although inter-site variability was high (Figure 3, Table 3). More specifically, cell densities decreased at most sites following the May treatment, and densities tended to be lowest on 5/18 and 5/23 (48 hours and 7–8 days after algaecide treatment, respectively); cell densities did not return to pre-treatment levels within the sampling period (Table 3). *Dolichospermum* was the exception to this rule in May, where densities of this genus increased by an order of magnitude over the course of the post-treatment sampling period. The highest cell counts for all other genera (and by extension total cyanobacteria) were measured on 5/9/2022, and generally declined by 5/16/2022 (Figure 3, Table 3). Total cyanobacteria cell counts measured during the May 2022 algaecide pre-treatment sampling period were generally higher than those measured during the March 2022 pre- and post-sampling period (Figure 4).

Consistent with the March 2022 algaecide treatment (Stillwater Sciences 2022), patterns in cell biovolume before and after the May treatment were dominated by *Planktothrix* (14.1 μm^3) and *Microcystis* (22.4 μm^3) due to their relatively large size and high abundance.

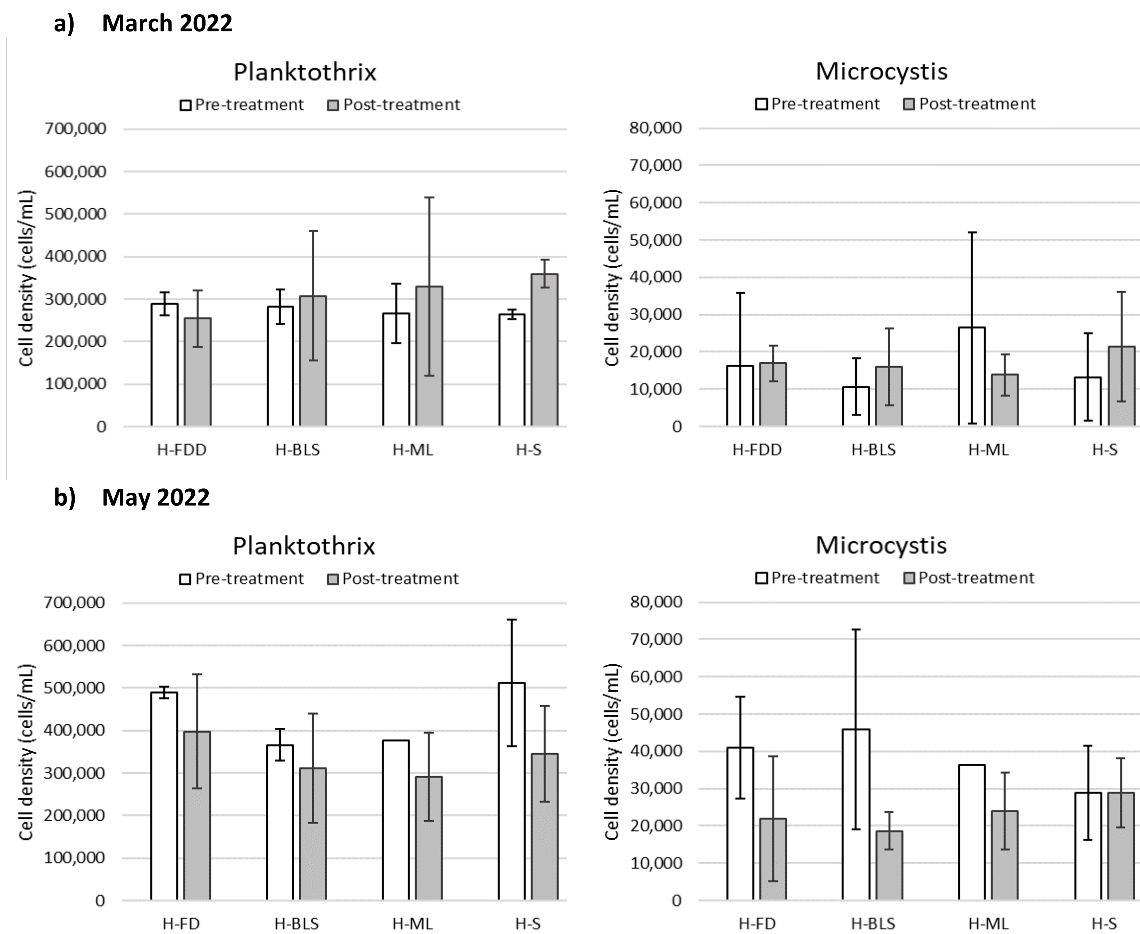
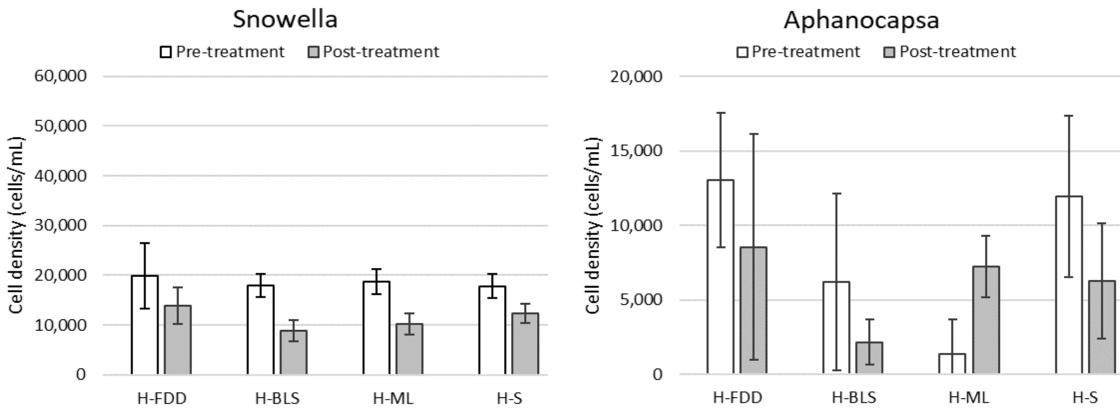


Figure 3. Cyanobacterial cell densities measured in Lake Henshaw before and after the a) March 2022 and b) May 2022 peroxide-based algaecide treatments. Data are presented as average ±1 standard deviation. Number of samples per site and sampling dates for the May 2022 algaecide treatment are presented in Table 3.

a) March 2022



b) May 2022

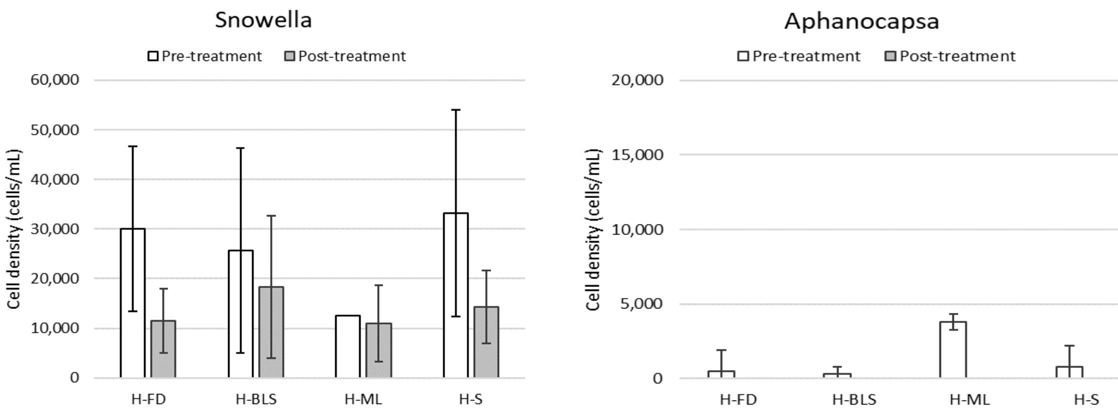
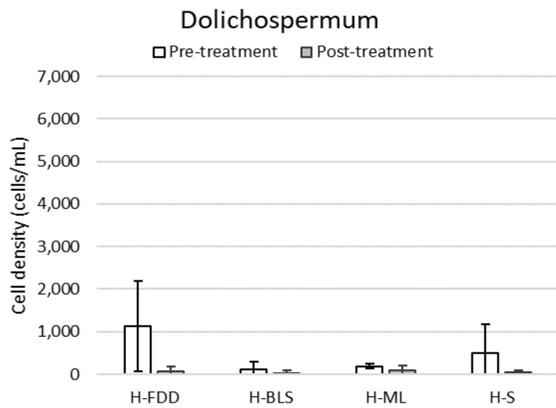


Figure 4. (Cont.) Cyanobacterial cell densities measured in Lake Henshaw before and after the a) March 2022 and b) May peroxide-based algaecide treatments. Data are presented as average ±1 standard deviation. Number of samples per site and sampling dates for the May 2022 algaecide treatment are presented in Table 3.

a) March 2022



b) May 2022

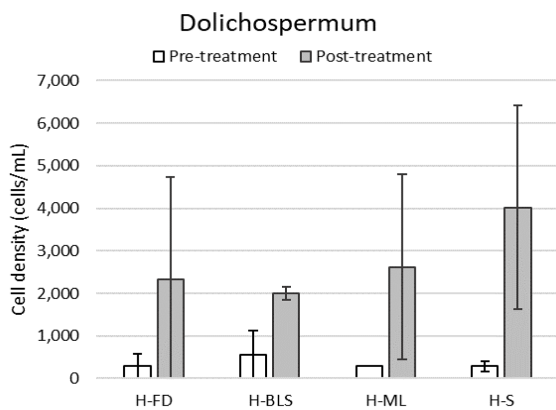


Figure 3. (Cont.) Cyanobacterial cell densities measured in Lake Henshaw before and after the a) March 2022 and b) May 2022 peroxide-based algaecide treatments. Data are presented as average ±1 standard deviation. Number of samples per site and sampling dates for the May 2022 algaecide treatment are presented in Table 3.

