IMPACTS OF THE LAKE ELSINORE ADVANCED PUMPED-STORAGE (LEAPS) PROJECT ON WATER QUALITY IN LAKE ELSINORE

FINAL REPORT

Prepared for:

The Federal Energy Regulatory Commission
Washington, DC
Project No. 14227

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30 January 2019

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EXECUTIVE SUMMARY

This study evaluated the impacts on water quality in Lake Elsinore from (i) operation of the Lake Elsinore Advanced Pumped-Storage (LEAPS) project at different lake levels, and (ii) storage of water in the Upper Reservoir. The study was conducted in response to needs highlighted by the Santa Ana Regional Water Quality Control Board and identified in the *Response to Additional Study Requests* prepared by the Federal Energy Regulatory Commission (FERC) (FERC, 2018). The study used the 3-D hydrodynamic-water quality Aquatic Ecosystem Model (AEM3D) (Hodges and Dallimore, 2016), and builds upon previous studies addressing potential impacts of LEAPS operation on Lake Elsinore (Anderson, 2006a,b; Anderson, 2007a,b).

The model grid for Lake Elsinore was developed from the hydroacoustic bathymetric survey conducted in 2010 (Anderson, 2010) and revised to 1255 ft based upon satellite imagery at known lake surface elevations. Bathymetry for the Upper Reservoir was taken from design documents. A horizontal grid of 40 m x 40 m was selected to represent the lateral dimensions of Lake Elsinore and the Upper Reservoir following consideration of spatial resolution, and number and duration of simulations needed for the study. The vertical dimension across the domain that included both Lake Elsinore and the Upper Reservoir was represented by 66 layers that were 0.3 m in thickness for the uppermost 12 m (representing the approximate maximum depth of Lake Elsinore) that then smoothly grade to 2 m in thickness at 32.5 m depth and remain at 2 m thickness to a depth of 50.5 m for the vertical discretization of the Upper Reservoir. This results in a total of 240,004 cells in the computational domain at full pool. The operations of the axial flow pumps and diffused aeration system installed in the lake were simulated using the jet/pump and bubble plume destratification subroutines, respectively. The diffused aeration system included twelve (12) 2500 ft diffuser lines arranged radially (6 on each side of the lake), with 325 1-mm holes per line driven by a total of 4 compressors yielding 50 psi line pressure. A total of 20 axial flow pumps each with a 0.8 m radius impeller at 1.8 m depth generating 872 N of thrust per unit were arranged in platforms of 4 pumps/platform on adjacent horizontal cells approximating the high-speed zone perimeter buoy line. The diffused aeration system and axial flow pumps were typically operated 5-6 h each morning from late spring to early fall as derived from monthly operating data. The operation of the axial flow pumps and diffused aeration system were considered part of the native condition of the lake for the purposes of this assessment. Dynamic Dirichlet boundary conditions linking water withdrawal from Lake Elsinore and delivery to the Upper Reservoir (and vice versa) were implemented so that water and the associated properties (e.g., temperature, dissolved oxygen (DO), nutrient concentrations, algae and algal toxin levels) are transferred between the two water bodies during operation of LEAPS. A timestep of 24 sec was used to meet Courant-Friedrich-Lewy and Lipschitz constant conditions.

The model was calibrated to water column and water quality data available for the period from February 8, 2016 – August 31, 2018; this period was selected for calibration and analysis based upon the availability of high quality monitoring data and the extreme conditions present in the lake over this time. Conditions included extremely low lake levels (<1233 ft) in 2016 with very high total dissolved solids (TDS) concentrations (nearly 4,000 mg/L) and very poor water quality conditions (concentrations of total N as high as 9.8 mg/L, total P nearly 0.5 mg/L, and chlorophyll a reaching 349 µg/L), followed by high runoff and marked improvements in lake level and water quality in 2017, and the return of drought conditions in 2018. The model accurately reproduced observed lake levels, with root mean-square error (RMSE) of 2.4 inches, accurately predicted TDS concentrations and water column temperatures (relative RMSE values of 3.7-4.9%), and reasonably reproduced total N and total P concentrations (relative RMSE values of 12.4 and 18.3%, respectively). The model struggled somewhat in reliably predicting observed chlorophyll a and DO concentrations in this hyper-eutrophic lake across the wide ranging conditions of this time period (relative RMSE values of 31.5 and 47.0%, respectively). Notwithstanding the poorer fit to DO and chlorophyll a, the model was considered adequate for a comparative analysis of water quality with and without operation of LEAPS.

The model was then used to simulate the lake in its native condition (including operation of the axial flow pumps and diffused aeration system) and with LEAPS operation that included initial supplementation of up to 15,000 acre-feet (af) of State Water Project (SWP) water and annual inputs of approximately 300 af of SWP water to offset evaporative losses from the Upper Reservoir. The SWP water is of high quality, with only 10-18% of the total P concentrations and 20-50% of the total N concentrations of current water sources for the lake (San Jacinto River, local runoff and recycled water). Three initial lake surface elevations were evaluated (1235, 1240 and 1247 ft) using the meteorological and hydrologic conditions from February 2016 – August 2018.

The operation of LEAPS with supplementation up to 15,000 af of SWP water significantly increased the lake level and reduced volume-weighted concentrations of TDS, nutrients, chlorophyll a and microcystin across the range of lake elevations, with the greatest relative improvement achieved at lowest initial surface elevation and poorest water quality. The operation of LEAPS had minimal effect on temperature gradients in the lake, defined here by the difference in near-surface and near- bottom temperatures (ΔT values), and volume-averaged DO concentrations. Operation of LEAPS did modify to varying degrees vertical and horizontal distributions of DO. Two anticipated operational schedules for LEAPS (nighttime pumping/daytime hydropower generation or morning pumping/afternoon-evening hydropower generation) were evaluated, as well as two alternative widths of the lake inlet/outlet (I/O) (50 m vs 150 m); neither operation schedules nor I/O widths were found to significantly alter average water column conditions in Lake Elsinore. The operation of LEAPS across a lake surface elevation range of at least 1235-1253 ft is supported by model results.

Storage of water in the Upper Reservoir was found to have variable effects on water quality that increased with increasing retention time. With regular weekday pumping and hydropower generation, the retention time of water in the Upper Reservoir was 1-2 days, and water quality followed quite closely that of Lake Elsinore with very similar temperatures and concentrations of TDS, nutrients, chlorophyll a and microcystin. The concentrations of DO did vary somewhat however, with concentrations in the Upper Reservoir generally lower than volume-averaged concentrations in Lake Elsinore. Increasing the duration of storage in the Upper Reservoir from approximately 2 days over the weekend under a typical schedule to 1 or 2 weeks yielded more significant differences, especially with respect to concentrations of chlorophyll a and DO where substantial reductions were predicted as residence time increased. Volume-averaged concentrations of nutrients, DO and microcystin in Lake Elsinore were only very modestly changed following flow of this water during hydropower generation due to the small volume compared with that present in the lake.

In addition to assessing water quality impacts of LEAPS operation, the potential for enhancement of water quality relative to native conditions was also evaluated. While not proposed as part of the license application for the project, augmentation to DO concentrations in return flows to the lake during hydropower generation was predicted to provide significant additional ongoing benefits to water quality beyond initial dilution with SWP water. Simulations indicated that LEAPS operation with DO augmentation helped distribute DO across the lake, including directly above the sediments, which would reduce fish kills, favorably shift biogeochemical cycling of nutrients and improve overall ecological conditions. The capacity for LEAPS to improve water quality is thought to be of value to the TMDL efforts at the lake.

INTRODUCTION

Pumped-storage hydroelectric plants play important roles in load balancing of electric supply grids by providing electricity during periods of peak demand, storing renewable energy and controlling supply frequency. A pumped-storage hydroelectric plant has been considered for construction at Lake Elsinore for over 20 years. The Lake Elsinore Advanced Pumped-Storage (LEAPS) project takes advantage of the strong elevation gradient between the Santa Ana Mountains and Lake Elsinore and is centrally located to both the electrical supply grid for Southern California and to wind and solar renewable energy sources.

In response to the request for comments by the Federal Energy Regulatory Commission (FERC) concerning the current license application of Nevada Hydro for the Lake Elsinore Advanced Pumped Storage (LEAPS) Project (filed Oct. 2, 2017), numerous studies were requested by regional, state and federal resource agencies, municipalities and others. Following review, FERC issued a Response to Additional Study Requests on June 15, 2018 that directed Nevada Hydro to develop two study plans related to water quality (FERC, 2018). These studies - Studies #4 and 7 - address the need, identified by the Santa Ana Regional Water Quality Control Board (RWQCB), for additional information on the impacts of the LEAPS Project on water quality in Lake Elsinore. Study #4 specifically addresses the impact of pumping, transient storage in the Upper Reservoir, and hydropower generation on total nitrogen (N), total phosphorus (P) and cyanotoxin concentrations in return flows to Lake Elsinore. Study #7 addresses the effects of LEAPS operation at different lake surface elevations on water quality in Lake Elsinore and identification of lake elevations when significant negative impacts would occur. A Study Plan was developed that outlines the approach and describes information to be developed in the studies.

This report summarizes the results from these studies using integrated hydrodynamic-water quality simulations that assessed impacts of LEAPS operation on water quality at different lake surface elevations and the consequences of transient storage in the Upper Reservoir on water quality of return flows to Lake Elsinore. An additional analysis evaluated the effects of the operation of LEAPS on water quality and strategies to enhance water quality in Lake Elsinore relative to current conditions through design of LEAPS.

Background

The LEAPS Project consists of 3 primary components: (i) Lake Elsinore, which serves as the Lower Reservoir and pumped-water supply; (ii) an Upper Reservoir that provides transient storage of water used for hydropower generation; and (iii) the turbines/penstocks and related hydroelectric power infrastructure.

Lake Elsinore is a shallow, eutrophic lake in southwestern Riverside County that has varied dramatically in lake surface elevation over time, from intervals of complete desiccation in the late 1950's to early 1960's to episodes of extreme flooding. Water quality has also varied profoundly, from salinity levels exceeding sea water with very high nutrient concentrations at extremely low lake levels, to low total dissolved solids (TDS) and nutrient concentrations. Lake Elsinore was placed on the State of California's Clean Water Act Section 303(d) list in 1994 due to hypereutrophication and listed in 1998 as impaired due to excess nutrients, organic enrichment/low dissolved oxygen (DO) and sedimentation/siltation. A total maximum daily load (TMDL) was developed by the RWQCB and incorporated into the Basin Plan in 2004. Since that time, several lake restoration projects have been undertaken, including fishery management through removal of carp (*Cyprinus carpio*) and stocking of hybrid striped bass (*Morone saxatilis*); delivery of up to about 5,000 acre-feet per year of recycled water to supplement natural rainfall and runoff during periods of low lake level and drought; installation in 2004 of 20 axial flow pumps to enhance natural wind-forced and convective mixing processes; and installation in 2007 of a dual diffused aeration system with >20 km of diffuser lines driven by four 200 horsepower compressors. The TMDLs for Lake Elsinore are currently undergoing revision. The Draft TMDL Technical Report was released in December 2018 for public and peer review.

The Upper Reservoir, proposed for siting in Decker Canyon at an elevation of over 2600 ft above mean sea level (MSL), will have a maximum capacity of 7175 acre-feet (af), useable storage volume of approximately 6300 acre-feet, maximum surface area of 76 acres and maximum depth over 150 ft.

The final component of LEAPS involves the turbines, penstocks and related hydraulic and hydroelectric elements that hydraulically link the Upper Reservoir to

Lake Elsinore. Water will be pumped from and returned to Lake Elsinore through an inlet-outlet (I/O) structure sited on the western shore of the lake, for modeling purposes, at a base elevation of 1220 ft and gate elevation of 1223 ft. Simulations evaluated I/O widths of 50 m and 150 m with a height of 10 m.

A series of studies were conducted in 2006-2007 to determine the potential water quality impacts of the LEAPS project at the request of the Santa Ana Regional Water Quality Control Board as part of a previous FERC application process for Project No. 11858. The studies included review of published studies of pumped-storage hydroelectric plant operations, analytical model calculations of turbulent kinetic energy inputs, water column stability, and organism entrainment (Anderson, 2006a), heat budgets for Lake Elsinore and the Upper Reservoir (Anderson, 2006b), 3-D numerical simulations of pumped-storage operation and effects on thermal stratification, sediment resuspension and organism entrainment using the Environmental Fluid Dynamics Code (EFDC) (Anderson, 2007a), and modeling of ecological impacts and trophic cascades using a simplified linear food chain model (Anderson, 2007b).

In addition to modeling studies conducted using EFDC (Anderson, 2006a,b; Anderson, 2007a,b), Lake Elsinore has been evaluated using the 1-D Dynamic Reservoir Simulation Model (DYRESM)- Computational Aquatic Ecosystem Dynamics Model (CAEDYM) model (e.g., Anderson, 2015a,b,c), including simulations in support of the TMDL revision for the lake (CDM-Smith, 2018). These simulations focused on long-term representations of lake level, TDS and water quality in Lake Elsinore over the period 1916-2016 that highlighted the tremendous variability present. The 1-D approximation allowed simulations over the decadal to century time-scales, although it does not capture the spatially and temporally complex hydrodynamic processes and potential water quality impacts resulting from operation of LEAPS.

As noted above, prior analyses of the LEAPS project included analytical model calculations and numerical 3-D hydrodynamic simulations. Since these studies were conducted, more sophisticated water quality and aquatic ecology models have been developed and linked to 3-D hydrodynamic models that allow comprehensive representation of the physics, chemistry and biology of lakes and reservoirs (e.g., Hodges and Dallimore, 2014; Hipsey, 2014). In addition, substantial improvements have been achieved in computational power over the past decade, allowing solutions for systems at fine temporal and spatial scales over seasonal and multi-year timescales (e.g., Preston et al., 2014a). Moreover, 3-D hydrodynamic models are increasingly used to optimize design and operation of hydraulic systems (Preston et al., 2014b), including reservoir water quality management (Anderson et al., 2014) and for compliance with regulatory requirements (e.g., SBDDW-16-02). A coupled 3-D hydrodynamic-water quality model dynamically solving heat, water and nutrient budget equations, transport equations, water quality and aquatic ecology was developed to address existing gaps in understanding about LEAPS and its impacts on Lake Elsinore.

<u>Objectives</u>

The objectives for this study are to:

i) Quantify effects of LEAPS operation at different lake surface elevations on water quality in Lake Elsinore and identification of lake elevations when significant negative impacts would occur;

ii) Assess impacts of pumping, transient storage in the Upper Reservoir, and hydropower generation on concentrations of total N, total P, cyanotoxins, DO and other constituents in return flows to Lake Elsinore during operation of LEAPS;

iii) Evaluate two anticipated operational modes of LEAPS and evaluate design strategies to *enhance* water quality in Lake Elsinore when compared with native conditions.

APPROACH

A 3-D hydrodynamic-water quality model using the Aquatic Ecosystem Model (AEM3D) was developed for Lake Elsinore and the Upper Reservoir. The model numerically simulated initial filling of the Upper Reservoir and the daily, seasonal and multi-year operation of LEAPS. The model includes rainfall, runoff, and water supplementation and the relevant physical, chemical and biological processes affecting water quality. The AEM3D model is based upon, and includes enhancements to, the Estuary Lake and Coastal Ocean Model (ELCOM)-Computational Aquatic Ecosystem Dynamics Model (CAEDYM) (Hodges and Dallimore, 2016). Dynamic Dirichlet boundary conditions linking water withdrawal from Lake Elsinore and delivery to the Upper Reservoir (and vice versa) were implemented so that water and the associated properties (e.g., temperature, DO, nutrient concentrations, algae and algal toxin levels) are transferred between the two water bodies during operation of LEAPS.

The grid for the Lake Elsinore model was developed from the hydroacoustic bathymetric survey conducted in 2010 (Anderson, 2010) and revised to 1255 ft based upon satellite imagery at known lake surface elevations. Bathymetry for the Upper Reservoir was taken from design documents. A horizontal grid of 40 m x 40 m was selected to represent the lateral dimensions of Lake Elsinore and the Upper Reservoir following consideration of spatial resolution, and number and duration of simulations needed for the study (Fig. 1).

The vertical dimension across the domain that included both Lake Elsinore and the Upper Reservoir was represented by 66 layers that were 0.3 m in thickness for the uppermost 12 m (representing the approximate maximum depth of Lake Elsinore) that then smoothly grade to 2 m in thickness at 32.5 m depth and remain at 2 m thickness to a depth of 50.5 m for the vertical discretization of the Upper Reservoir. This results in a total of 240,004 cells in the computational domain at full pool. A timestep of 24 sec was used to meet Courant-Friedrich-Lewy and Lipschitz constant conditions during operation of LEAPS.