Table 3. Cyanobacterial cell densities measured in Lake Henshaw before and after the May 2022 peroxide-based algaecide treatment.

Date	Site ID			
	H-S	H-FD	H-BLS	H-ML
Planktothrix **(cells/mL)**				
5/2/2022	383,408	476,875	381,500	-
5/9/2022	674,355	503,350	323,758	-
5/16/2022	475,823	484,930	392,725	375,650
Average Pre-treatment	*511,195*	*488,385*	*365,994*	*375,650*
5/17/2022	473,787	536,813	-	409,130
5/18/2022	*268,212*	*269,514*	*401,250*	*214,830*
5/23/2022	291,005	384,800	219,336	248,538
Average Post-treatment	*344,335*	*397,042*	*310,293*	*290,833*
Average % Difference	*-33%*	*-19%*	*-15%*	*-23%*
Microcystis **(cells/mL)**				
5/2/2022	23,620	59,050	76,765	-
5/9/2022	43,320	25,270	29,783	-
5/16/2022	19,896	38,678	31,038	36,263
Average Pre-treatment	*28,945*	*40,999*	*45,862*	*36,263*
5/17/2022	30,240	40,488	-	20,160
5/18/2022	19,240	17,760	22,200	35,520
5/23/2022	11,797	7,550	15,100	16,044
Average Post-treatment	*20,426*	*21,933*	*18,650*	*23,908*
Average % Difference	*-29%*	*-47%*	*-59%*	*-34%*
Snowella **(cells/mL)**				
5/2/2022	24,525	31,338	20,438	-
5/9/2022	56,980	46,127	48,840	-
5/16/2022	18,107	12,933	7,760	12,610
Average Pre-treatment	*33,204*	*30,133*	*25,679*	*12,610*
5/17/2022	21,067	16,853	-	11,850
5/18/2022	15,520	13,192	28,453	18,624
5/23/2022	6,450	4,300	8,256	3,363
Average Post-treatment	*14,346*	*11,448*	*18,355*	*10,994*
Average % Difference	*-57%*	*-62%*	*-29%*	*-13%*

Date	Site ID			
	H-S	H-FD	H-BLS	H-ML
***Aphanocapsa* (cells/mL)**				
5/2/2022	0	969	0	-
5/9/2022	2,400	528	0	-
5/16/2022	0	0	908	3,814
Average Pre-treatment	*800*	*499*	*303*	*3,814*
5/17/2022	0	0	-	0
5/18/2022	0	0	0	0
5/23/2022	0	0	0	0
Average Post-treatment	*0*	*0*	*0*	*0*
Average % Difference	*-100%*	*-100%*	*-100%*	*-100%*
***Dolichospermum* (cells/mL)**				
5/2/2022	368	202	46	-
5/9/2022	330	146	413	-
5/16/2022	156	686	1192	304
Average Pre-treatment	*285*	*345*	*550*	*304*
5/17/2022	6,720	1,870	-	175
5/18/2022	2160	187	1890	4320
5/23/2022	3165	4923	2110	3363
Average Post-treatment	*4,015*	*2,327*	*2,000*	*3,842*
Average % Difference	*1310%*	*575%*	*263%*	*1164%*
***Total cyanobacteria* (cells/mL)**				
5/2/2022	431,921	568,434	478,749	-
5/9/2022	777,385	575,421	402,794	-
5/16/2022	513,982	537,227	433,623	428,641
Average Pre-treatment	*574,429*	*560,361*	*438,389*	*428,641*
5/17/2022	531,814	596,024	-	441,315
5/18/2022	305,132	300,653	453,793	273,294
5/23/2022	312,417	401,573	244,802	271,308
Average Post-treatment	*383,121*	*432,750*	*349,298*	*272,301*
Average % Difference	*-33%*	*-23%*	*-20%*	*-36%*

Shaded cells indicate results from samples collected following algaecide treatment.
- Indicates no sampling occurred.

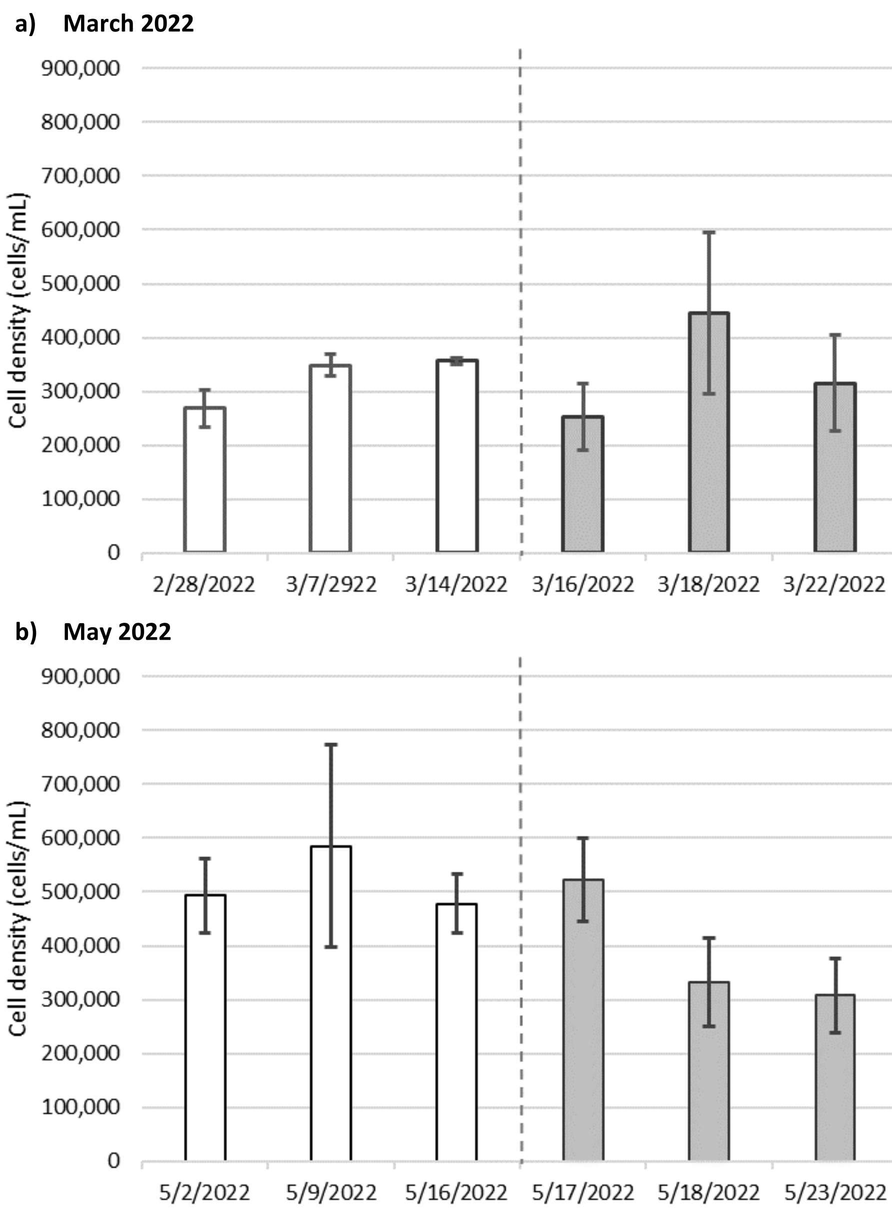


Figure 4. Total cyanobacterial cell densities measured in Lake Henshaw before and after the a) March 2022 and b) May 2022 peroxide-based algaecide treatments. Data are presented as average ±1 standard deviation. Number of samples per site and sampling dates for the May 2022 algaecide treatment are presented in Table 3.

3.1.8 Microcystin and anatoxin-a

Anatoxin-a was detected once (0.15 µg/L) at H-FD on 5/9/2022 (Table 4). Microcystin concentrations were similar throughout the lake during each sampling event (Figure 5, Table 4). Microcystin concentrations ranged 0.13–0.51 µg/L prior to algaecide treatment and 0.22–1.28 µg/L following treatment. Microcystin concentrations were generally stable (between approximately 0.2–0.3 µg/L) prior to algaecide treatment and increased modestly immediately after treatment (Figure 5, Table 4). Microcystin concentrations peaked (between 1.02–1.28 µg/L) at H-S, H-FDD, and H-BLS—the only sites sampled—on 5/31/2022, approximately two weeks after the algaecide treatment, but returned to pre-treatment levels at H-FD and H-BL by the next sampling event. Since the ELISA method used to analyze cyanotoxin concentrations includes a cell lysing step, reported concentrations should represent both microcystin within the cyanobacterial cells and dissolved microcystin in the water column, and any increases following algaecide treatment are expected to be the result of additional cellular production rather than simply cell wall lysing during senescence. Additionally, since peroxide has the potential to chemically break down microcystin during treatment, the higher concentrations post-treatment suggest that either the anticipated breakdown did not occur, or more cyanotoxin was produced by the cyanobacteria during or immediately following treatment such that the net amount measured in the lake post-treatment was still generally greater than concentrations measured prior to treatment.

Table 4. Cyanotoxin concentrations in Lake Henshaw before and after the May 2022 peroxide-based algaecide treatment.

Date	Site ID											
	H-S	**H-FD**	**H-FDD**	**H-BL**	**H-BLS**	**H-NL**	**H-ML**	**H-MLD**	**H-SL**	**H-NS**	**H-ES**	**H-SS**
***Microcystin* (µg/L)**												
4/25/2022	0.13	0.27	-	0.51	-	-	-	-	-	-	-	-
5/2/2022	0.25	0.29	-	0.33	-	-	-	-	-	-	-	-
5/9/2022	0.26	0.23	-	0.29	-	-	-	-	-	-	-	-
5/16/2022	0.29	0.26	0.29	0.32	0.28	0.28	0.26	-	0.30	0.30	0.21	0.31
Average Pre-treatment	*0.23*	*0.26*	*0.29*	*0.36*	*0.28*	*0.28*	*0.26*	-	*0.30*	*0.30*	*0.21*	*0.31*
5/17/2022	0.46	0.29	0.34	-	-	-	0.29	-	0.26	-	0.3	0.34
5/18/2022	0.32	0.24	0.35	0.3	0.24	0.25	0.32	-	-	0.23	0.23	-
5/19/2022	-	-	-	-	-	-	-	-	0.42	-	0.34	0.32
5/23/2022	0.34	0.42	0.32	0.38	0.4	0.36	0.31	-	0.36	0.35	0.3	0.28
5/31/2022	1.02	1.03	-	1.28	-	-	-	-	-	-	-	-
6/6/2022	-	0.22	-	0.24	-	-	-	-	-	-	-	-
Average Post-treatment	*0.54*	*0.44*	*0.34*	*0.55*	*0.32*	*0.31*	*0.31*	-	*0.35*	*0.29*	*0.29*	*0.31*
Average % Difference	*130%*	*68%*	*16%*	*52%*	*14%*	*9%*	*18%*	-	*16%*	*-3%*	*39%*	*1%*
***Anatoxin-a* (µg/L)**												
4/25/2022	<0.15	<0.15	-	<0.15	-	-	-	-	-	-	-	-
5/2/2022	<0.15	0.15	-	<0.15	-	-	-	-	-	-	-	-
5/9/2022	<0.15	<0.15	-	<0.15	-	-	-	-	-	-	-	-
5/16/2022	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	-	<0.15	<0.15	<0.15	<0.15
5/17/2022	<0.15	<0.15	<0.15	-	-	-	<0.15	-	<0.15		<0.15	<0.15
5/18/2022	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	-	-	<0.15	<0.15	-
5/19/2022	-	-	-	-	-	-	-	-	<0.15		<0.15	<0.15
5/23/2022	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	-	<0.15	<0.15	<0.15	<0.15
5/31/2022	<0.15	<0.15	<0.15	-	-	-	-	-	-	-	-	-
6/6/2022	-	<0.15	-	<0.15	-	-	-	-	-	-	-	-

Shaded cells indicate results from samples collected following algaecide treatment.
\- Indicates no sampling occurred

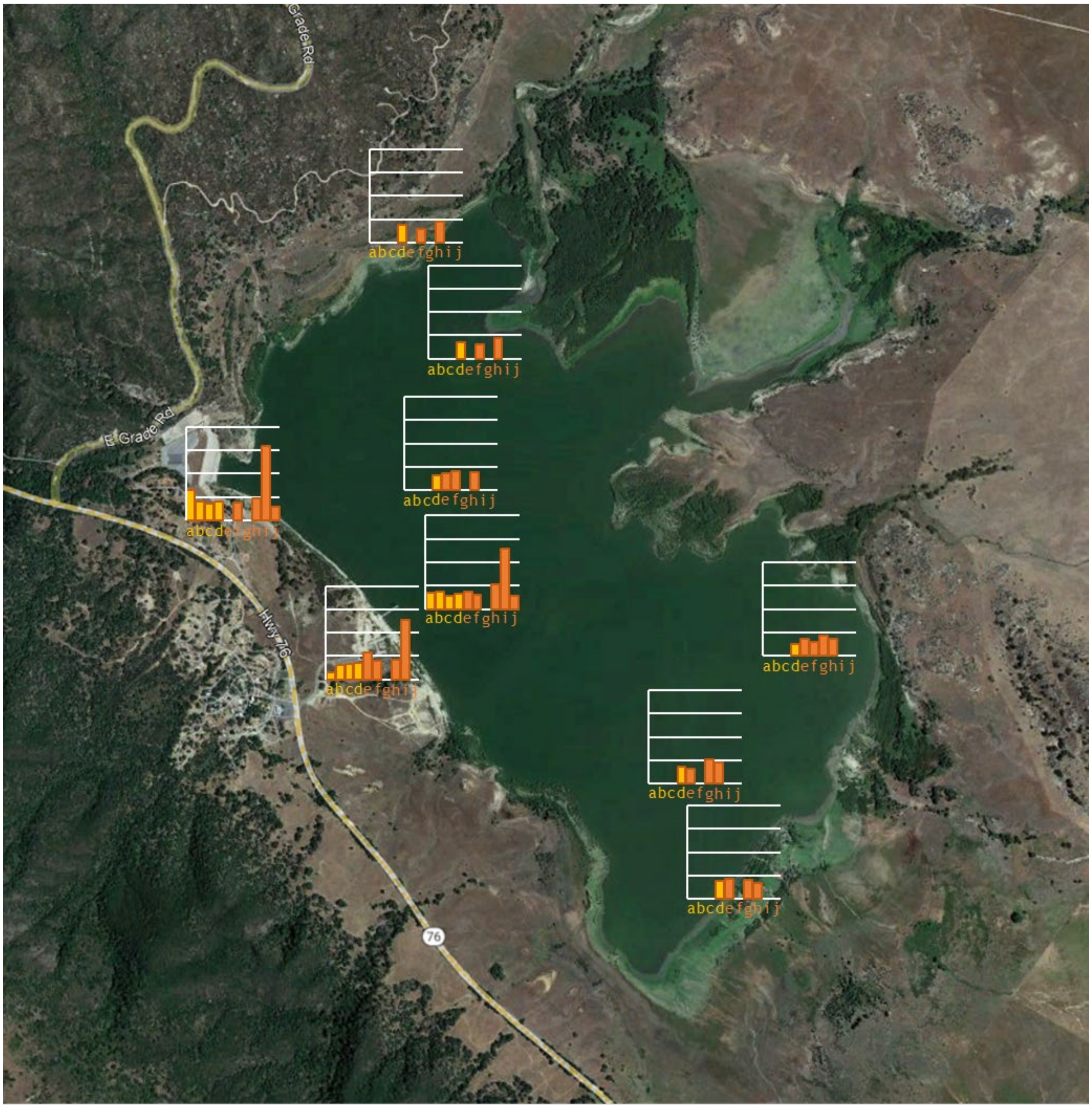


Figure 5. Microcystin concentrations in Lake Henshaw before and after the May 2022 algaecide treatment.

Yellow bars and lower-case letters represent microcystin concentrations from samples collected prior to algaecide treatment. Orange bars and lower-case letters represent microcystin concentrations from samples collected after algaecide treatment. Sampling dates are as follows: a = 4/25/22; b = 5/2/22; c = 5/9/22; d = 5/16/22; e = 5/17/22; f = 5/18/22; g = 5/19/22; h = 5/23/2022; i = 5/31/2022; and j = 6/6/2022. White horizontal lines indicate 0.0, 0.4, 0.8, 1.2, and 1.4 ug/L microcystin. Missing bars indicate that no sampling occurred at a given sampling site on a given date.

3.1.9 Nutrients

Fewer analytical results are available for nutrients sampled prior to algaecide treatment than for the post-treatment sampling period, except for those collected at H-FD, H-BL and H-BLS. In general, most nutrient species were similar before and after treatment, except orthophosphate which was detected but below the MRL at most sites between 4/25 and 5/18/2022, and above the MRL on 5/23, 5/31, and 6/6/2022. Nutrient concentrations were generally similar across sites within given sampling dates.

Total nitrogen concentrations ranged 2.86–5.07 mg/L prior to algaecide treatment and 2.41–4.53 mg/L following treatment (Table 5). Nitrate concentrations were generally similar throughout the sampling period, ranging 0.11–0.23 mg/L and 0.08–0.34 mg/L prior to and following treatment, respectively. Ammonia concentrations ranged 0.01–0.19 mg/L prior to treatment and 0.01–0.08 mg/L following treatment. Total nitrogen was slightly lower than during the March 2022 algaecide application sampling period, and nitrate and ammonia concentrations were generally similar to those measured during the March sampling period (Stillwater Sciences 2022).