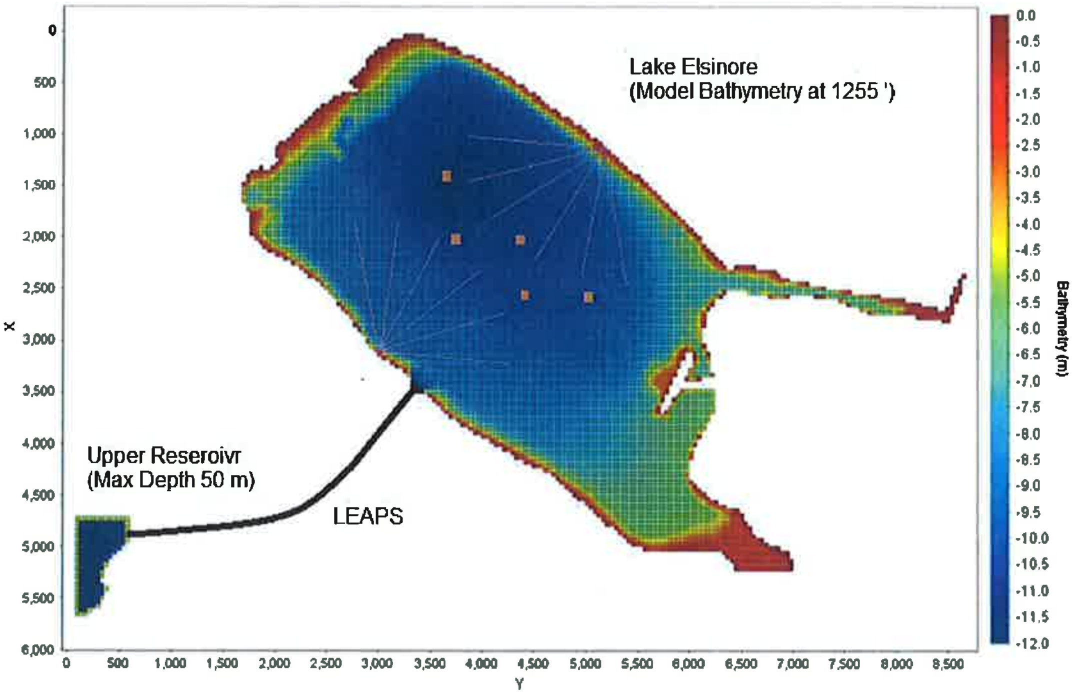

Fig. 1. Model grid and bathymetry (40 m x 40 m x 0.3 m; 240,004 cells in computational domain at full pool). Axial flow pumps depicted as brown squares, approximate location of diffused aeration lines shown as light purple lines.

The operations of the axial flow pumps and diffused aeration system were simulated using the jet/pump and bubble plume destratification subroutines, respectively. The diffused aeration system included twelve (12) 2500 ft diffuser lines arranged radially (6 on each side of the lake, Fig. 1), with 325 1-mm holes per line driven by a total of 4 compressors yielding 50 psi line pressure. A total of 20 axial flow pumps each with a 0.8 m radius impeller at 1.8 m depth generating 872 N of thrust per unit were arranged in platforms of 4 pumps/platform on adjacent horizontal cells approximating the high-speed zone perimeter buoy line (Fig. 1). The diffused aeration system and axial flow pumps were typically operated 5-6 h each morning from late spring to early fall (derived from monthly operating data). The operation of the axial flow pumps and diffused aeration system were considered part of the native condition of the lake for the purposes of this assessment.

Calibration

The period from February 8, 2016 – August 31, 2018 was selected for calibration and analysis due to two primary factors: (i) the availability of high quality monitoring data, including cyanotoxin concentrations, over this period, and (ii) the extreme conditions present in the lake. Conditions included 2016, which had the lowest lake level and poorest water quality in the lake in several decades (thus representing a sort of worst case condition), 2017 that included high runoff inputs that resulted in large increase in lake level and marked decrease in salinity, and 2018, which was representative of typical drought conditions for the region.

Meteorological data (hourly air temperature, relative humidity, atmospheric pressure, cloud cover, shortwave radiation, windspeed, wind direction and rainfall were taken from nearby meteorological stations (e.g., NOAA #1275, KAJO, ECSC1, CNAC1 and California Irrigation Management Information System (CIMIS) station #057 (Table 1, Fig. 2).

Table 1. Summary of key hourly meteorological data over simulation period (2/8/2016 – 8/31/2018)

	Solar Rad (W/m^2)	Air Temp (°C)	Rel Humidity (%)	Wind Speed (m/s)	Wind Dir (°)
Mean	218	22.2	50	1.7	214
Median	16	21.3	50	1.4	245
95%	829	35.2	83	4.2	322
5%	0	11.9	14	0.1	105

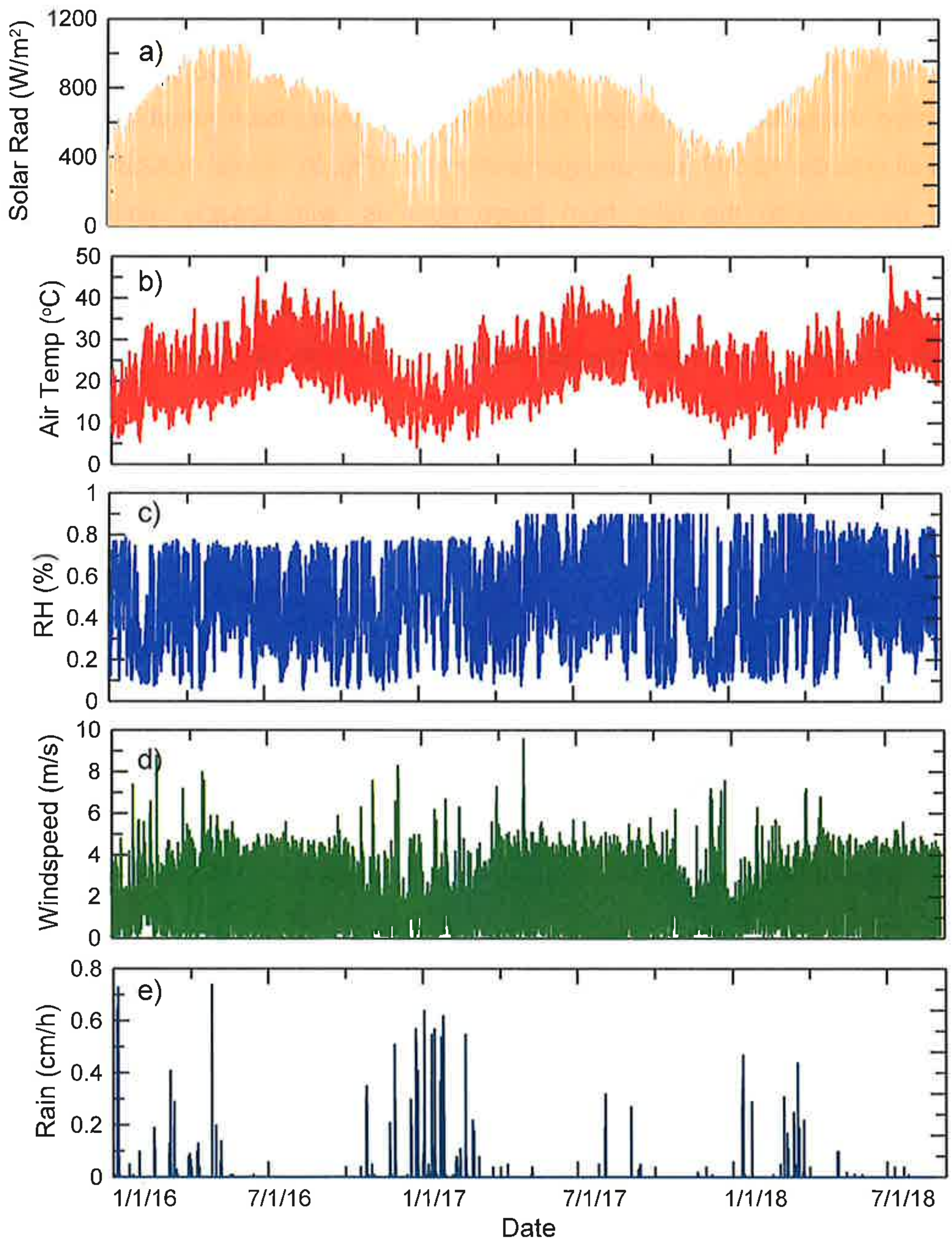

Fig. 2. Hourly meteorological data used in model simulations: a) solar radiation, b) air temperature, c) relatively humidity, d) windspeed and e) rainfall.

Hydrologic inputs for the model were taken from daily flow records for the San Jacinto River into Lake Elsinore at USGS gage #11070500, recycled water flows from Elsinore Valley Municipal Water District (EVMWD) and local runoff estimated from rainfall onto the 13,340 acre ungaged watershed (Fig. 3). A total of 42,808 af of water was delivered to the lake from these sources, with broadly similar individual contributions (Table 2). While recycled water was added consistently at an average flow near 15 af/d, 80% of the watershed contributions and more than 50% of total water delivered to the lake occurred between December 15, 2016 – March 15, 2017 (Fig. 3).

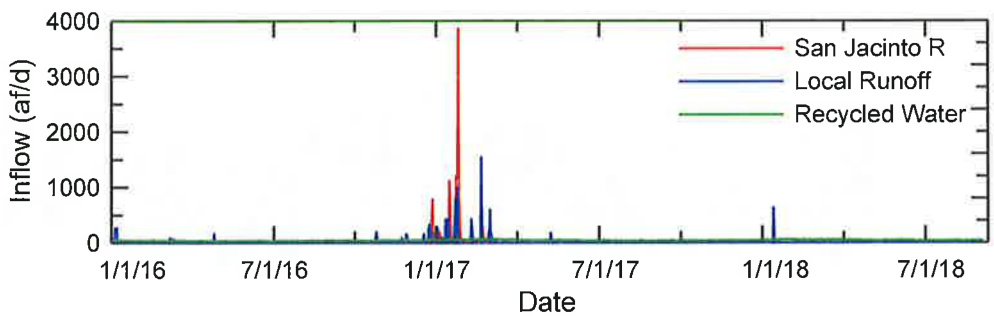

Fig. 3. Inflows to Lake Elsinore during simulation period (recycled water was rescaled by 3x to be visible on figure).

Table 2. Cumulative inflow (af) to Lake Elsinore (Feb.8, 2016 – Aug 31, 2018).

San Jacinto River	Local Runoff	Recycled Water	Total
16,195 (39.4%)	11,061 (26.9%)	13,881 (33.7%)	42,808

The concentrations of nutrients and TDS in these waters were taken from available monitoring data provided by Foster Amec Wheeler and EVMWD (Table 3). Included in Table 3 is the average water quality in State Water Project (SWP) water (discussed later).

Table 3. Volume-weighted concentrations (mg/L) of inflows (Feb. 8, 2016 – Aug. 31, 2018)

Source	**TDS**	**Total N**	**Total P**	**PO_4-P**	**NH_4-N**	**NO_3-N**
SJR	213	1.86	0.39	0.20	0.09	0.79
Local Runoff	120	1.82	0.48	0.20	0.22	0.80
Recycled Water	699	4.83	0.70	0.57	1.38	3.02
State Water Project	254	0.93	0.07	0.06	0.03	0.53

Initial conditions for the simulation were set at measured water column conditions on February 8, 2016 from sampling station E2 near the center of the lake (Amec Foster Wheeler, 2017). Model parameterization from the AEM3D "Round Lake" example (Hodges and Dallimore, 2016) and recent DYRESM-CAEDYM simulations for Lake Elsinore were used as a starting point in AEM3D for representation of nutrient cycling, algal production, algal toxin levels, DO dynamics and other processes in the lake. Key model parameters are provided in the Appendix. Phytoplankton were simulated using a 2-phytoplankton class model representing blue-green algae that dominate through most of the year and diatoms that are present at generally lower levels during the winter and early spring (green algae and other groups typically comprise minor components of the phytoplankton assemblage in the lake) (Anderson et al., 2010). Simulation results were compared with water column and water quality data collected on 20 monitoring days over the period Feb. 8 2016 – Aug 31, 2018 (Amec Foster Wheeler, 2017; Amec Foster Wheeler, unpubl. data). Predicted lake surface elevations were compared with levels measured daily to weekly by EVMWD.

Calibration Results

<u>Lake level</u>

The model accurately predicted lake level over the February 2016 – August 2018 period (Fig. 4). It very accurately captured the reduction in lake surface elevation from 1235.2 ft in February 2016 to 1232.1 ft in October-November 2016 resulting from evaporation, the dramatic increase in elevation in January-February 2017 to 1240.5 ft, and the subsequent protracted decline in 2017-18.The root-mean square error (RMSE) over the entire simulation was 2.4 inches.

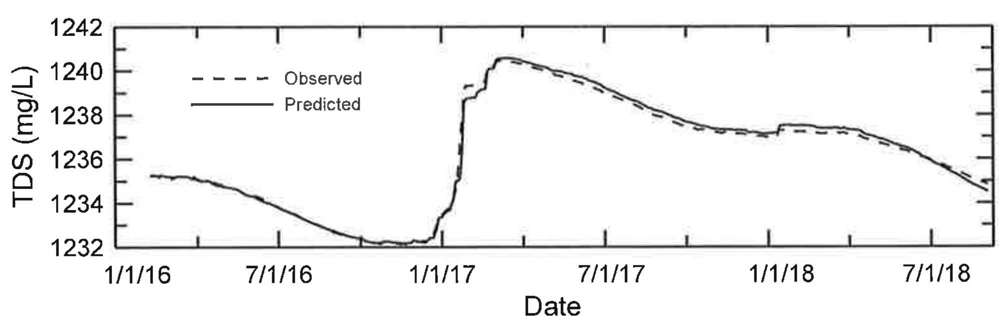

Fig. 4. Predicted and observed lake surface elevations over time.

Total Dissolved Solids

The model accurately reproduced TDS concentrations during the simulation period, including the rapid increase in TDS in 2016 due to evapoconcentration and the pronounced decrease in TDS due to runoff-dilution in the winter of 2017 (Fig. 5). The RMSE was 102 mg/L (relative RMSE = 3.7%).

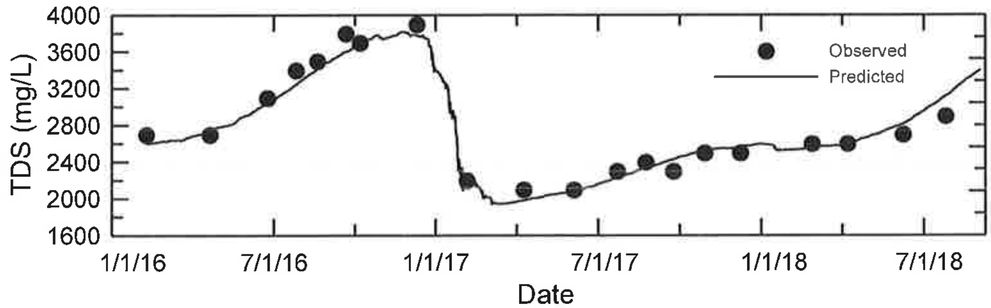

Fig. 5. Predicted and observed TDS concentrations over time.

Temperature Profiles

Temperature profiles were adequately reproduced with errors typically <0.8 °C (e.g., Fig. 6), with warming through summer, relatively rapid cooling in the fall, and modest temperature gradients with depth (RMSE= 0.99°C, relative RMSE=4.9%).

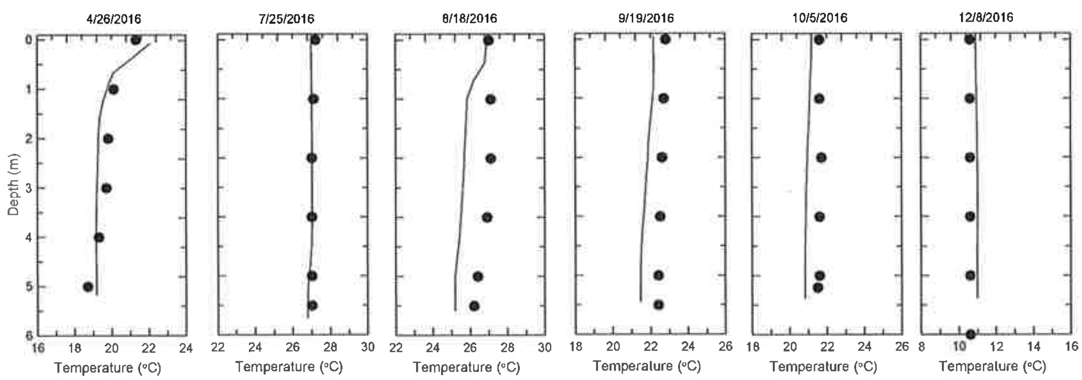

Fig. 6. Predicted and observed temperature profiles (2016) (symbols = observed values; lines = predicted values).