Total phosphorous concentrations ranged 0.20–0.29 mg/L prior to algaecide treatment and 0.19–0.31 mg/L following treatment. Orthophosphate concentrations were generally low, ranging 0.02–0.05 mg/L before treatment and 0.05–0.08 mg/L after treatment, with several samples reported between the MDL and MRL. Of all the nutrient species analyzed, orthophosphate exhibited the most consistent increase following algaecide treatment, increasing approximately 50% to 150% between pre- and post-treatment values, depending on site (Table 5), although given several values between the MDL and MRL, there is higher uncertainty in these results. Total phosphorous was slightly lower than during the March 2022 algaecide application sampling period, and orthophosphorus was slightly higher (Stillwater Sciences 2022).

Table 5. Nutrients in Lake Henshaw before and after peroxide-based algaecide treatment.

Date	Site ID				
	H-FD	**H-FDD**	**H-BLS**	**H-BL**	**H-MLD**
Total Nitrogen **(mg/L)**					
4/25/2022	3.64	-	5.07	3.06	-
5/2/2022	3.06	-	3.15	2.86	-
5/9/2022	3.35	-	4.49	3.20	-
5/16/2022	4.02	4.05	4.53	4.32	4.39
5/17/2022	3.20	3.20	-	-	3.15
5/18/2022	3.04	3.27	3.76	3.48	3.18
5/19/2022	-	-	-	-	-
5/23/2022	2.48	3.46	2.41	3.13	2.68
5/31/2022	3.80	-	4.53	3.76	-
6/6/2022	2.73	-	3.06	2.97	-

Date	Site ID				
	H-FD	H-FDD	H-BLS	H-BL	H-MLD
***Nitrate* (mg/L)**					
4/25/2022	0.11	-	0.17	0.11	-
5/2/2022	0.11	-	0.12	0.11	-
5/9/2022	0.17	-	0.23	0.18	-
5/16/2022	0.13	0.14	0.13	0.20	0.13
5/17/2022	0.17	0.15	-	-	0.16
5/18/2022	0.15	0.16	0.17	0.19	0.14
5/19/2022	-	-	-	-	-
5/23/2022	0.34	0.12	0.15	0.08	0.11
5/31/2022	0.10	-	0.15	0.18	-
6/6/2022	0.10	-	0.10	0.11	-
***Ammonia* (mg/L)**					
4/25/2022	0.06	-	0.19	0.08	-
5/2/2022	0.06	-	0.04	0.01	-
5/9/2022	0.04	-	0.11	0.01	-
5/16/2022	0.03	0.01	0.02	0.01	0.01
5/17/2022	0.02	0.02	-	-	0.02
5/18/2022	0.03	0.02	0.03	0.02	0.02
5/19/2022	-	-	-	-	-
5/23/2022	0.01	0.02	0.02	0.03	0.03
5/31/2022	0.02	-	0.08	<MRL	-
6/6/2022	0.02	-	0.02	0.01	-
***Total Phosphorous* (mg/L)**					
4/25/2022	0.25	-	0.20	0.29	-
5/2/2022	0.23	-	0.22	0.23	-
5/9/2022	0.26	-	0.27	0.27	-
5/16/2022	0.26	0.28	0.26	0.29	0.27
5/17/2022	0.25	0.24	-	-	0.23
5/18/2022	0.22	0.26	0.22	0.27	0.22
5/19/2022	-	-	-	-	-
5/23/2022	0.22	0.20	0.19	0.22	0.22
5/31/2022	0.25	-	0.27	0.31	-
6/6/2022	0.25	-	0.29	0.28	-

Date	Site ID				
	H-FD	H-FDD	H-BLS	H-BL	H-MLD
***Orthophosphate* (mg/L)**					
4/25/2022	0.03[C1,J]	-	0.03[C1,J]	0.05	-
5/2/2022	0.02[C1,J]	-	0.02[C1,J]	0.03[C1,J]	-
5/9/2022	0.03[C1,J]	-	0.03[C1,J]	0.03[C1,J]	-
5/16/2022	0.03[C1,J]	0.03[C1,J]	0.03[C1,J]	0.03[C1,J]	0.03[C1,J]
5/17/2022	0.03[C1,J]	0.03[C1,J]	-	-	0.03[C1,J]
5/18/2022	0.03[C1,J]	0.04[C1,J]		0.04[C1,J]	0.03[C1,J]
5/19/2022	-	-	-	-	-
5/23/2022	0.07	0.08	0.07	0.07	0.08
5/31/2022	0.05	-	0.06	0.07	-
6/6/2022	0.05	-	0.06	0.05	-

Shaded cells indicate results from samples collected following algaecide treatment.
[C1,J] Indicates the value of the result is below the MRL but above the threshold of sensitivity for the analytical instrument.
- Indicates no sampling occurred.

4 CONCLUSIONS

The application of 120,000 pounds of a peroxide-based (SePRO PAK 27) algaecide to Lake Henshaw on May 16–19, 2022, appears to have had a minor effect on HABs as evidenced by the below. Note that compared with an earlier dose of SePRO PAK27 in March (i.e., 40,000 pounds, approximately 2.9 mg/L [ppm], applied to approximately 40% of the lake surface area over two days), the May dose was higher (3.3 mg/L [ppm] on average) and covered twice as much area (i.e., 80–90% of the lake surface area) applied over four days.

- Low DO concentrations measured in surface waters (~ 4 mg/L and ~50% saturation) and in bottom waters (< 1 mg/L and <10% saturation) on the fourth day of treatment, suggesting oxidative stress and/or cell death resulting from treatment, particularly in deeper waters where re-oxygenation from the atmosphere occurs more slowly. Within eight days following the onset of treatment, DO concentrations had largely recovered to pre-treatment concentrations throughout the water column.
- Modest decreases in chl-*a* concentrations (i.e., 10%–50% post-treatment decrease) at the deeper open water sites and three shallow shoreline sites, and no change or modest increases at other sites. The 10%–50% decrease in chl-*a* observed at deeper open water sites following the May peroxide dose (higher and across a larger application area) was somewhat greater than the decrease (10%–40% post-treatment decrease; Stillwater Sciences [2022]) observed following the March peroxide dose (lower and across a smaller application area).
- Modest increases in the ratio of phe-*a* to chl-*a* following treatment at most sites, albeit to a lesser degree than following the March 2022 treatment (Stillwater Sciences 2022), suggesting a limited degree of senescing (dying) algae.

- Decreased cyanobacteria cell densities within 48 hours and 7–8 days following algaecide treatment, despite generally higher cell density prior to treatment in May. The exception to this pattern was *Dolichospermum*, which exhibited an order of magnitude increase in cell densities over the course of the May post-treatment sampling period.
- Generally increased microcystin concentrations approximately two weeks following treatment, with a return to pre-treatment levels at the deep open water sites by three weeks post-treatment. Note that since peroxide has the potential to chemically break down microcystin during treatment, the higher concentrations post-treatment suggest that either this did not occur or more cyanotoxin was produced by the cyanobacteria during or immediately following treatment such that the net amount measured in the lake post-treatment was still generally greater than concentrations measured prior to treatment. The small increase in microcystin following the May peroxide dose (higher and across a larger application area) was generally consistent with the small increase observed following the March dose (lower and across a smaller application area).

Conditions in Lake Henshaw during the May peroxide-based algaecide application event were characterized by a lack of thermal stratification at both deep (> 10 feet) and moderate depth (< 10 feet) open water sites, and across open water and shoreline locations alike. Algal activity was high at all sites, as evidenced by occasional supersaturated DO in surface waters, pH > 8.5 throughout the lake, elevated turbidity, and chl-*a* concentrations ranging 16–79 µg/L. DO in May was generally lower than DO measured before and after the March 2022 algaecide treatment, and low DO (3.2 mg/L; 33.5% saturation) in bottom waters at the deepest site near the dam on the pre-treatment sampling date indicates that a fair amount of water column and/or sediment oxygen demand was present prior to algaecide treatment. With the exception of a brief increase in microcystin approximately two weeks following treatment, microcystin (< 0.8 ug/L) and anatoxin-a (<0.015 ug/L) concentrations were low at all sites. Nutrients were relatively low in surface and bottom waters compared with summer and fall 2021 concentrations, although in general nutrients do not appear to be limiting cyanobacteria growth in Lake Henshaw. Of all the nutrient species analyzed, orthophosphate exhibited the most consistent increase following the May algaecide treatment, increasing approximately 50% to 150% between pre- and post-treatment values, depending on site, although there is uncertainty in this result given the low concentrations.

In conclusion, the higher dose of peroxide-based algaecide, applied over a majority of the lake surface in May appears to have had more of an effect on the Lake Henshaw cyanobacterial community as compared with the lower dose and lower application area undertaken in March. However, the overall effect was still minor, which may be due to the generally higher total cyanobacteria cell counts measured during the May algaecide pre-treatment sampling period compared with those measured during the March pre- and post-treatment sampling period. More frequent (e.g., weekly or every other week) and high doses of peroxide across the majority of the lake surface area, and/or a copper-based algaecide, may be required to have a meaningful effect on HABs in Lake Henshaw under conditions where cyanobacteria cell counts are quite high. A combination of algaecides may also provide a better lake response. The potential for benthic cyanotoxin production should be investigated in Lake Henshaw.

5 REFERENCES

AquaTechnex. 2022. Treatment Report. First Lake Henshaw Treatment. Prepared by AquaTechnex, Bellingham, Washington for Vista Irrigation District, Vista, California. June 2022.

Marine Biochemists. 2021. Aquatic Pesticide Application Plan for Lake Henshaw and the Warner Ranch. Prepared by Marine Biochemists, Anaheim, California for Vista Irrigation District, Vista, California.

Stillwater Sciences. 2022. Assessment of March 2022 algaecide treatment effectiveness for Lake Henshaw. Prepared by Stillwater Sciences, Berkeley, California for Vista Irrigation District, Vista, California. April 2022.

Appendices

Appendix A

In Situ Figures

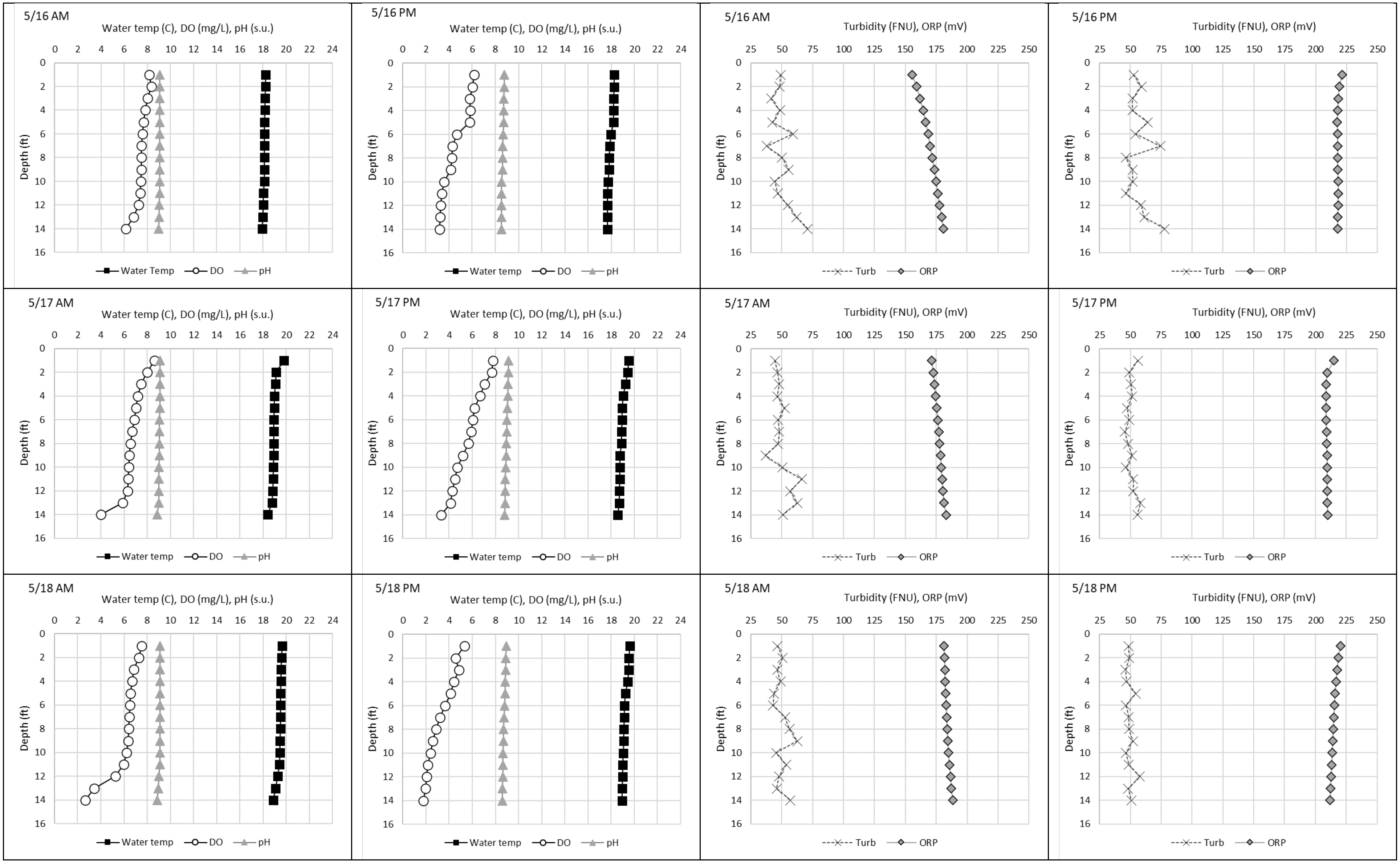


Figure A-1a. Buoy Line (H-BL) *in situ* water quality morning and afternoon results for dates indicated.

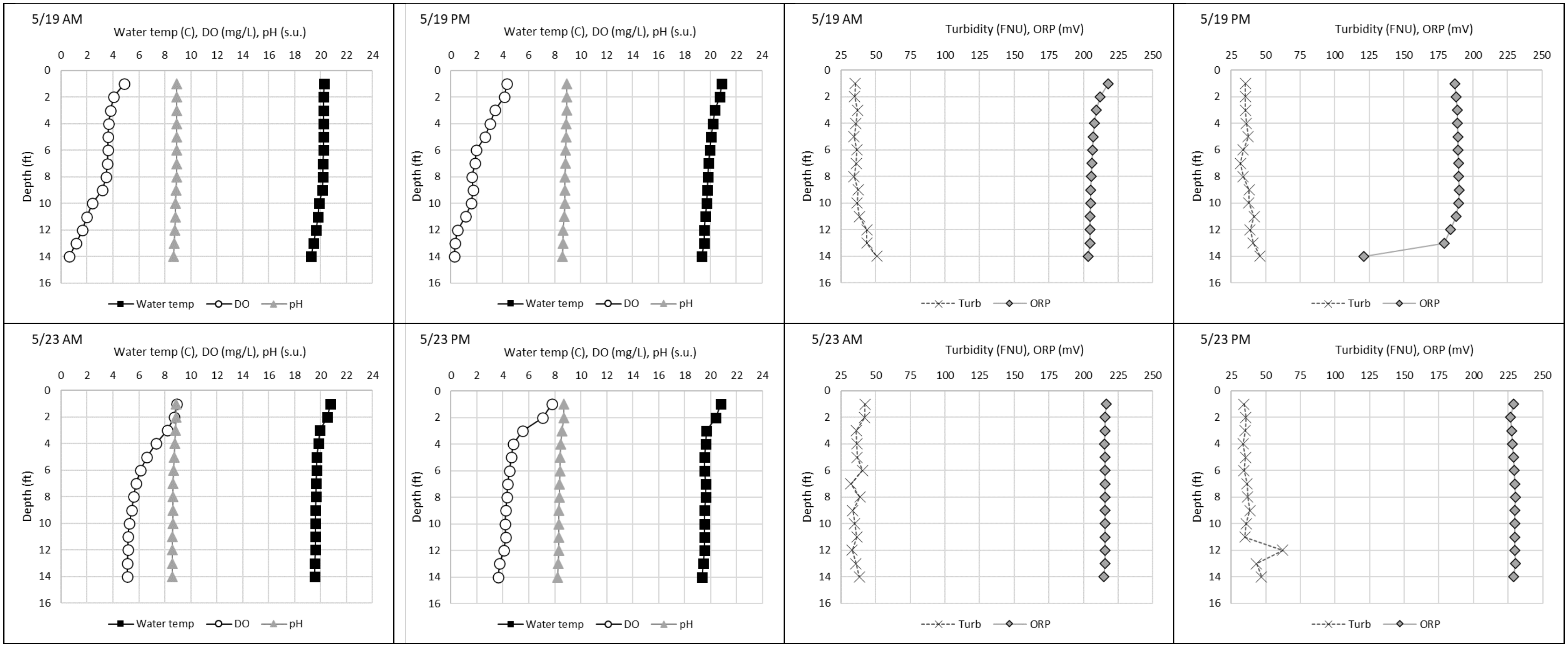


Figure A-1b. Buoy Line (H-BL) *in situ* water quality morning and afternoon results for dates indicated.