Dissolved Oxygen

Reported DO levels were routinely well-below saturation values (e.g., Fig. 7, dashed lines) indicating very high oxygen demand in lake. The model reproduced general trends on most dates although errors were often quite large, especially on days such as July 25, 2016 in which the model substantially over-predicted concentrations when strongly anoxic conditions were present throughout water column (Fig. 7) (RMSE=2.4 mg/L, relative RMSE=47%).

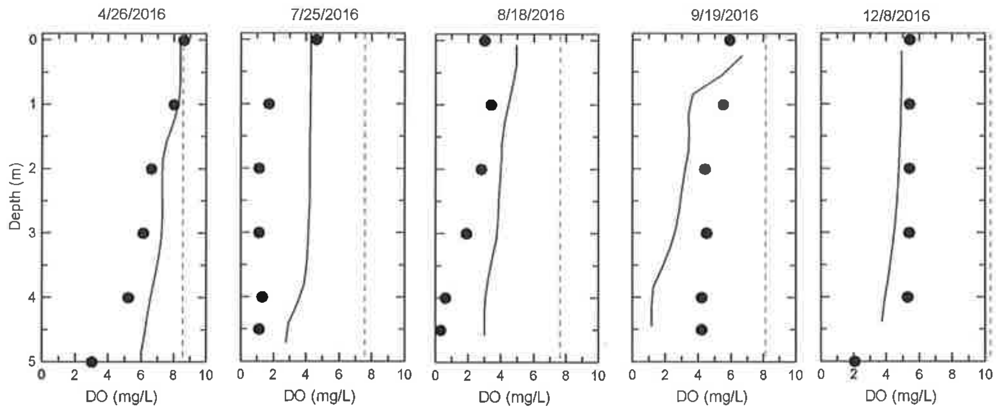

Fig. 7. Predicted and observed DO profiles (2016) (symbols = observed values; lines = predicted values; red dashed line = saturation values).

Total Nitrogen

The model reasonably reproduced observed total N levels over 2016-218 period, with very high total N concentrations in 2016, marked concentration decline with winter runoff in 2017, and gradual increase in 2018 (Fig. 8). The model did not capture the apparent short-lived near-doubling of TN concentration in late summer 2017, however. The RMSE was 0.74 mg/L (relative RMSE value of 12.4%) when excluding the 8/21/2017 data point (and an RMSE of 1.05 mg/L and relative RMSE of 17.6% when included).

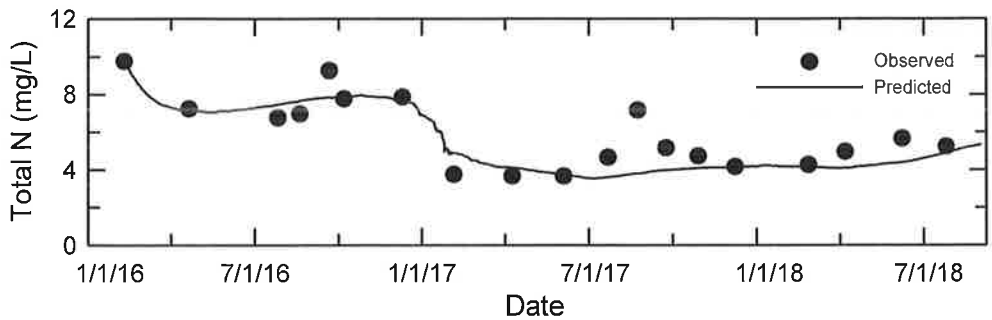

Fig. 8. Predicted and observed total N concentrations over time.

Total Phosphorus

The model reasonably reproduced total P concentrations over time, including the brief increase in concentration in early 2017 to 0.43 mg/L, subsequent decrease in spring 2017 and gradual increase in 2018 (Fig. 9). (A concentration of 0.82 mg/L was reported for 9/19/2016, although the mechanism by which total P would rapidly increase by about 2x and then quickly decrease is not clear; a Grubbs test identified this value was an outlier ($p<0.05$), so it was not included in the figure or in error calculations.) The RMSE for total P was 0.052 mg/L (relative RMSE = 18.3%). Notably, the observed marked decrease in total P concentration in winter-spring 2017, despite high inflow concentrations, indicate total P removal processes operating within the lake associated with settling of particulate P, sorption and uptake.

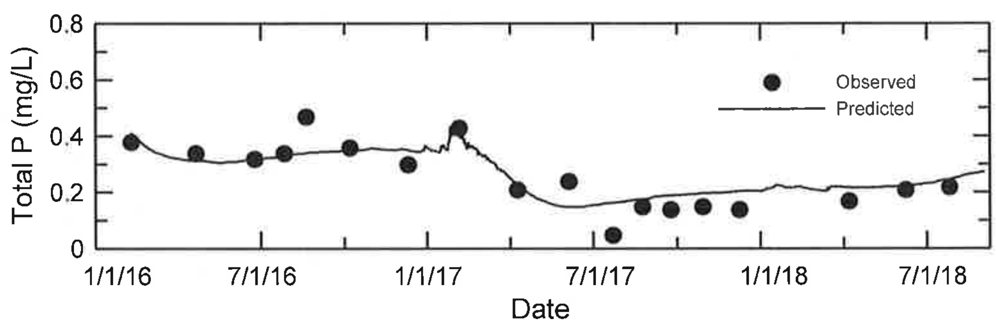

Fig. 9. Predicted and observed total P concentrations over time.

Chlorophyll a

Chlorophyll a levels varied strongly since 2016, with very high concentrations during much of 2016, followed by a sharp reduction in late 2016-early 2017 and with winter runoff. The levels moderated somewhat in 2017-18 but were routinely >100 µg/L and exceeded 200 µg/L in late fall 2017. The model predicted these general trends on most dates, but overall goodness of fit was poorer than most other water quality parameters (Fig. 10). For example, the model did not capture the marked decline in concentration in late winter-early spring 2016 and failed to predict the late fall 2017 bloom (Fig. 10), although the complexity of phytoplankton dynamics, especially in a highly variable and typically hypereutrophic lake, makes it difficult to predict algal blooms. The RMSE was 60 µg/L (relative RMSE = 31.5%).

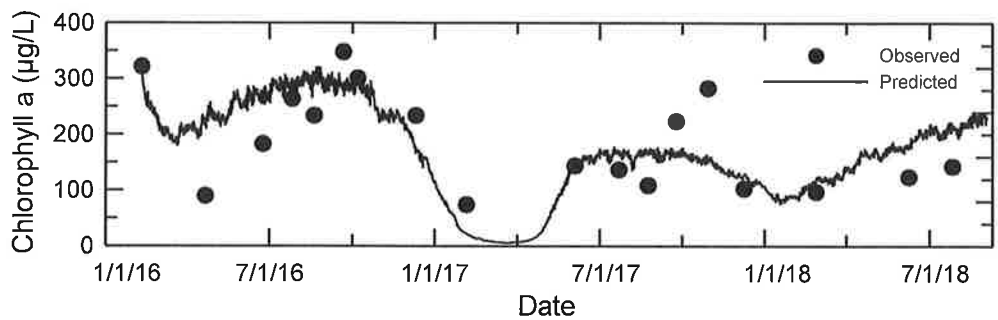

Fig. 10. Predicted and observed chlorophyll a concentrations over time.

<u>Algal Toxins</u>

AEM3D simulates algal toxin concentrations through internal toxin concentrations within algae that are considered to be a linear function of their growth rate (Long et al., 2001) and release to water through cell lysis and excretion modulated by bacterially-mediated decay (Hodges and Dallimore, 2016; Hipsey et al., 2014). The model failed to reproduce trends in algal toxin concentrations in Lake Elsinore (data not shown); monitoring data indicate that, with the exception of a single measurement point, the concentrations of the primary cyanotoxin (microcystin) were correlated with chlorophyll a levels (Buckley et al., 2018) (Fig. 11).

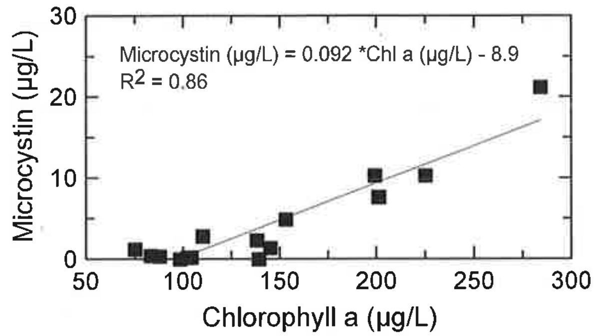

Fig. 11. Microcystin concentration vs total chlorophyll a concentration (excludes surface sample collected on 7/20/2017 with reported microcystin concentration of 67.4 µg/L and chlorophyll a concentration of 104 µg/L).

Use of the regression equation yielded better qualitative agreement than model predictions and were used to estimate microcystin concentrations from predicted chlorophyll a concentrations (Fig. 12). RMSE was 3.1 µg/L across this limited dataset.

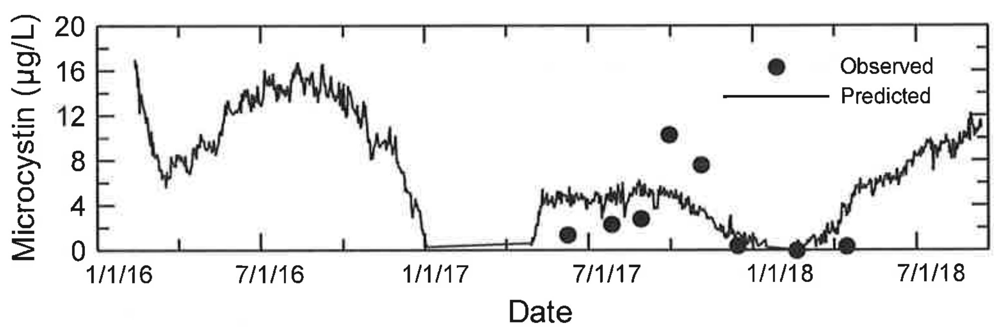

Fig. 12.Predicted and observed microcystin concentration over time.

ANALYSIS OF WATER QUALITY IMPACTS OF LEAPS OPERATION

It is recognized that the construction and operation of LEAPS will change the nature of Lake Elsinore in some ways. The initial filling of the Upper Reservoir will transfer water from Lake Elsinore and thus lower the lake surface elevation. The operation of LEAPS, with pumping of water to the Upper Reservoir and subsequent return of water during hydropower generation, will result in regular incremental oscillations in lake surface elevation. The Upper Reservoir also provides an additional approximately 70 acres of surface area from which water will evaporate, altering somewhat the water balance. Hydropower generation will also add turbulent kinetic energy to the water column, although the prior studies indicated that a very wide intake structure (150 m x 10 m) minimized TKE inputs, kept velocities near the I/O low (<5 cm/s), and limited resuspension of bottom sediments (Anderson, 2007; Anderson, 2010).

To address the issues related to initial filling of the Upper Reservoir and to help meet the chronic water supply challenge in Lake Elsinore, 15,000 af of State Water Project (SWP) water will be imported and delivered to the lake. Approximately 6,500 af will be used to support the initial filling of the Upper Reservoir and 8,500 af will increase the surface elevation of Lake Elsinore. This supplemental SWP water is of high quality, with an average total N concentration that is 20-50% and total P concentration that is only 15-20% of other sources (Table 3). Supplementation with SWP water will thus increase surface elevation, reduce salinity and lower concentrations of dissolved and total nutrients. The precise timing and duration of supplementation will be dependent upon conditions present at the time of startup; for the simulations described herein, the SWP water was delivered via the San Jacinto River channel with routing through the small (approximately 400 acre) upstream Canyon Lake at a rate of 250 cfs for 30 days over the period Feb. 9 – Mar. 9, 2016. The water quality was assumed to not change substantially during delivery and a preliminary simulation with the 3-D Canyon Lake model that assumed the lake was near full-pool indicated that steady-state water quality similar to SWP influent concentrations was reached within about 3-4 days.

It is recognized that the increased elevation at the Upper Reservoir will yield slightly lower local air temperatures related to the environmental lapse rate as well as a slightly higher relative humidities. Based upon the difference in elevation between

Lake Elsinore (approximately 1240 ft) and the Upper Reservoir (approximately 2800 ft), a reduction in air temperature of 3.1 °C from values used for Lake Elsinore was used for the Upper Reservoir. The lower air temperature would hold a bit less water vapor and have a slightly higher relative humidity; the relative humidity was increased by about 5% (typically 3-8% depending upon air temperature and relative humidity at Lake Elsinore). Weather stations at approximately 2500 to 3080 ft above MSL near the Upper Reservoir site (e.g., National Weather Service's ECSC1 station at El Cariso Village) confirm the general trend of slightly cooler air temperatures and correspondingly slightly higher relative humidities. Substantial differences in wind-speed and wind direction were not evident, so the values for Lake Elsinore were applied across the computational domain.

While the precise pumping-generation schedule will be established by the California Independent System Operator (CAISO) and dependent upon the electric power supply, demand, grid voltage and other factors, LEAPS is expected to operate regularly throughout the year. For this analysis, two weekly pump-generation schedules were considered: (a) a nighttime-pumping/daytime-generation cycle during the work week (Fig. 13a), and (b) a schedule which maximizes use of early-to-mid-day renewable energy production for pumping and late afternoon and evening hydropower generation (Fig. 13b). The nighttime-pumping/daytime-generation schedule was evaluated at each lake elevation scenario, while the maximum renewable schedule (Fig. 13b) was evaluated at a nominal 1240 ft lake level scenario. The 50 m wide I/O was used for simulations unless otherwise noted.

Three lake elevation scenarios were developed to assess impact of LEAPS on water quality in Lake Elsinore with initial elevations of:

1. 1235 ft (extremely low lake level) (represented by 2016-2018 conditions)
2. 1240 ft (moderate lake level)
3. 1247 ft (high lake level)

For each of these scenarios, the meteorological and hydrological conditions present in 2016-2018 were used. As previously noted, these conditions are representative of those often present at lake, with hot dry summers and cool winters with limited rainfall and runoff in most years interspersed with winters of higher rainfall-runoff. Key variables assessed include those stipulated in the TMDL (total N, total P, chlorophyll a and DO concentrations), as well as algal toxins, TDS and other properties.

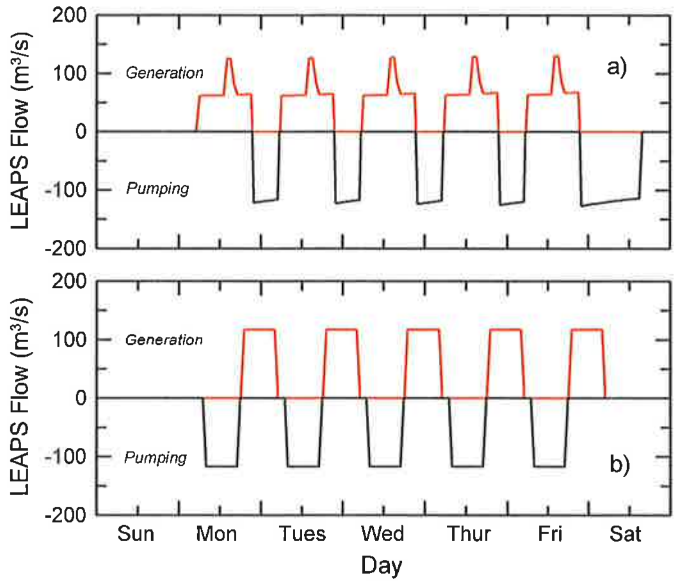

Fig. 13. Pumping-generation schedules used in simulations: a) conventional nighttime pumping-daytime hydropower generation, and b) morning pumping-afternoon/evening hydropower generation (referred to as Schedule 2).

RESULTS

Objective 1. Effects of LEAPS Operation on Water Quality in Lake Elsinore at Different Lake Surface Elevations

Scenario 1: 1235 ft (low lake level)

i. Lake Level

The supplementation of Lake Elsinore with 15,000 af of SWP water increased the lake surface elevation from 1235.2 ft in February 2016 to 1238.2 ft following filling of the Upper Reservoir (Fig.14). Superimposed on the increased lake level are the regular oscillations in lake level of about 1 ft due to water withdrawal and return associated with pumping and hydropower generation (Fig. 14). The Upper Reservoir also experienced regular more dramatic oscillations in lake level (about 60-70 ft) over the course of a day (discussed later).