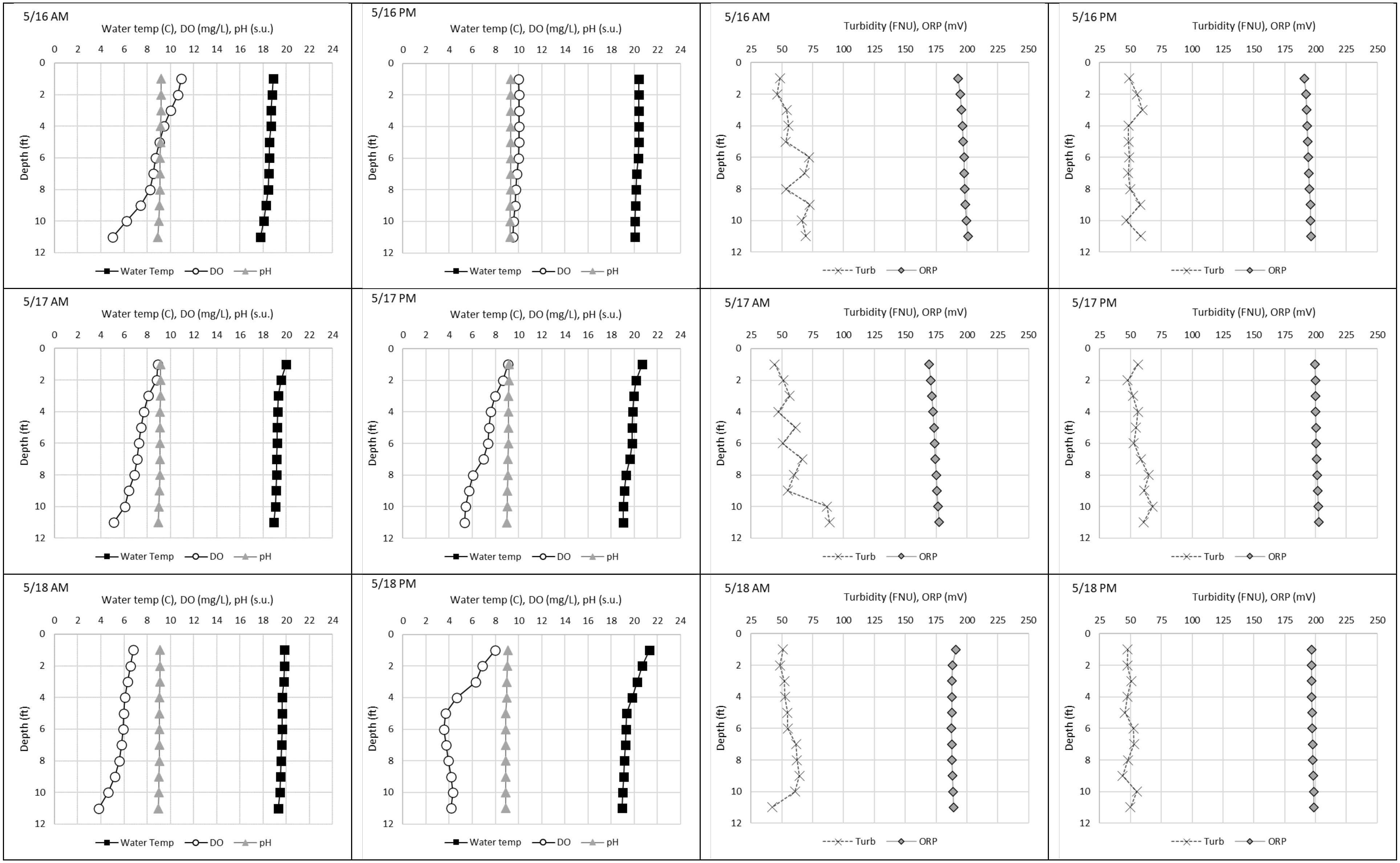


Figure A-2a. Fish Dock (H-FD) *in situ* water quality morning and afternoon results for dates indicated.

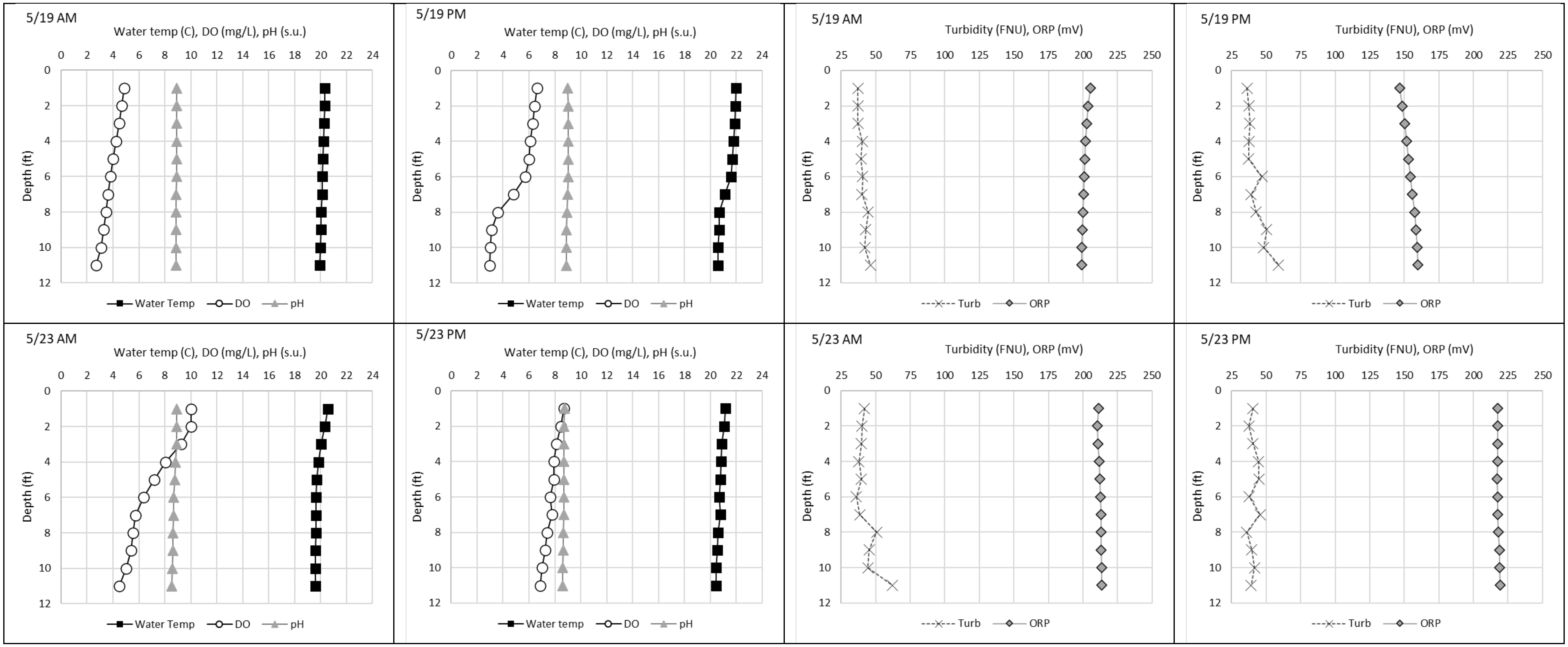


Figure A-2b. Fish Dock (H-FD) *in situ* water quality morning and afternoon results for dates indicated.

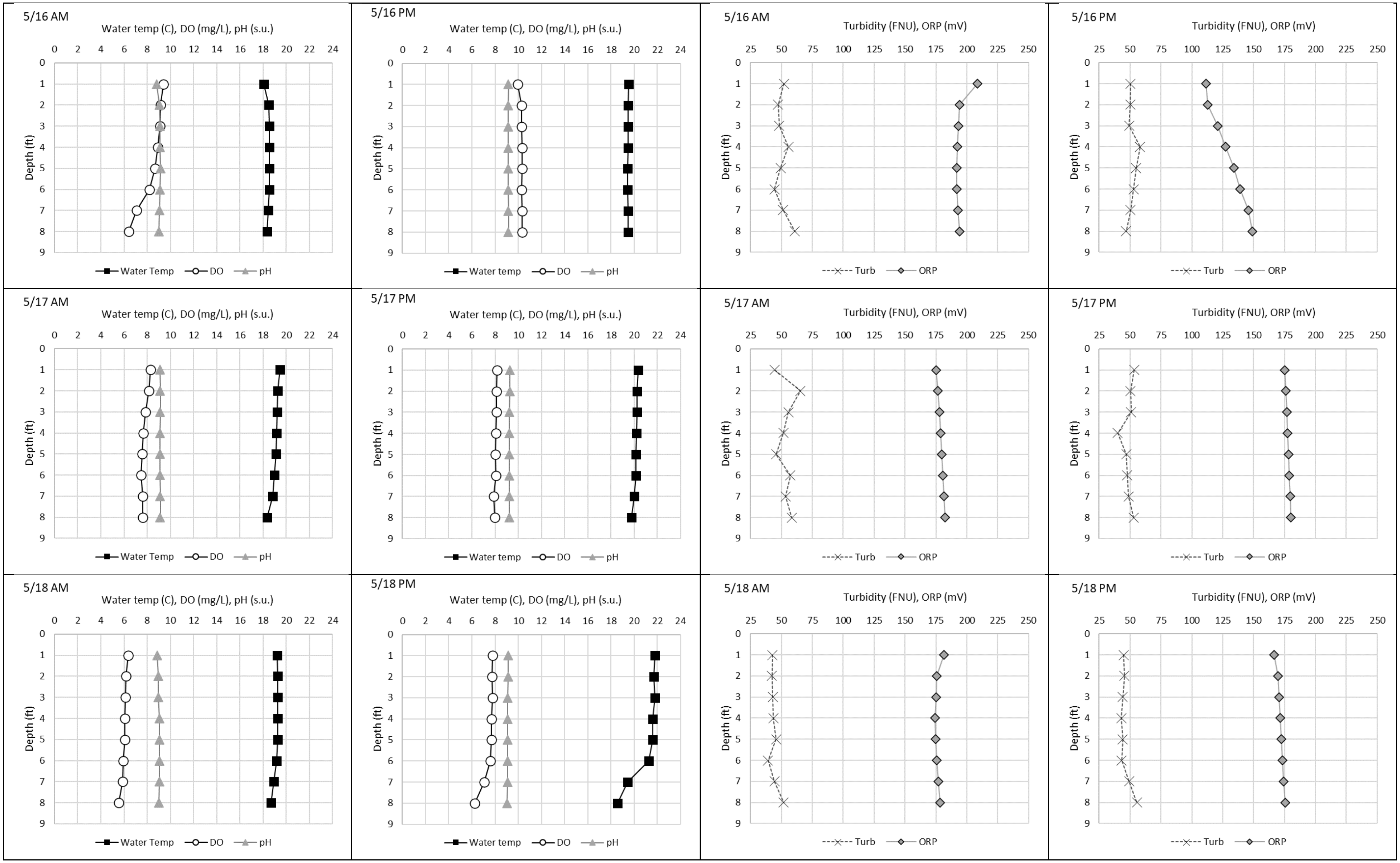


Figure A-3a. Middle Lake (H-ML) *in situ* water quality morning and afternoon results for dates indicated.