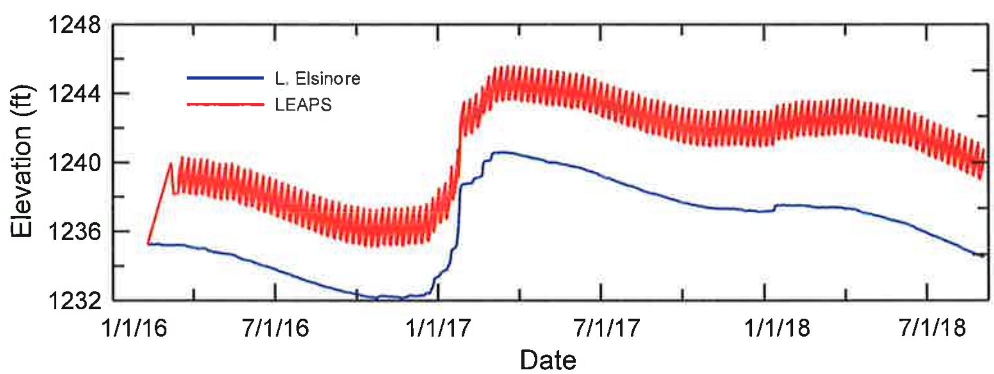

Fig. 14. Surface elevation of Lake Elsinore over time (Scenario 1).

ii. TDS

Supplementation with SWP water as part of the LEAPS project reduced TDS concentrations in Lake Elsinore at the start of simulation as a result of dilution of the high TDS lake water with high quality SWP water (Fig. 15). Runoff in winter 2017 further reduced TDS concentrations, reaching a minimum of nearly 1500 mg/L in March 2017 coinciding with maximum surface elevation (compared with a minimum TDS concentration of about 2000 mg/L for the natural condition). Evapoconcentration resulted in rapid subsequent increases in TDS levels, especially under native conditions in the lake.

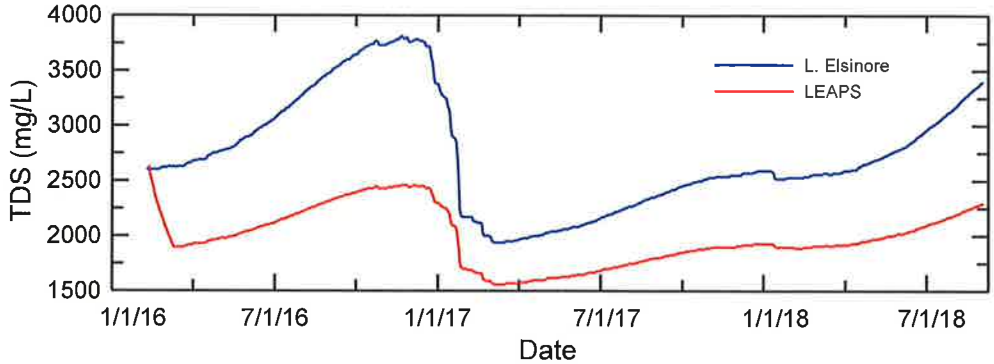

Fig. 15. TDS concentrations in Lake Elsinore over time (Scenario 1).

iii. Temperature

The operation of LEAPS did not substantially alter the temperature regime in the lake (Fig. 16). That is, although lake elevation and volume both increased as a result of supplementation, and TKE inputs increased during hydropower generation, water column temperatures and temperature profiles at site E2 near the center of the lake remained very similar (Fig. 16).

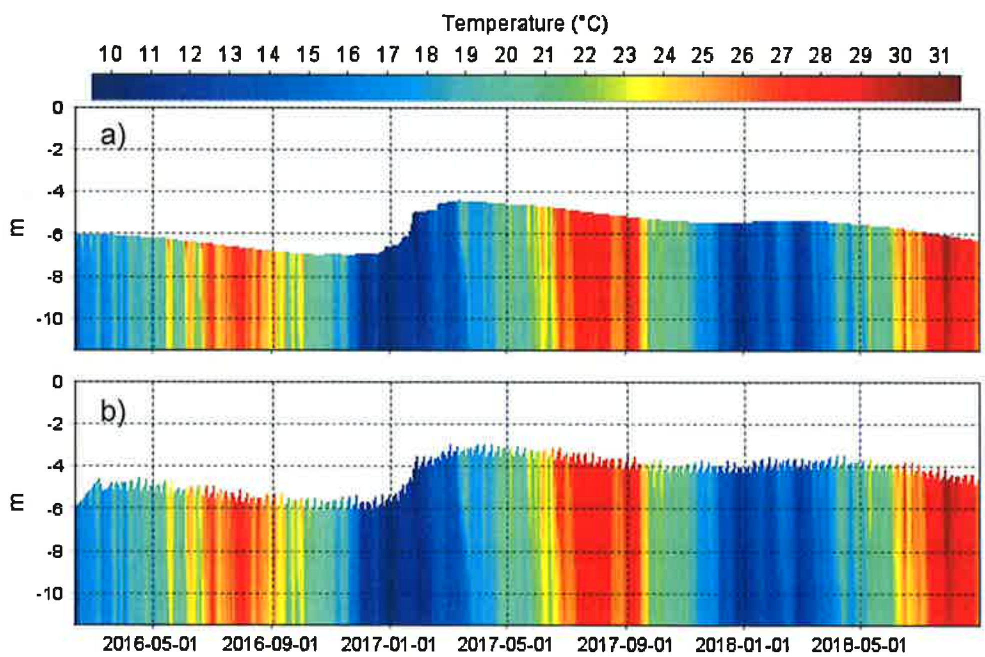

Fig. 16 Temperature profiles in Lake Elsinore over time: a) native condition; b) with LEAPS (Scenario 1).

This can be also be seen in the volume-averaged temperatures in Lake Elsinore that were effectively independent of operation of LEAPS (Fig. 17). Temperatures varied from about 10 – 30 °C seasonally, with short-term increases and decreases due to prevailing weather conditions. Increased lake level, storage in the Upper Reservoir, and the pump-hydropower generation cycles thus did not alter the overall heat budget for the lake.

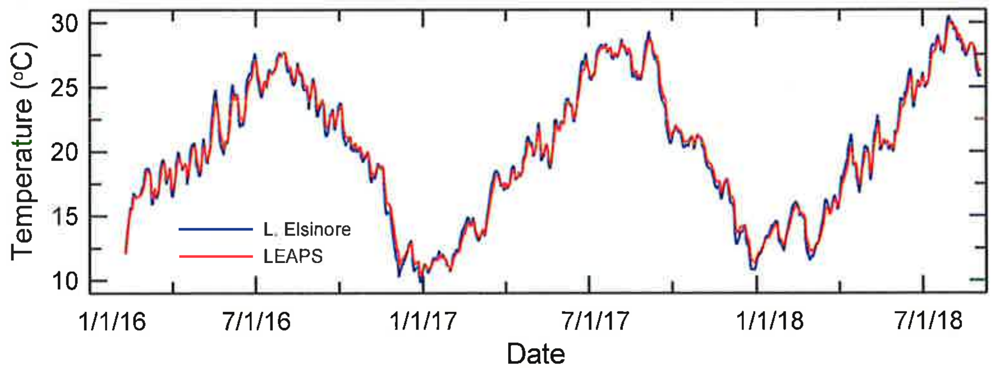

Fig. 17. Volume-averaged temperatures over time (Scenario 1).

iv. Dissolved Oxygen

Dissolved oxygen exhibited greater vertical variation than temperature, with high concentrations near the surface in the late spring and intervals of low concentrations present throughout 2016-18. The operation of LEAPS was not found to substantially change that, although LEAPS exerted more of an effect ón DO concentrations when compared with temperature (Fig. 18a,b).

This can be seen more clearly when comparing volume-averaged DO concentrations (Fig. 18c). For example, LEAPS resulted in slightly lower average DO concentrations in the early spring of 2016 and late spring of 2017 and somewhat higher concentrations in winter of 2017 and 2018 (Fig. 18c). Over the entire simulation period for this low lake surface elevation, the ensemble mean DO concentration with LEAPS operation (5.63 mg/L) was statistically significantly higher than without it (5.49 mg/L), indicating that LEAPS provides some modest net benefits to average DO concentrations in Lake Elsinore.

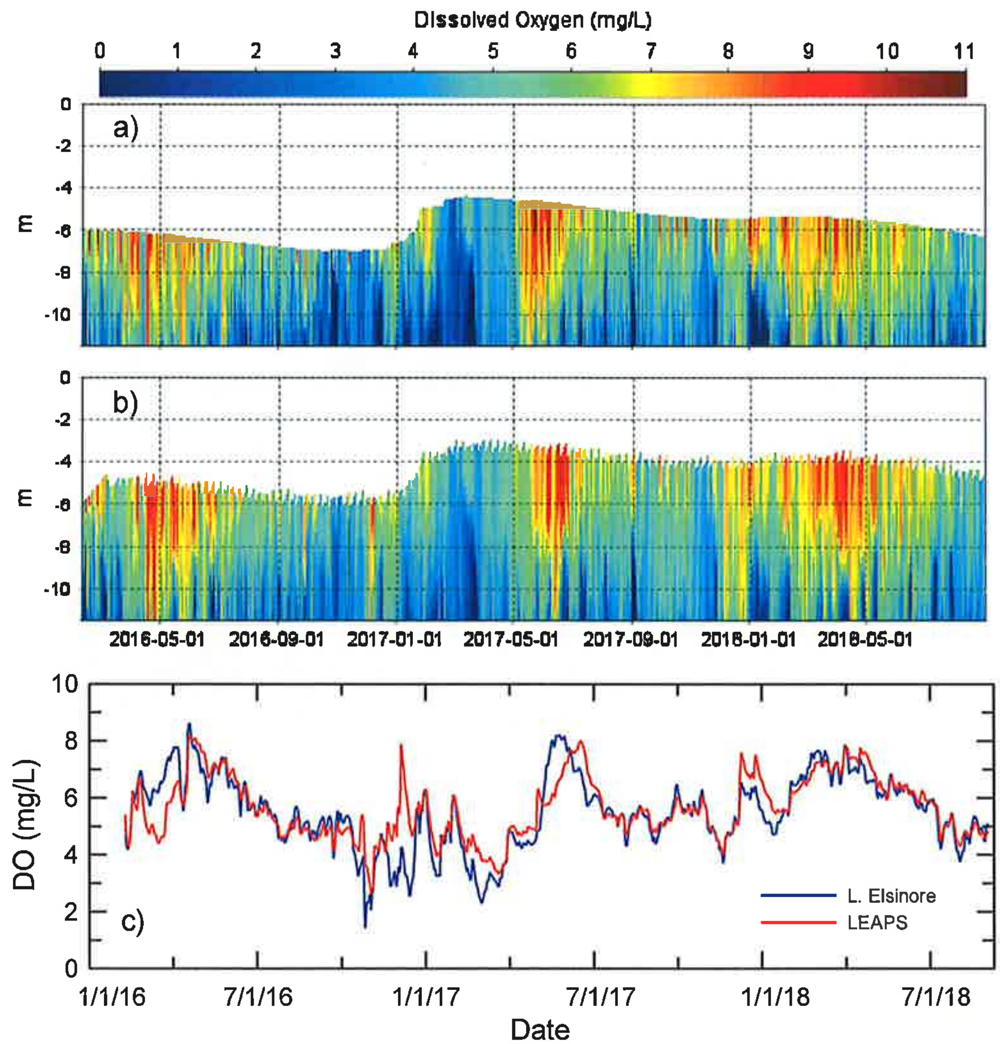

Fig. 18. Dissolved oxygen in Lake Elsinore: a) native condition, b) with LEAPS and c) volume-weighted concentrations over time (Scenario 1).

v. Total N

Supplementation with SWP water and operation of LEAPS was found to have a more pronounced effect on total N concentrations in the lake (Fig. 19). Supplementation with SWP water with low total N (Table 3) resulted in rapid reduction of total N from about 10 mg/L to about 5 mg/L (Fig. 19b); concentrations remained near 5 mg/L through the summer and fall before decreasing further with the winter rains and through the spring of 2017 . This compares, e.g., with values near 8 mg/L in

2016 under natural conditions. Limited vertical differences in total N were typically predicted (Fig. 19).

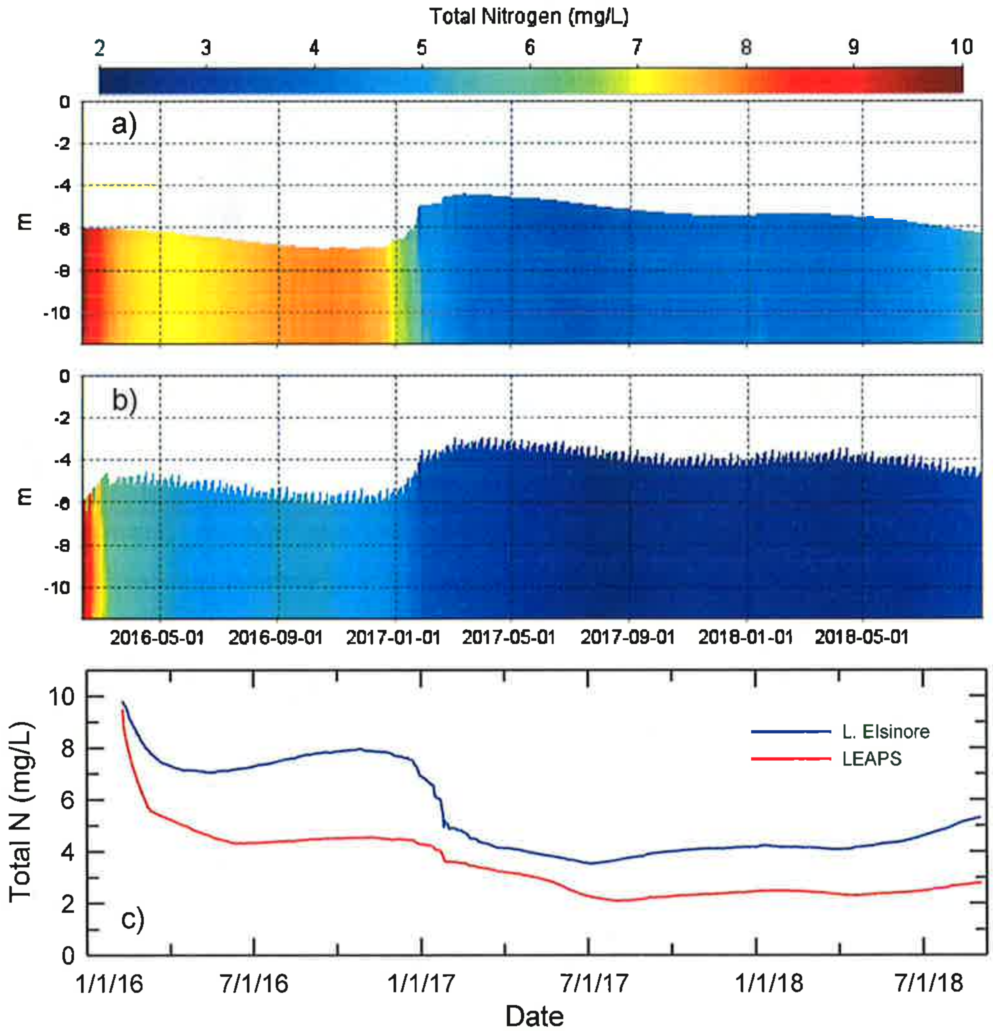

Fig. 19. Total N in Lake Elsinore: a) native condition, b) with LEAPS and c) volume-weighted concentrations over time (Scenario 1).

vi. Total P

Supplementation and operation of LEAPS also yielded lower concentrations of total P when compared with those predicted for the lake without LEAPS (Fig. 20). Total P concentrations were reduced from about 0.3-0.4 mg/L in Lake Elsinore without LEAPS in 2016, to 0.2 mg/L in spring 2016 with LEAPS and remained lower throughout the year. Concentrations in the lake under both native conditions and with

LEAPS declined further following brief increases from runoff inputs in January-February 2017, with concentrations about 50% lower with LEAPS (Fig. 20).

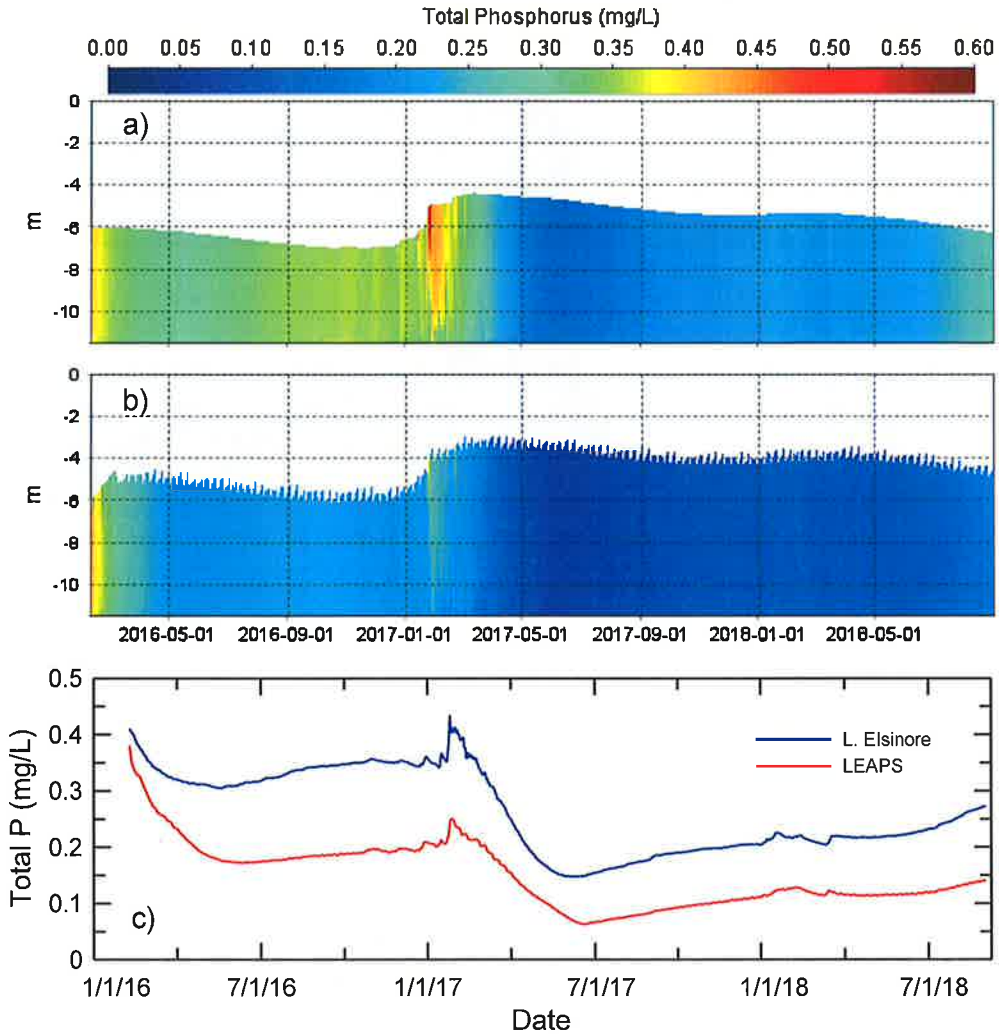

Fig. 20. Total P in Lake Elsinore: a) native condition, b) with LEAPS and c) volume-weighted concentrations over time (Scenario 1).

vii. Chlorophyll a

Concentrations of chlorophyll a were about 50 µg/L lower with water supplementation and operation of LEAPS compared with natural lake values (Fig. 21). The reductions were due in large measure to the above noted dilution of nutrients through addition of low nutrient SWP water (Table 3).

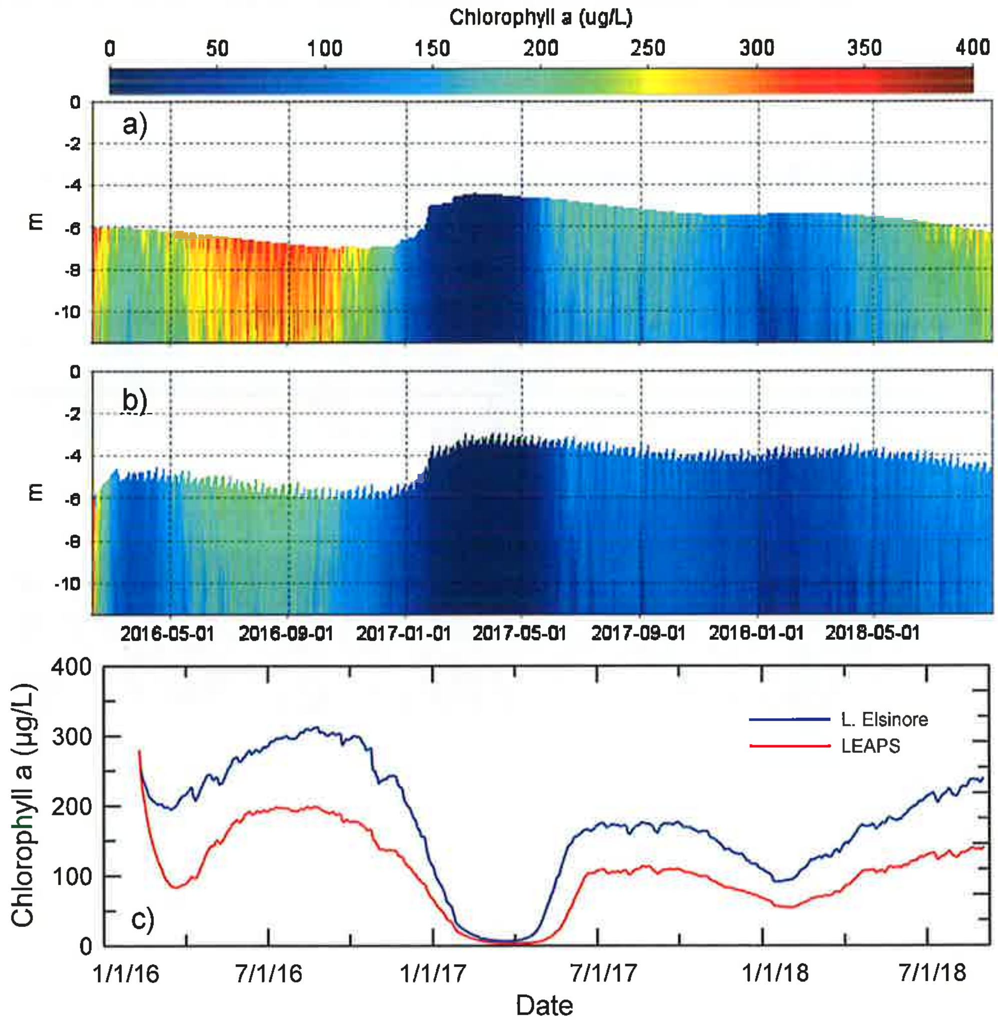

Fig. 21. Chlorophyll a in Lake Elsinore: a) native condition, b) with LEAPS and c) volume-weighted concentrations over time (Scenario 1).

Chlorophyll a concentrations were generally higher in the shallow southern end of the lake under native conditions (Fig. 22a), while LEAPS lowered concentrations and weakened slightly horizontal gradients in concentration (Fig. 22b).

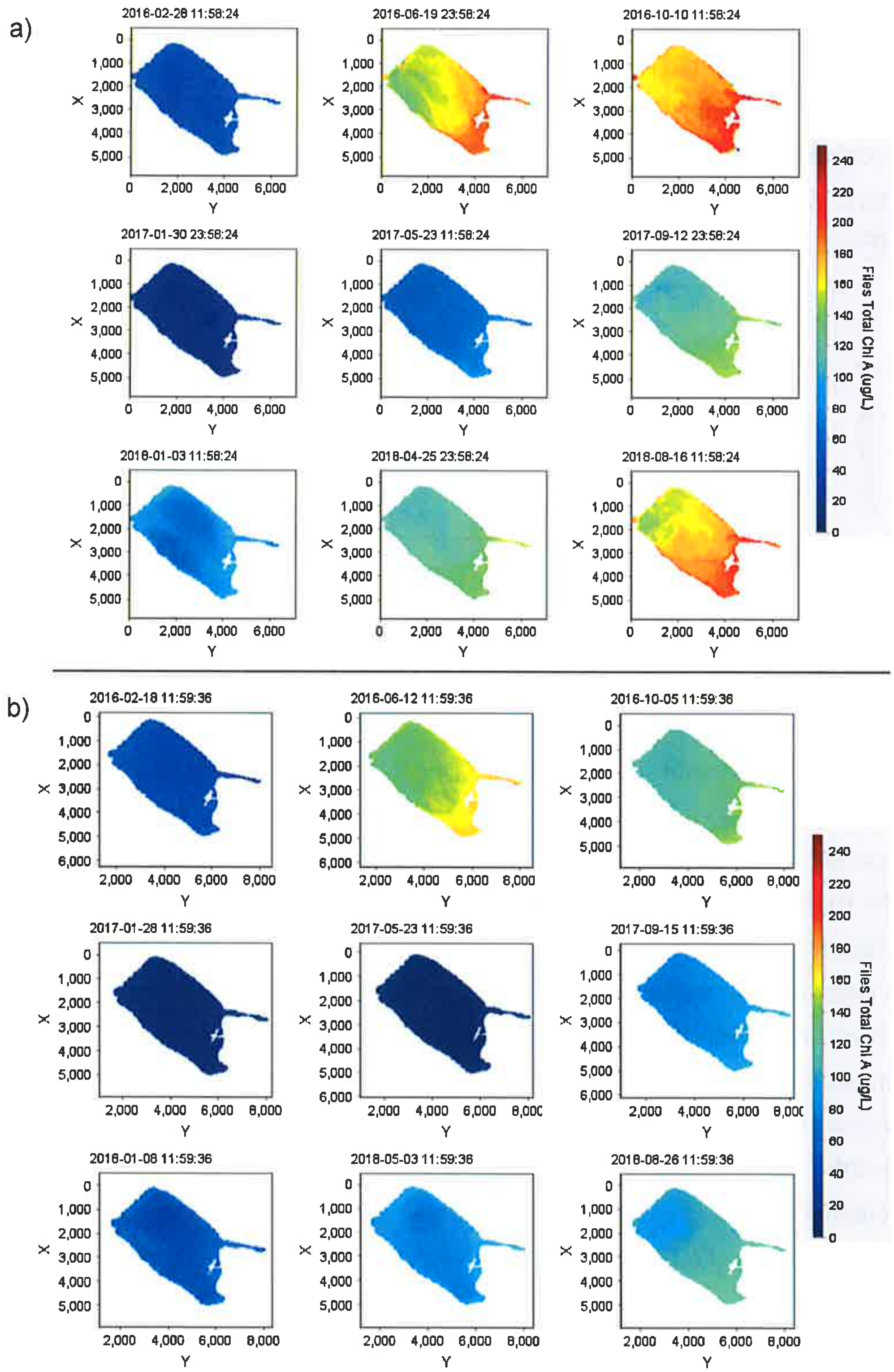

Fig. 22. Distribution of chlorophyll a in Lake Elsinore: a) native condition, and b) with LEAPS (Scenario 1).

viii. Microcystins

Based upon the linear regression of measured microcystin concentrations with chlorophyll a levels (Fig. 11), LEAPS was predicted to yield substantially lower microcystin levels in the lake compared with native conditions, with concentrations about 5-10 µg/L lower over most of the simulation and negligible during much of 2017-18 (Fig. 23).

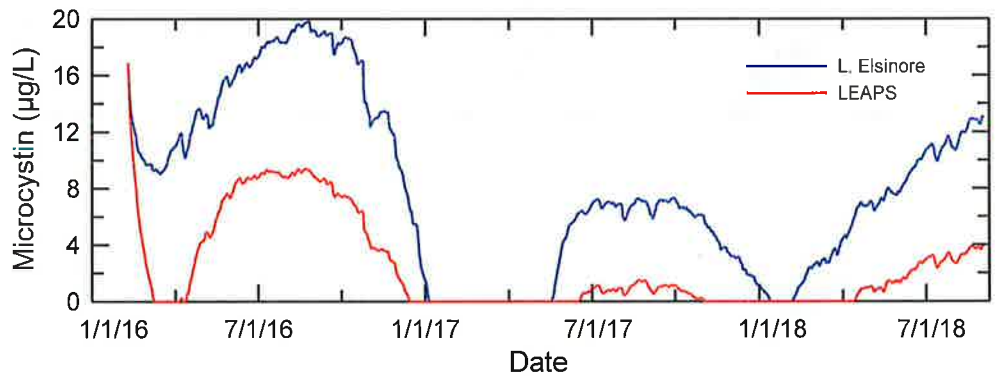

Fig. 23. Microcystin concentrations in Lake Elsinore over time (Scenario 1).

Cumulative Distribution Functions

The time series data depicted in Figs. 14-23 are presented as cumulative distribution functions (CDFs) that summarize the frequency and distribution of concentrations or other water column properties under the native condition present in 2016-18 (Fig. 24, blue lines) and with LEAPS in place over this same time period (Fig. 24, red lines). An exceedance frequency of 50% corresponds to the median value for the dataset. Lake surface elevation was shifted to higher values and TDS to lower concentrations with LEAPS through supplementation with SWP water (Fig. 24a,b). Temperature (Fig. 24c) was unaffected by operation of LEAPS, while LEAPS shifted DO concentrations to higher values at low concentrations (Fig. 24d). Supplementation with SWP water and operation of LEAPS yielded lower concentrations of nutrients, chlorophyll a and microcystins when compared with the natural conditions (i.e., without LEAPS) (Fig. 24e,f,g,h).

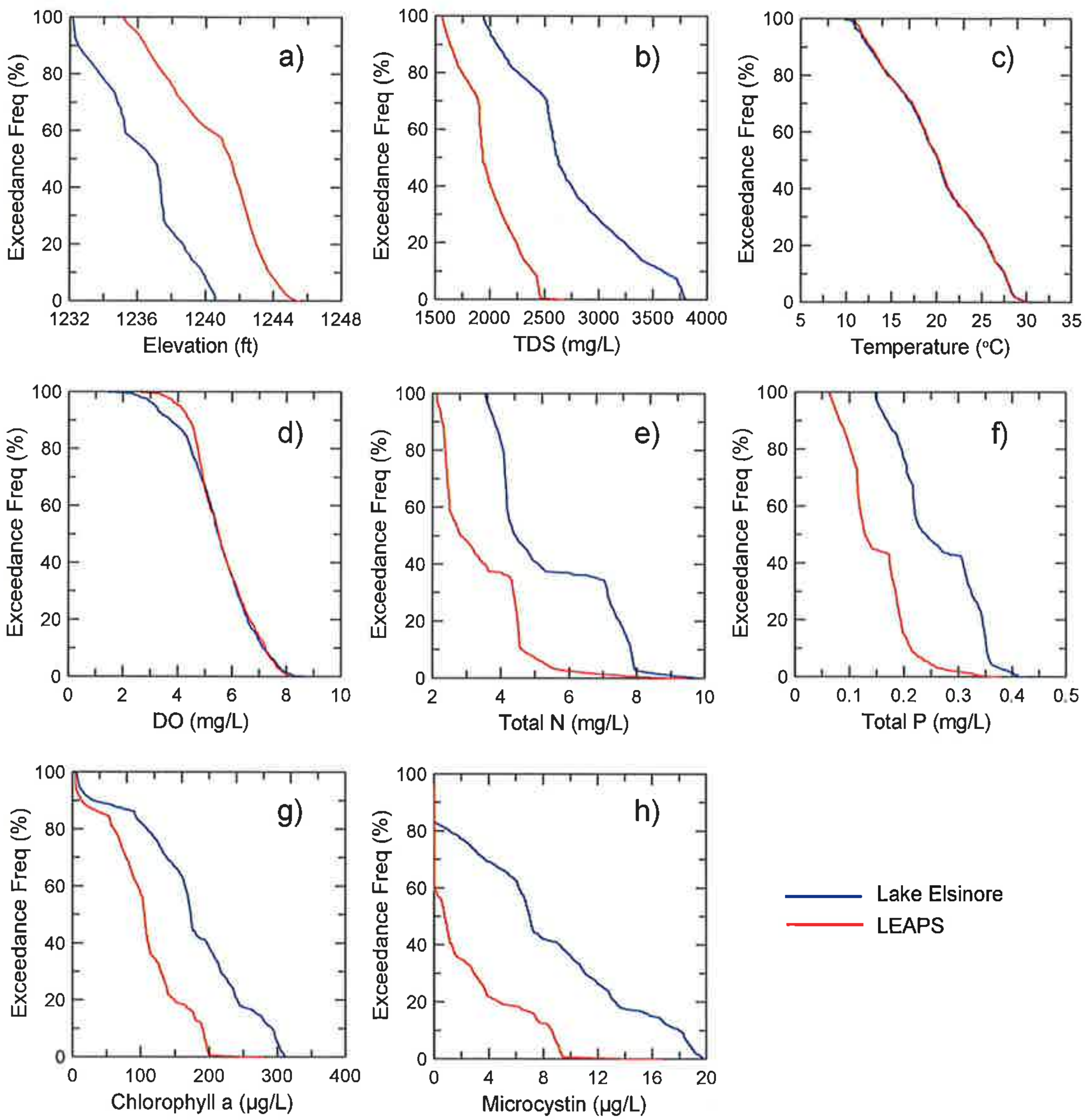

Fig. 24. Cumulative distribution functions for water column properties in Lake Elsinore under native conditions (blue lines), and with LEAPS (red lines): a) elevation, b) TDS, c) temperature, d) DO, e) total N, f) total P, g) chlorophyll a and h) microcystin (Scenario 1).