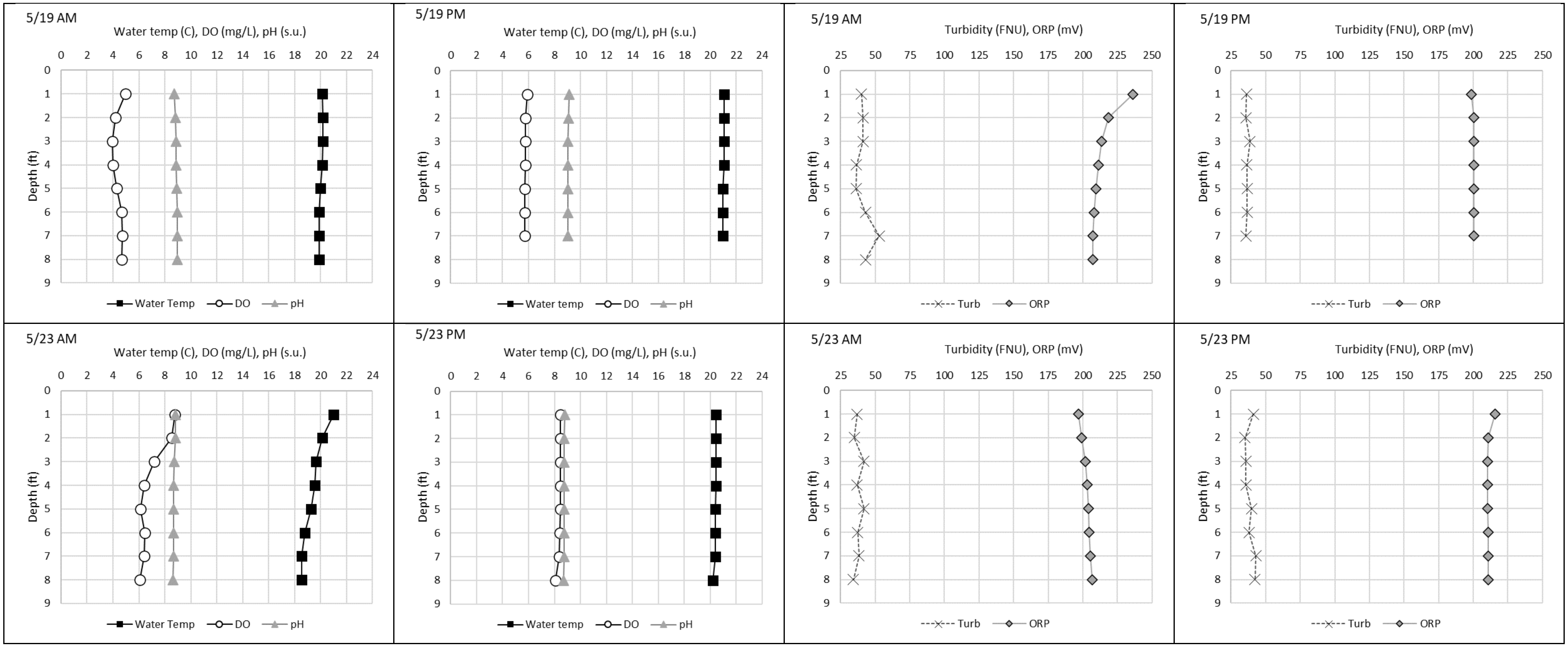


Figure A-3b. Middle Lake (H-ML) *in situ* water quality morning and afternoon results for dates indicated.

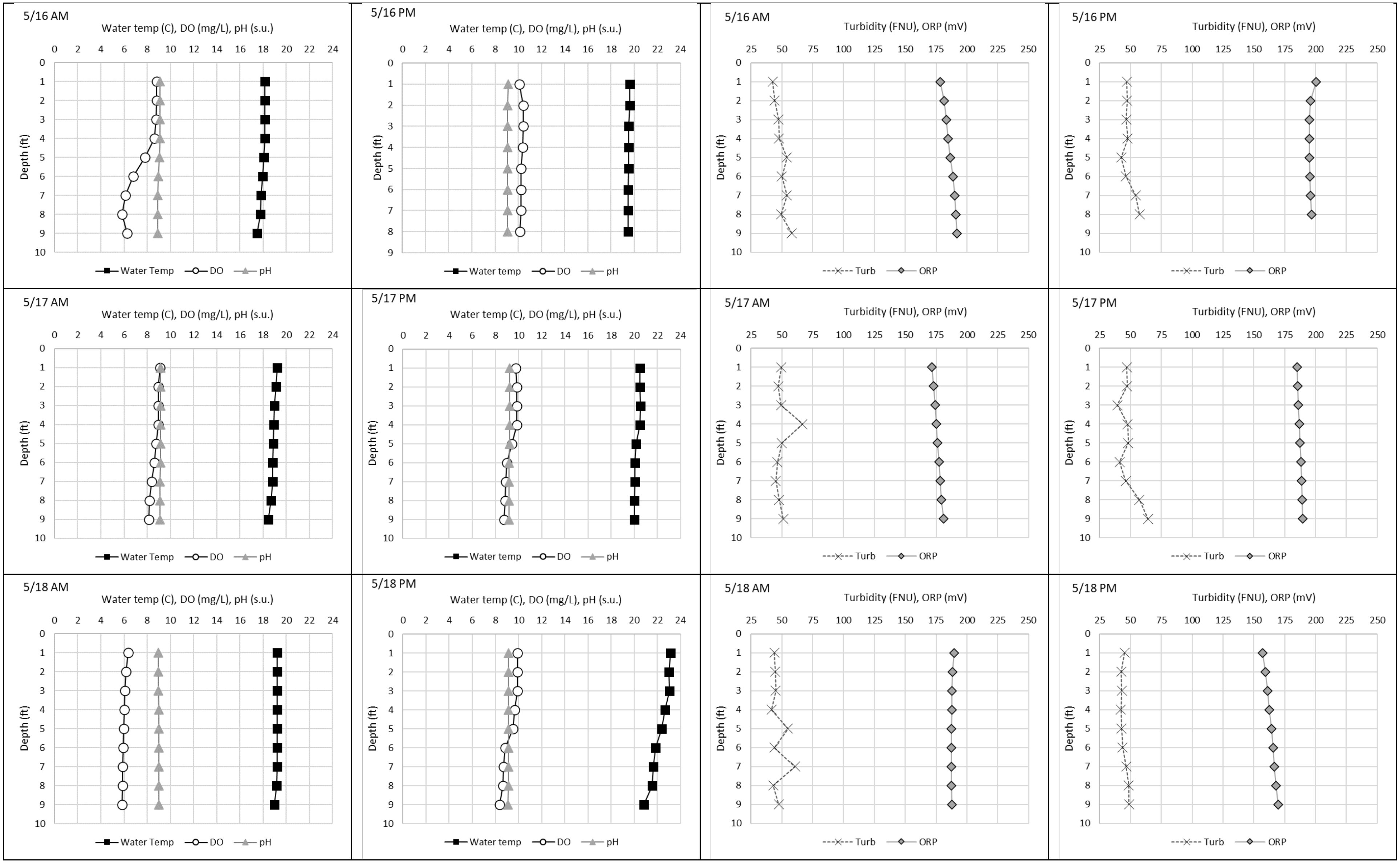


Figure A-4a. North Lake (H-NL) *in situ* water quality morning and afternoon results for dates indicated.