Scenario 2: 1240 ft (moderate lake level)

Simulations also evaluated water quality associated with operation of LEAPS at higher lake surface elevations, in this case assuming an initial elevation of approximately 1240 ft, or nearly 5 ft higher than the preceding analysis. For this scenario, the measured water quality in Lake Elsinore on February 3, 2017 (Amec Foster Wheeler, 2017), when the lake was at 1240 ft (EVMWD, unpubl data), was used as the initial condition for the simulation. The meteorological and hydrologic conditions present in 2016-18 were again used to drive the model since they represent relatively common conditions at the lake and allow direct evaluation of lake surface elevation effects.

Findings from prior EFDC modeling indicated that somewhat greater mixing of the water column could be achieved at narrower I/O dimensions than the originally proposed approximately 150 m wide I/O without chronic sediment resuspension (Anderson, 2007). This was further evaluated here, where two I/O designs were evaluated: (i) an approximately 150 m wide I/O and (ii) a narrower approximately 50 m wide I/O. The lateral extent is taken as the hypotenuse of the square cells. In addition, the two pump-generation schedules (Fig. 12) were evaluated for LEAPS operation at this nominal minimum target operating level (for the 50 m wide I/O) to evaluate how details of the operation of LEAPS would affect water column conditions at the lake.

i. Lake Level

The surface elevation of Lake Elsinore was predicted to undergo similar changes over time at this higher initial lake level as in Scenario 1, with more than 3 ft of water lost due to evaporation between February and November 2016, followed by a dramatic increase of nearly 8 ft in early 2017 (Fig. 25). The absolute increase was somewhat less than seen at the low lake level due to the larger area and initial volume. Supplementation with SWP water increased lake level by 5 ft to a maximum surface elevation of 1243.9 ft in early March 2016, followed by regular oscillations in lake level associated with pumping-hydropower generation that were independent of schedule or dimensions of the I/O. Seasonal trends with LEAPS operations followed very closely that in Lake Elsinore under native conditions (Fig. 25).

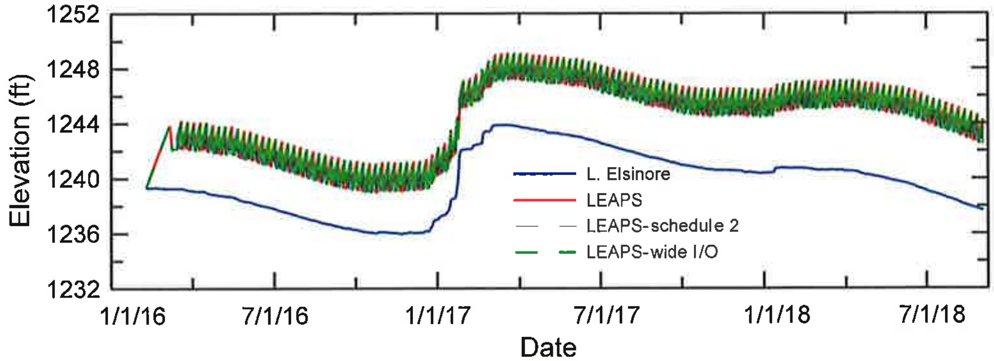

Fig. 25.Surface elevation in Lake Elsinore over time (Scenario 2).

ii. TDS

Supplementation with 15,000 af of high quality SWP water increased lake level by about 5 ft and significantly lowered concentration of TDS in the lake (Fig. 26). The effect is somewhat less dramatic compared with results for the lake at extremely low lake elevations and high background TDS levels (Fig. 15) owing to the lower initial TDS concentration at the 1240 ft surface elevation. Notwithstanding, the lower TDS would be expected to improve freshwater ecology in the lake relative to the natural condition. The pump-hydropower generation schedule predictably did not affect lake TDS levels (Fig. 26). The width of the I/O also had no effect and yielded TDS concentrations that were the same as those shown for LEAPS and with the two different operational schedules.

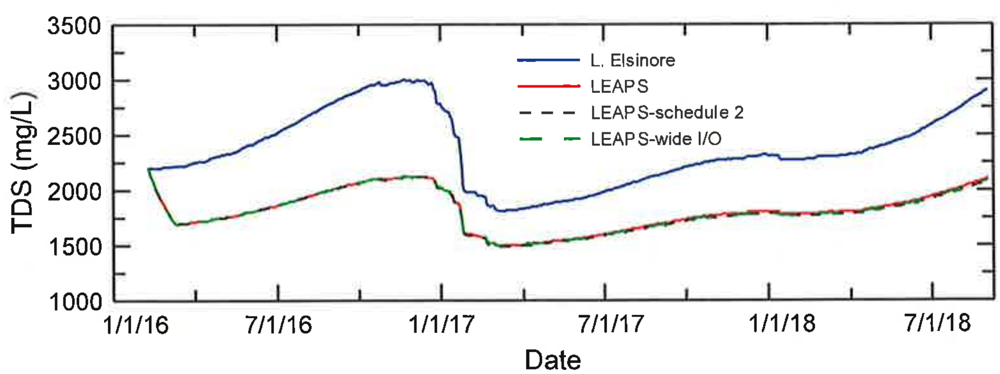

Fig. 26 Volume-weighted TDS concentrations in Lake Elsinore over time (Scenario 2).

iii. Temperature

The higher lake elevation in this scenario yielded occasional evidence of slightly stronger thermal stratification in Lake Elsinore (Fig. 27a) when compared with the extremely low lake level scenario (Scenario 1) (Fig. 16), but the water column was relatively well-mixed most of the time. Supplementation with SWP water increased lake level further, although evidence of additional increases in thermal stratification were not evident for either of the 2 pump-generation schedules likely due at least in part to increased TKE inputs (Fig. 27b,c). Simulation of the operation of LEAPS with the wide (about 150 m) I/O on Lake Elsinore yielded very similar results (Fig. 25d).

The supplementation with SWP water and operation of LEAPS did not affect the overall heat budget for the lake, with volume-averaged temperatures effectively indistinguishable with those for Lake Elsinore (Fig. 28a). While averaged values provide information about overall heat budget, they do not provide information about the presence and intensity of any thermal stratification however, which can be seen more clearly by comparing surface and bottom water temperatures over time. Here temperature differences outputted 6 times a day at station E2 with and without LEAPS are presented as time-series (Fig. 28b).

Weak diurnal stratification was evident most afternoons (ΔT often 0.5-1 °C or more) followed by uniform temperatures later in the evening-early morning. Intervals of more persistent stratification were often present during the spring time (Fig. 28b) with temperature differences of 2ºC or more, and without nighttime cooling and return to uniform temperatures (Fig. 28b). Operation of LEAPS following supplementation with SWP water had comparatively little effect, but did weaken slightly the median ΔT value (Table 4). The effect was influenced by the I/O design, with slightly lower mean, median and 95% ΔT values with a ~50 m wide I/O compared with the wide I/O due to higher TKE inputs during hydropower generation. The operation of LEAPS with night-time pumping and day-time generation increased slightly the number of days when ΔT exceeded 2°C, from 33 to 38, while the morning pumping/afternoon-evening generation schedule with the ~50 m wide I/O lowered the number of days and yielded the lowest set of ΔT values in this analysis (Table 4).

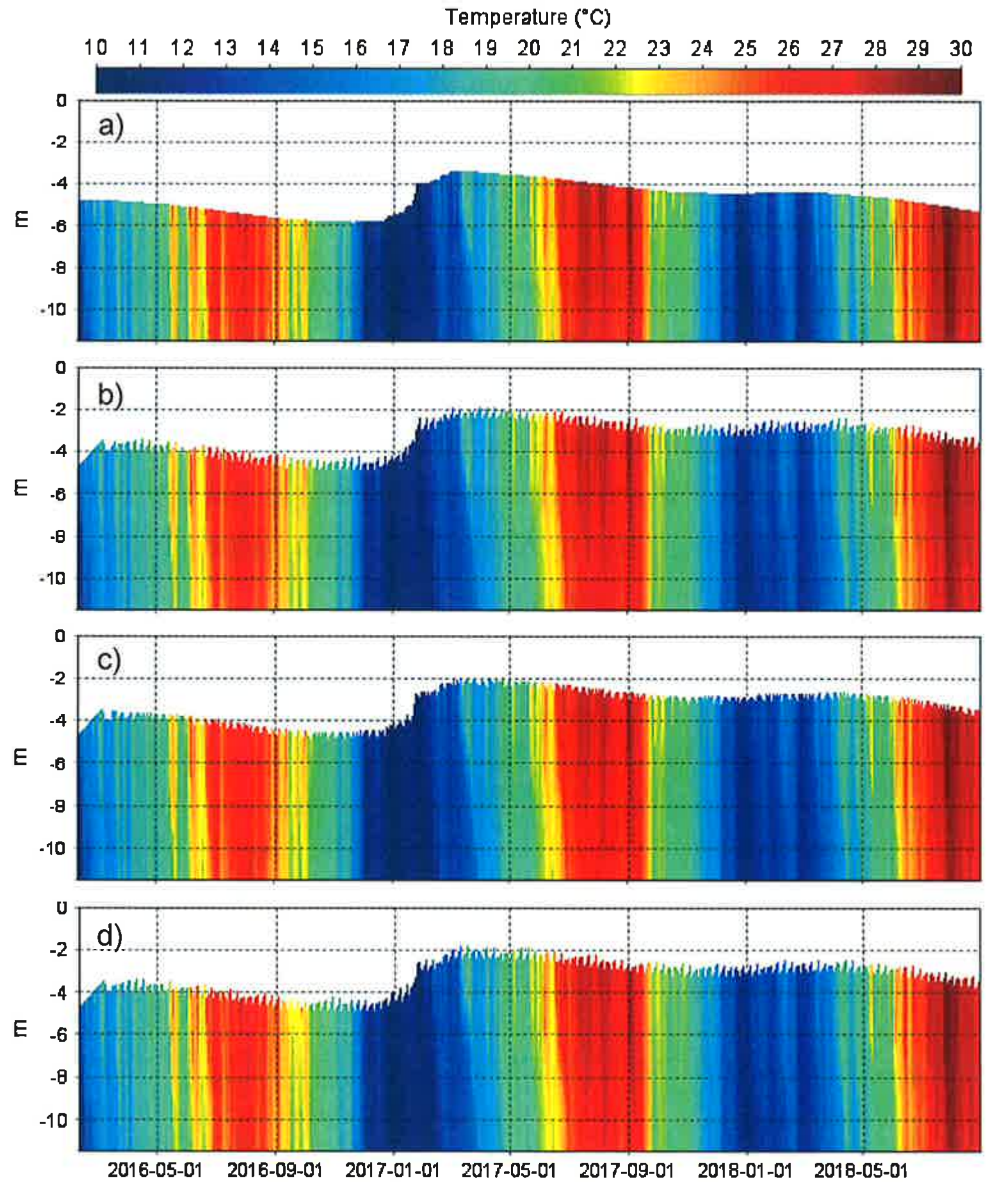

Fig. 27. Temperature profiles in Lake Elsinore over time: a) native condition, b) with LEAPS, c) with LEAPS with morning pumping- afternoon/evening hydropower generation (schedule 2), and d) with LEAPS with wide (150m) I/O (Scenario 2).

The dimensions of the I/O influence the TKE input to the lake during hydropower generation, with smaller intake cross-sectional area A (m^2) yielding a larger outflow velocity U (m/s) at a given volumetric flow rate Q (m^3/s) following:

$$U = \frac{Q}{A} \qquad (1)$$

The outflow velocity (and flow rate) in turn affect the TKE input (W/m^2) to the lake during power generation (Imboden, 1980):

$$TKE = \frac{QU\rho_{in}}{2A_0} \qquad (2)$$

where ρ_{in} is the density of inflowing water (kg/m^3) and A_0 is the lake surface area (m^2). From these equations, one can see that the TKE input will vary inversely with I/O cross-sectional area at a given flow rate, so a reduction to one-third the cross-sectional area of the I/O will increase the velocity and thus also the TKE input by a factor of 3 (e.g., under average operational conditions, from 2.1x10^{-4} W/m^2 to 6.4x10^{-4} W/m^2). This is supported by ΔT values (Table 4), although the effect away from the I/O and deeper in the lake is quite small.

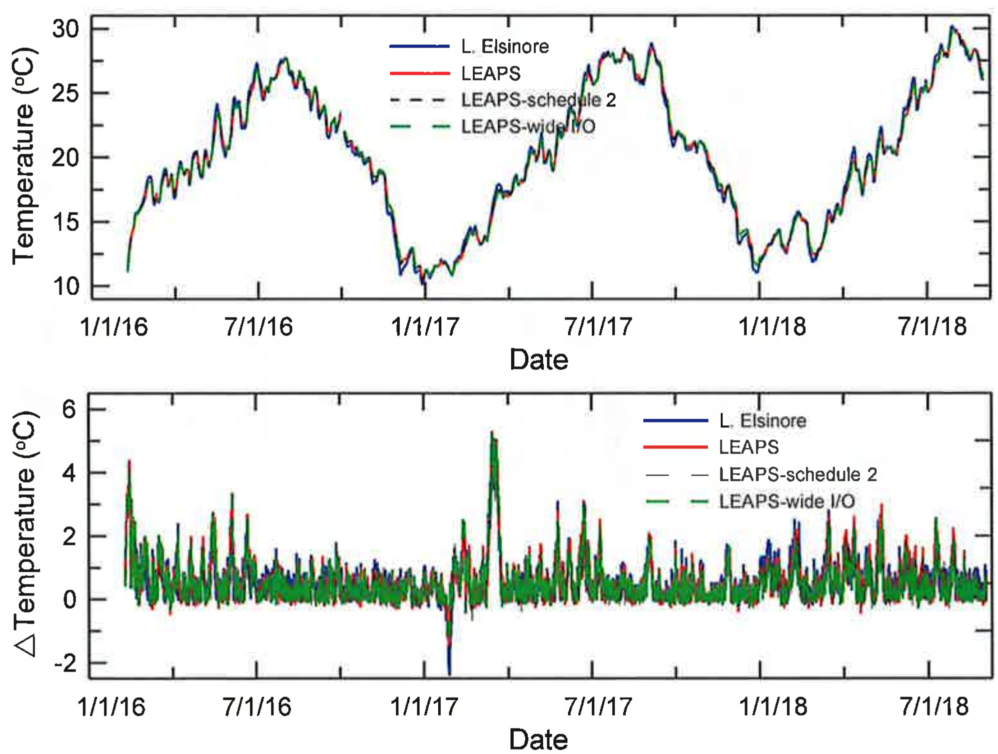

Fig. 28. Temperature in Lake Elsinore: a) Volume-weighted temperatures over time; b) temperature difference (ΔT) between near-surface and lower depth waters (1 m and 5 m, respectively) (Scenario 2).

Table 4. Statistical results for temperature difference (ΔT) between near-surface and lower depth waters (Scenario 2). (LEAPS and LEAPS-schedule 2 simulations with 50 m I/O, LEAPS-wide simulation with 150 m I/O.)

ΔT (°C)	L. Elsinore	LEAPS	LEAPS-sched 2	LEAPS-wide
Mean	0.58	0.57	0.52	0.58
Median	0.44	0.40	0.37	0.41
95%	1.75	1.87	1.74	1.96
Maximum	5.10	5.30	5.01	5.25
# Days >2°C	33	38	31	36

iv. DO

Natural wind-driven mixing/aeration and operation of the diffused aeration system and axial flow pumps failed to sufficiently mix and aerate the lake such that low DO concentrations were often present at this higher elevation as well (Fig. 29a). The DO concentrations were influenced by lake levels, which tended to yield lower values near the lake bottom and, although subtle, by operation of LEAPS which generally reduced duration and intensity of intervals of low DO (e.g., in winter and early spring in 2017 and 2018) (Fig. 29b,c,d).

Periodic intervals of low DO concentrations in the lower water column underscores the difficulty in maintaining oxic conditions near the bottom sediments in this highly eutrophic lake, although LEAPS operation was observed to reduce somewhat duration and magnitude of low DO concentrations (Fig. 29). The volume-averaged DO concentrations indicate some seasonal variation that was not strongly altered by operation of LEAPS (Fig. 30); the number of days in which the volume-averaged DO concentration in the lake was <5 mg/L was reduced from 13 days (1.4%) in the native condition to 5 days (0.5%) with both the nighttime pumping/daytime generation schedule for both I/O configurations and with morning pumping/afternoon-evening generation. Thus, there is not predicted to be a significant effect on average DO concentrations in the lake under either pumping-hydropower generation schedule.

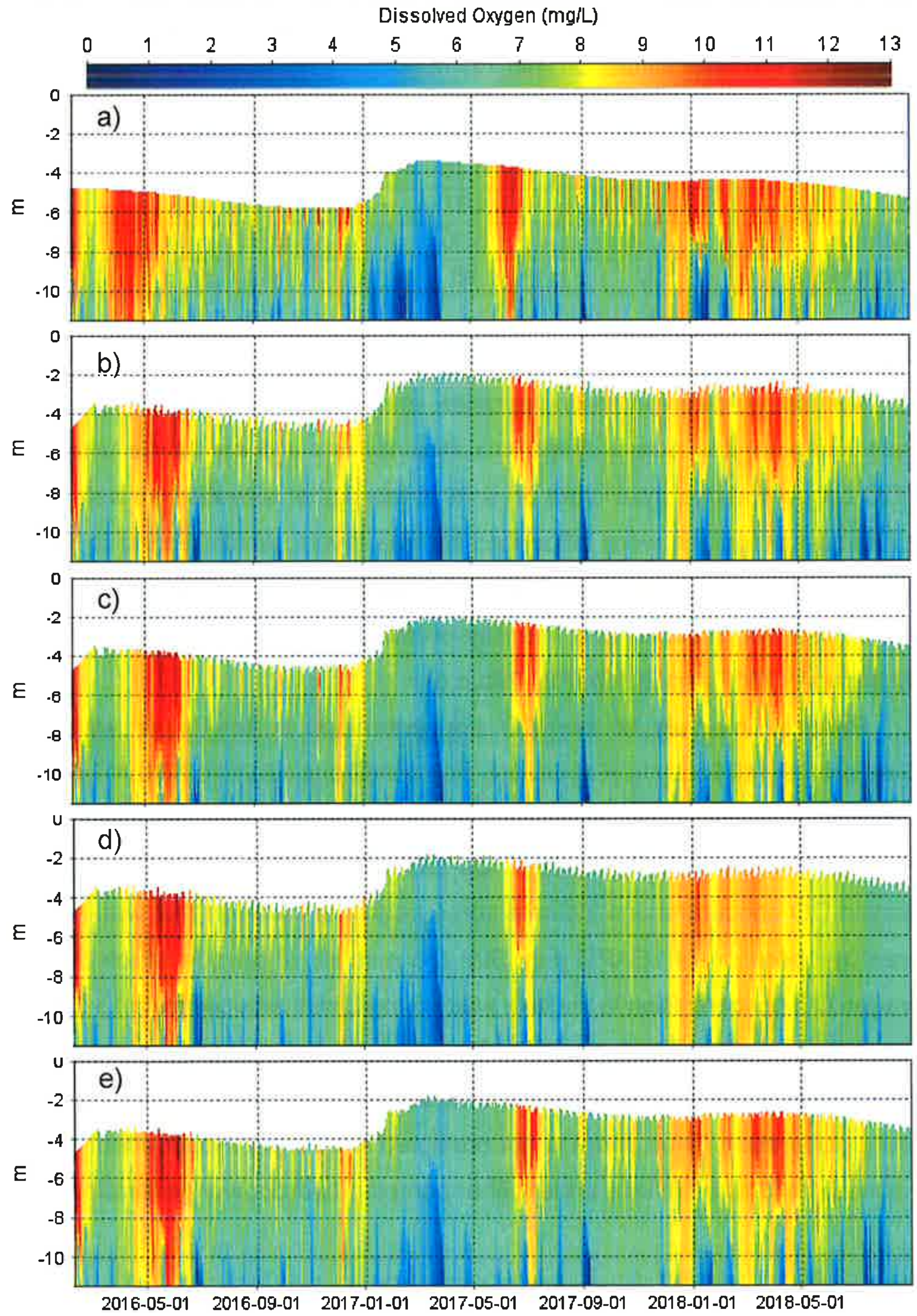

Fig. 29. DO profiles in Lake Elsinore over time: a) native condition, b) with LEAPS, c) with LEAPS with morning pumping- afternoon/ evening hydropower generation, and d) with LEAPS with wide (150m) I/O (Scenario 2).

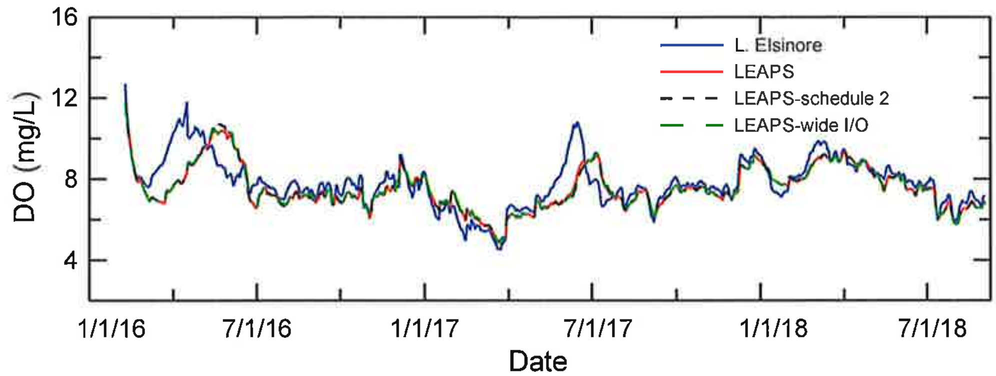

Fig. 30. Volume-weighted DO concentrations over time (Scenario 2).

While volume-averaging of DO provides useful basin-wide measures of water quality in the lake, it is recognized that the potential exists for the presence of strong spatial gradients. For example, the volume-averaged concentration of DO in mid-July 2017 was about 7 mg/L in the lake under native conditions and with LEAPS operation (Fig. 30), although the DO concentration above the bottom sediments varied quite strongly across the lake and with LEAPS operation (Fig. 31). Here we see high DO concentrations in the shallow waters in the southern and eastern part of Lake Elsinore, and a region of lower DO, on the order of 2-3 mg/L, near the deeper central part of the lake under native conditions (Fig. 31a). With LEAPS in place, higher concentrations of DO were predicted for much of the lake bottom, including bottom waters on the northern and eastern margins of the lake, as well as in the central deeper part of the lake (Fig. 31b). The spatial distributions of DO varied markedly over time as a function of both LEAPS operation and wind-driven circulation and mixing, with instances of little differences in DO, and occasionally lower DO values as well, with LEAPS operation compared to native conditions.

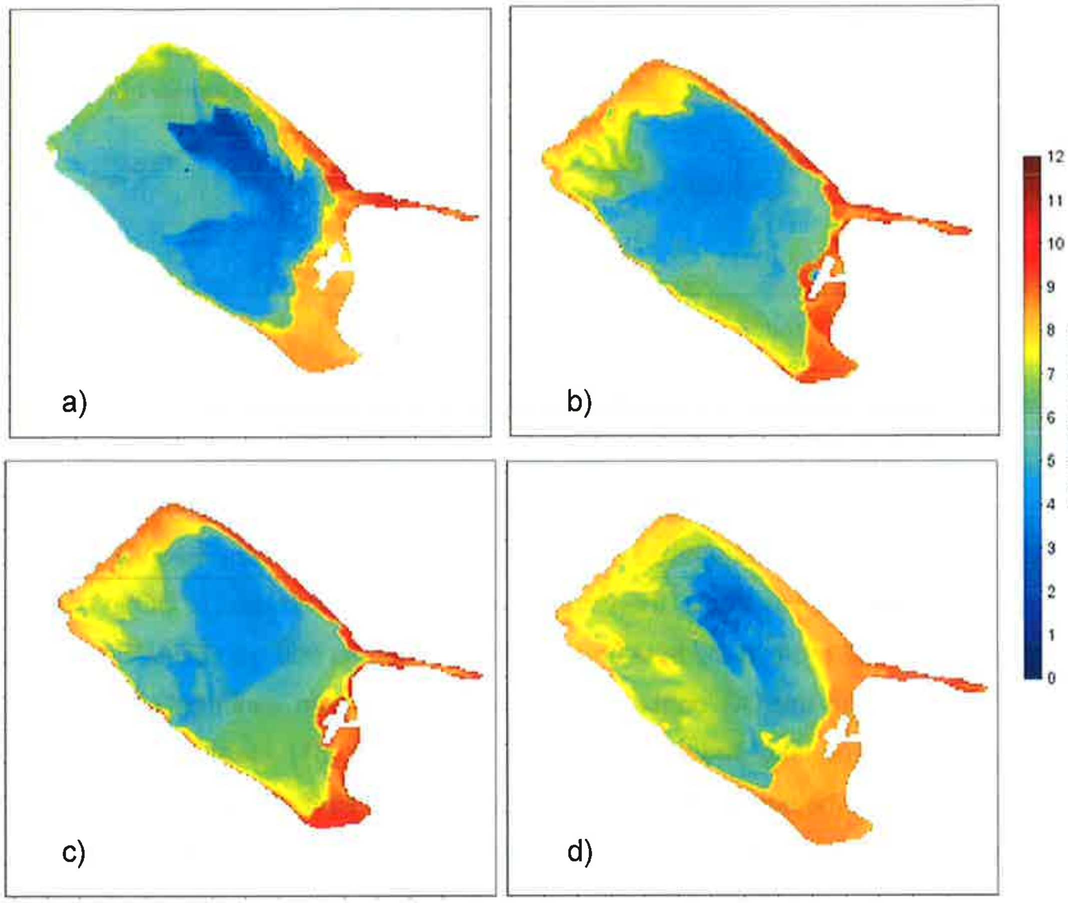

Fig. 31. Distribution of DO directly above sediments on 7/12/2017: a) native condition, b) with LEAPS, c) with LEAPS with morning pumping-afternoon-evening hydropower generation, and d) with LEAPS with wide (150 m) I/O (Scenario 2).

v. Nutrient Concentrations

Time-series of the volume-averaged concentrations of total N, total P, and NH_4-N are presented in Fig. 32. As previously noted, supplementation with low nutrient SWP water significantly lowered nutrient concentrations. As observed with TDS, the absolute and relative reductions in concentrations were somewhat lower than observed when the lake was at its extremely low lake level, but total N, total P and NH_4-N concentrations were all significantly reduced (Fig. 32). The details of LEAPS design and operation had a negligible effect on predicted average nutrient concentrations; for example, the two LEAPS operation schedules yielded volume-averaged total N, total P and NH_4-N concentrations that were essentially identical over the duration of the simulations (Fig. 30). The width of the I/O also had no substantive effect on nutrient concentrations (Table 5).

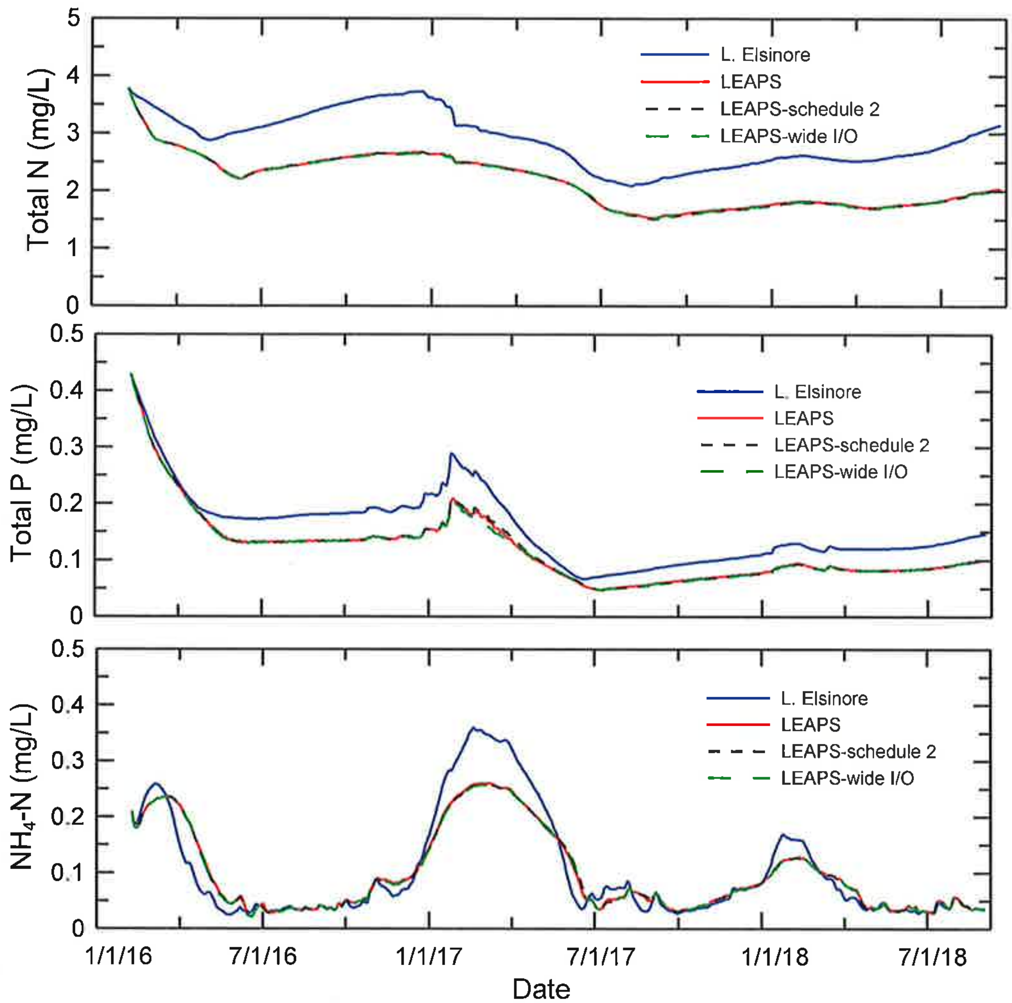

Fig. 32. Volume-averaged concentrations of nutrients in Lake Elsinore over time: a) total N, b) total P, and c) NH_4-N (Scenario 2).

vi. Chlorophyll a

Chlorophyll a levels varied seasonally, with low concentrations predicted during cool winter months and much higher values in late spring, summer and fall (Fig. 33). Supplementation and operation of LEAPS yielded reductions of about 50 µg/L, e.g., from about 180 µg/L to approximately 130 µg/L under summer 2016 conditions. Predicted average chlorophyll a concentrations were not affected by either LEAPS operation schedule nor by the two I/O widths evaluated (Fig. 33, Table 5).

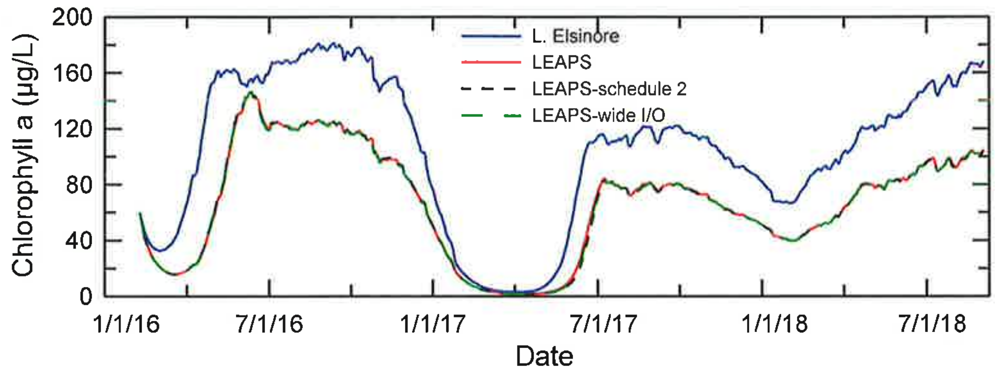

Fig. 33. Volume-weighted chlorophyll a concentrations in Lake Elsinore over time (Scenario 2).

vii. Microcystin

Volume-weighted microcystin concentrations predicted from linear regression (Fig. 11) necessarily followed trends in chlorophyll a closely, with concentrations reaching 6-8 µg/L in summer-fall 2016 under the native condition and lower concentrations of 2-4 µg/L with LEAPS (Fig. 34). Reduced concentrations were predicted in the lake under native conditions in 2017, but values were predicted to rise more significantly in 2018, while microcystin was absent in 2017 and most of 2018 with LEAPS (Fig. 34). Details of LEAPS operation were not predicted to have a meaningful effect on volume-averaged microcystin concentrations (Fig. 34, Table 5).

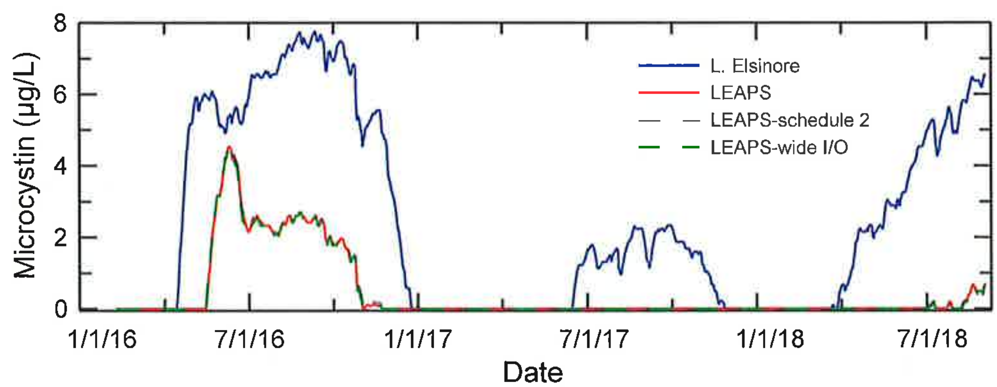

Fig. 34. Volume-weighted microcystin concentrations in Lake Elsinore over time (Scenario 2).