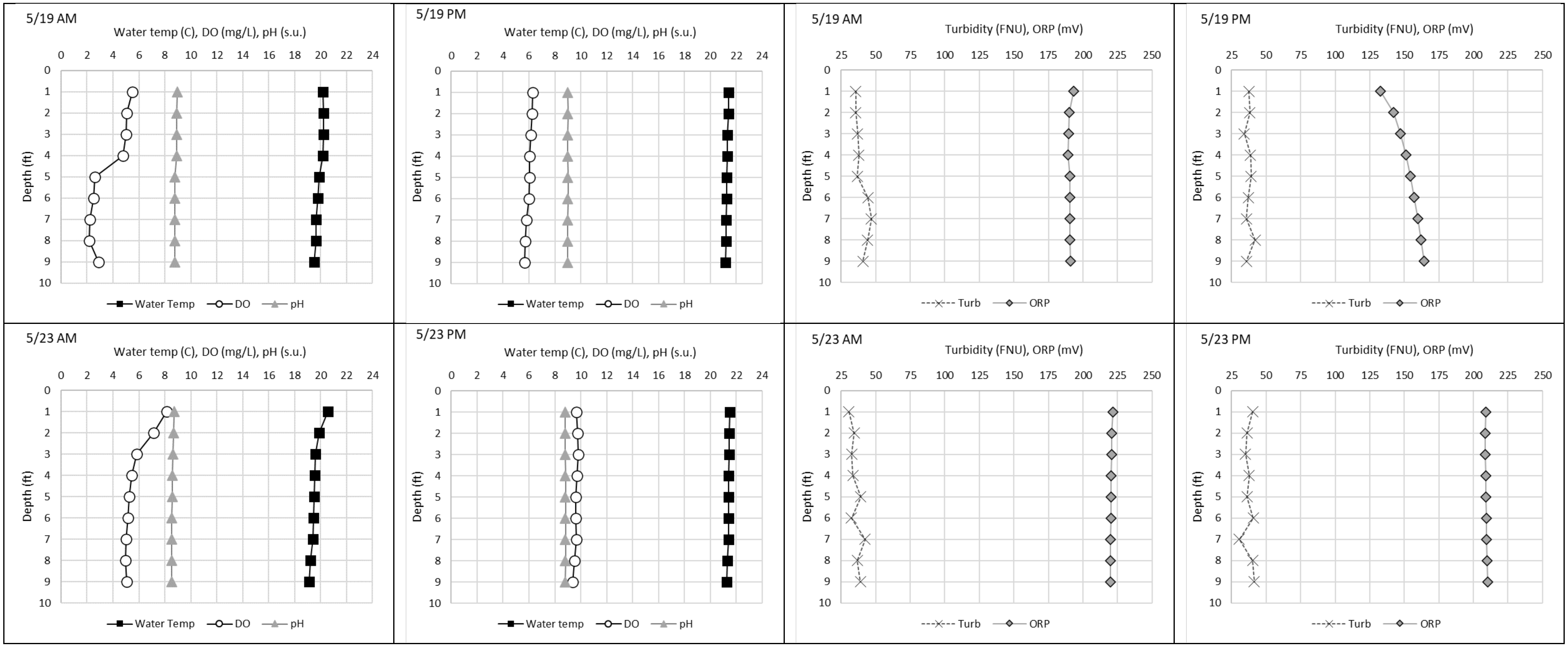


Figure A-4b. North Lake (H-NL) *in situ* water quality morning and afternoon results for dates indicated.

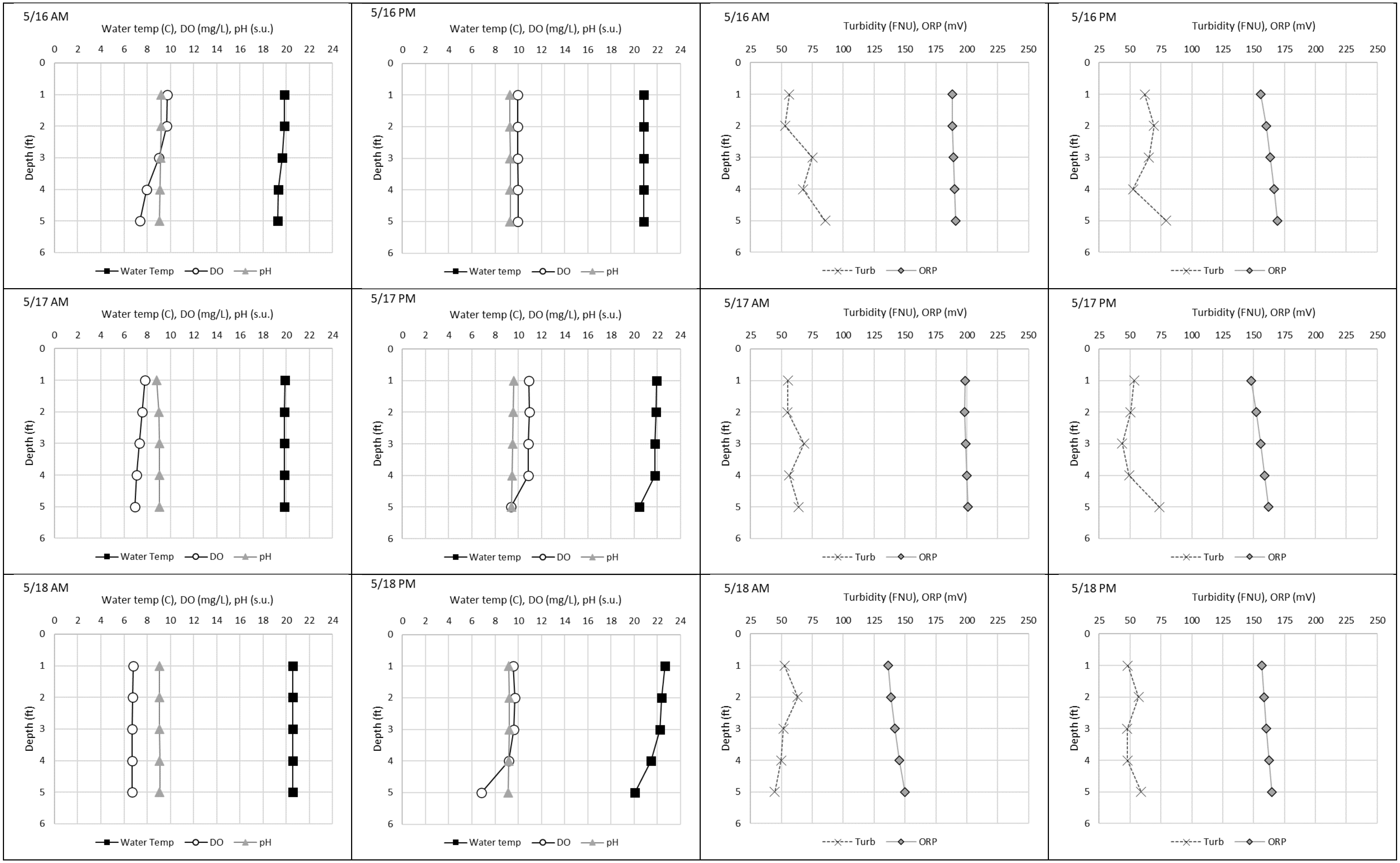


Figure A-5a. South Lake (H-SL) *in situ* water quality morning and afternoon results for dates indicated.

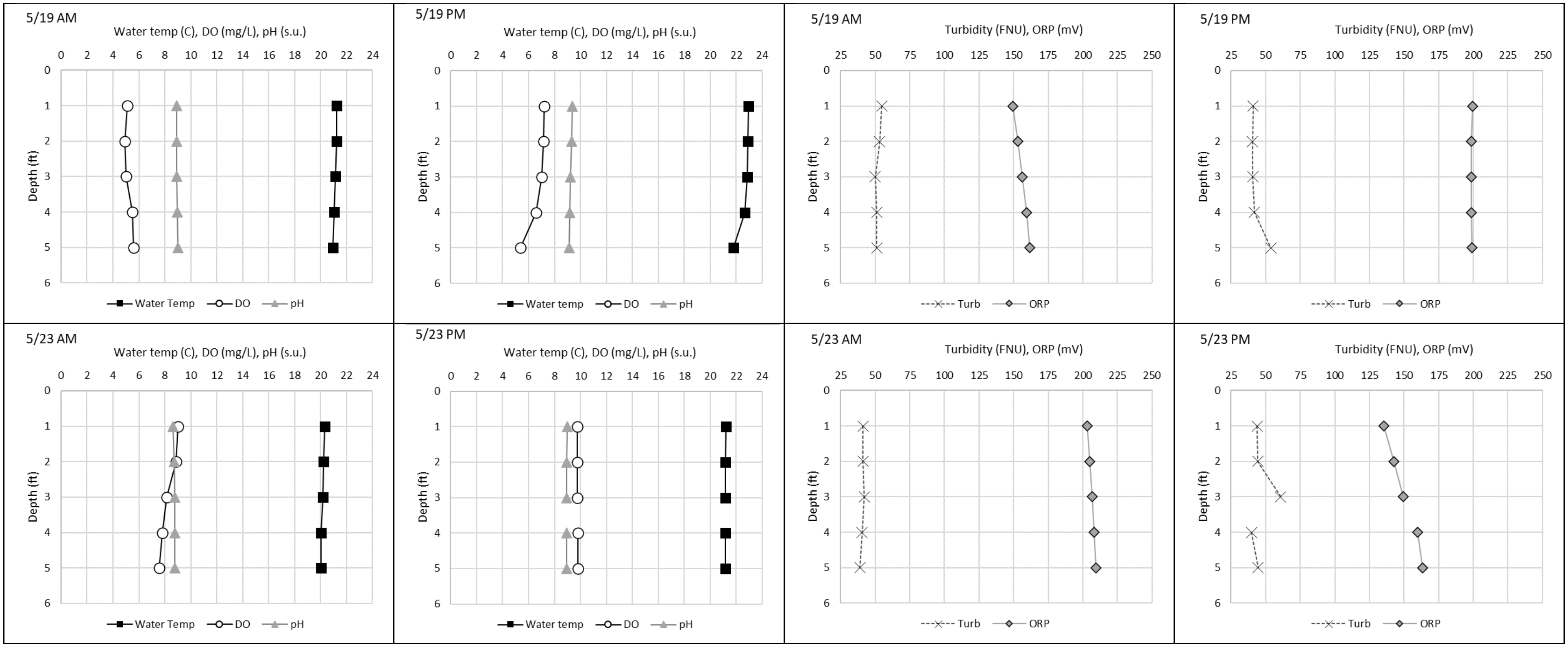


Figure A-5b. South Lake (H-SL) *in situ* water quality morning and afternoon results for dates indicated.