Table 5. Statistics for Scenario 2 (1240 ft - moderate lake level) operational schemes.

Property	Value	L Elsinore	LEAPS	LEAPS - Schedule 2	LEAPS - Wide I/O
Lake Level	Mean	1239.8	1244.1	1244.1	1244.1
	Median	1240.1	1245.2	1245.2	1245.2
	Min	1236	1239	1239	1239
	Max	1243.9	1248.3	1248.3	1248.3
TDS	Mean	2364	1810	1800	1810
	Median	2300	1800	1780	1800
	Min	1810	1490	1490	1500
	Max	3000	2200	2200	2200
Total N	Mean	2.89	2.16	2.15	2.16
	Median	2.89	2.17	2.16	2.17
	Min	2.09	1.53	1.51	1.52
	Max	3.79	3.79	3.79	3.79
Total P	Mean	0.16	0.12	0.12	0.12
	Median	0.14	0.10	0.10	0.10
	Min	0.07	0.05	0.05	0.05
	Max	0.43	0.43	0.43	0.43
DO	Mean	7.8	7.6	7.6	7.6
	Median	7.7	7.4	7.4	7.4
	Min	4.5	4.8	4.9	4.8
	Max	12.7	11.8	11.8	11.9
Chlorophyll a	Mean	105	67	67	69
	Median	113	73	73	74
	Min	3.1	1.7	1.6	1.7
	Max	181	146	148	147
Microcystin	Mean	2.4	0.4	0.5	0.4
	Median	1.5	0.0	0.0	0.0
	Min	0	0.0	0.0	0.0
	Max	7.8	4.5	4.7	4.5

Scenario 3: 1247 ft (high lake level)

The effect of LEAPS operation on water column conditions in Lake Elsinore at high lake levels was also assessed. As with other simulations, meteorological and hydrological conditions from February 2016 – August 2018 were used as input files for the model with the ~50 m wide I/O. Due to the higher lake elevation, supplementation with SWP water was reduced to that needed to fill the Upper Reservoir (delivered at a rate of approximately 3.2 m^3/s for 30 days). Initial water quality conditions were set as those observed on September 28, 2006 when the lake was at 1247 ft above MSL and for which initial conditions were available. The initial water column temperature was set to 12.0 - 12.2°C as found in the lake on February 8, 2016.

i. Lake Level

The supplementation with sufficient SWP water to fill the Upper Reservoir and annually balance evaporative losses (but provide no additional net water to Lake Elsinore) yielded lake elevations that followed those for Lake Elsinore quite closely. That is, with LEAPS operation with the Upper Reservoir filled, the elevation of Lake Elsinore was essentially equivalent to the level of the lake in its native (non-LEAPS) state; hydropower generation temporarily increased the lake level by about 1 ft and pumping subsequently lowered the lake level (Fig. 35).

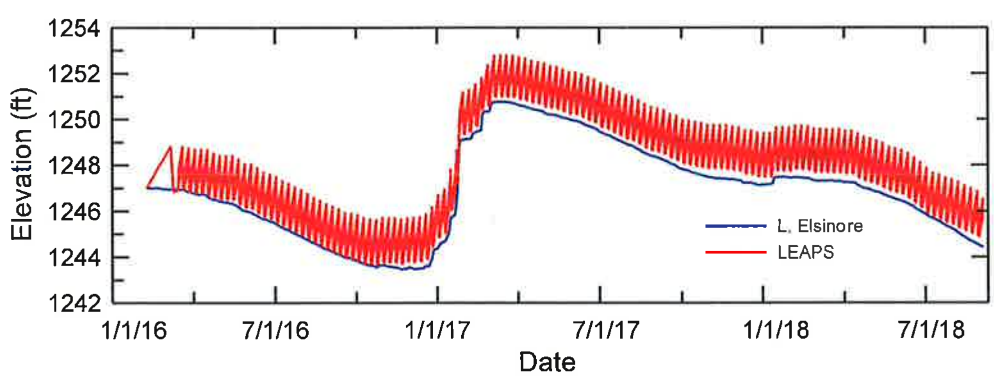

Fig. 35. Surface elevation of Lake Elsinore over time (Scenario 3).

ii. TDS

High lake levels are associated with large runoff events that deliver low TDS water to the lake that in turn dilute background salinity levels in the lake. Predictably, this scenario had the lowest concentration of TDS of any of the scenarios evaluated and was less strongly affected by the large runoff event in winter 2017 owing to the low initial TDS concentration and the larger lake volume (Fig. 36). Supplementation with SWP water also had only a modest effect on TDS concentrations due to the reduced volume added and smaller difference in TDS of the lake and influent.

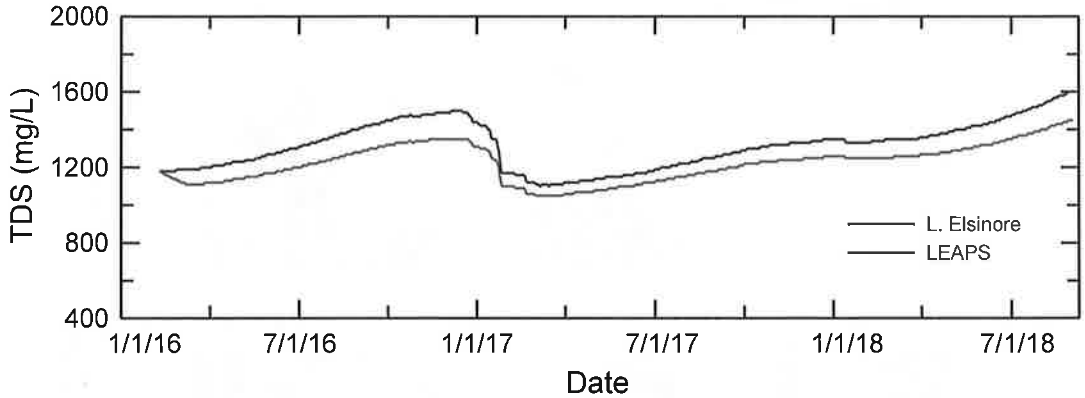

Fig. 36. Volume-weighted TDS concentrations in Lake Elsinore over time (Scenario 3).

iii. Temperature

Water column temperatures in the lake were similar to those observed at lower surface elevations, increasing from lows of 10-12°C in the winter to maximum summer values of 28-30°C (Fig. 37a). Some thermal stratification was predicted during the winter-spring of each year; this can be seen more clearly in the ΔT time series, with values periodically >2°C and persisting for several days up to a week or more (Fig. 37c).

Operation of LEAPS yielded similar temperature profiles over time with the daily lake level oscillations superimposed on the seasonal elevation and temperature trends (Fig. 37b). Clear differences associated with LEAPS operation are not apparent in the temperature profiles (Fig. 37a,b), and only occasional subtle differences evident in ΔT values (e.g., slightly lower ΔT values in January-February 2017) (Fig. 37c). Across the dataset, LEAPS operation yielded ΔT values that were slightly lower than those predicted for the native condition (Table 6). At these higher elevations, a total of

179 days were predicted when ΔT in the lake exceeded 2°C under native conditions, and 169 with operation of LEAPS. This compares with 31-38 days when ΔT>2°C at lower (about 1240 ft) surface elevation for Scenario 2 (Table 4). The intensity and duration of stratification is a function of lake surface elevation and is not strongly affected by operation of LEAPS under either of the two hydropower schedules nor by the two I/O widths evaluated in this study.

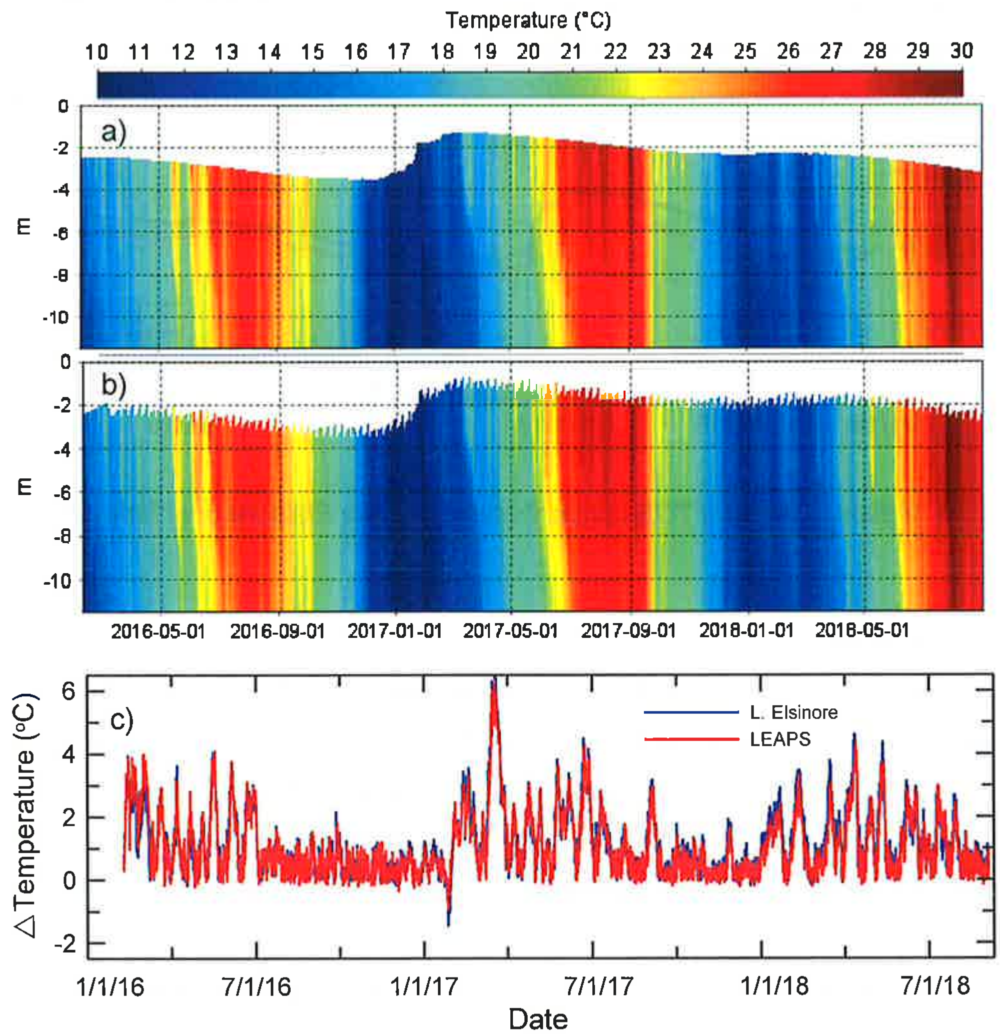

Fig. 37. Temperature in Lake Elsinore: a) profiles over time in native condition, b) with LEAPS and c) temperature difference between surface and bottom (Scenario 3).

Table 6. Statistical results for temperature difference (ΔT) between near-surface and near-bottom waters (Scenario 3).		
ΔT (°C)	L. Elsinore	LEAPS
Mean	1.11	1.06
Median	0.80	0.76
95%	3.13	3.02
Maximum	6.50	6.35
# Day >2°C	179	169

iv. DO

Dissolved oxygen concentration profiles resembled those previously described, with high concentrations in the surface, especially during the spring, and lower concentrations deeper in the water column (Fig. 38). Coinciding with periods of thermal stratification (Fig. 37), concentrations of DO in the lower water column periodically decreased to <2-3 mg/L under native conditions (Fig. 38a), while DO values were often higher there with LEAPS operation (Fig. 38b). LEAPS operation also yielded somewhat lower concentrations of DO near the surface, indicating some mixing of surface waters deeper into the water column. The consequence of this apparent mixing was that volume-averaged DO concentrations were quite similar, routinely >6 mg/L and averaged almost 8 mg/L (minimum value of 5.15 mg/L) (Fig. 38c) reflecting in part the larger volume of well-aerated water.

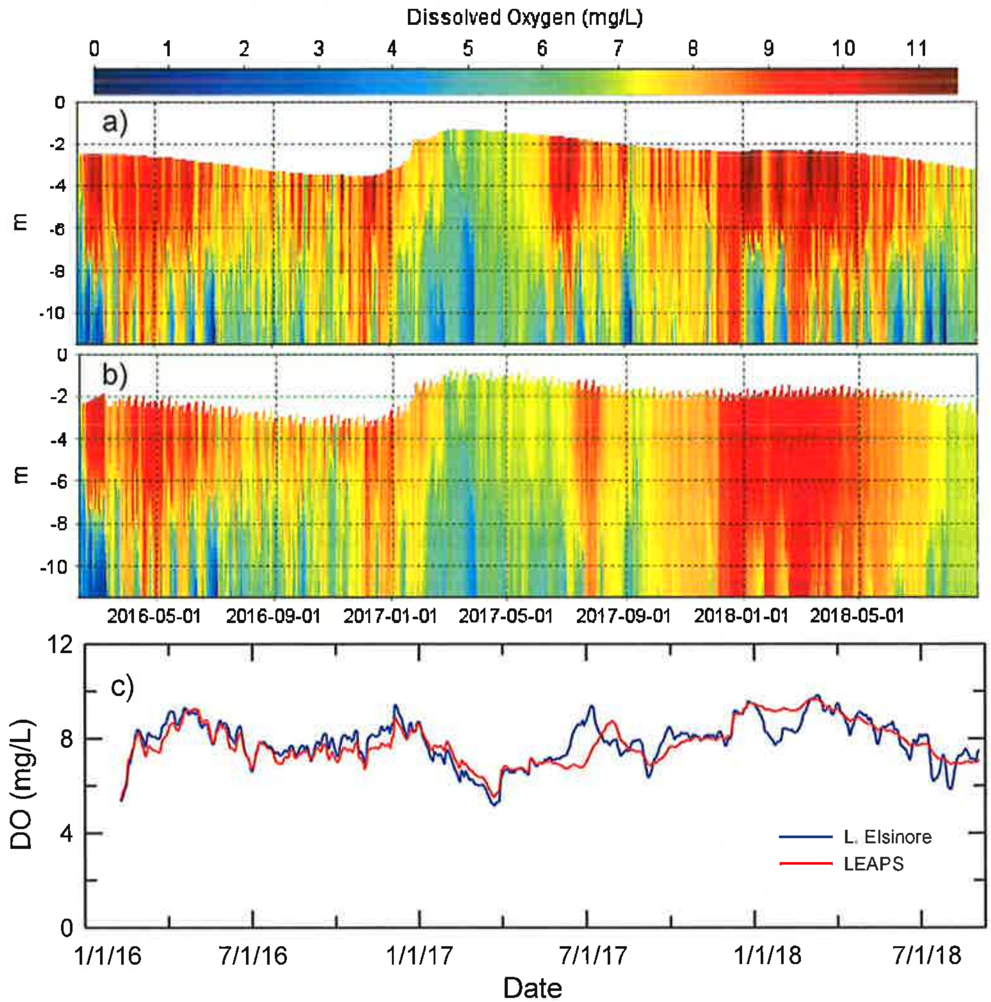

Fig. 38. Dissolved oxygen in Lake Elsinore: a) profiles over time in native condition, b) with LEAPS and c) volume-averaged DO concentrations over time (Scenario 3).

v. Nutrient Concentrations

Nutrient concentrations in the lake were smaller than those predicted in earlier scenarios at lower surface elevations. As observed with TDS, total N concentrations under the native condition varied modestly over the simulation, ranging between 1.5-2.6 mg/L (Fig. 39a). LEAPS yielded TN concentrations that were about 0.3-0.5 mg/L lower than the native condition through the first half of the simulation and that decreased further in 2017-2018. Total P concentrations ranged from 0.06-0.26 mg/L under the native condition and were predicted to reach minimum concentration in early summer 2017 before subsequently increasing over time (Fig. 39b). Operation of

LEAPS resulted in reductions in TP concentrations on the order of 0.01-0.03 mg/L through June 2017 and then increased minimally through the summer of 2018 (Fig. 39b). Concentrations of NH_4-N exhibited some seasonality, with low concentrations in the summer-fall, on the order of 0.05 mg/L, and higher levels in winter (e.g., about 0.25 mg/L in winter 2017 and nearly 0.15 mg/L in winter 2018) (Fig. 39c). Operation of LEAPS was predicted to reduce the concentration of NH_4-N in winter 2017 from 0.25 to 0.15 mg/L (Fig. 39c); changes in nutrient concentrations in the latter part of the simulation appear to be due to some ability of LEAPS to distribute DO through the water column, suppress PO_4-P release and increase nitrification-denitrification reactions.

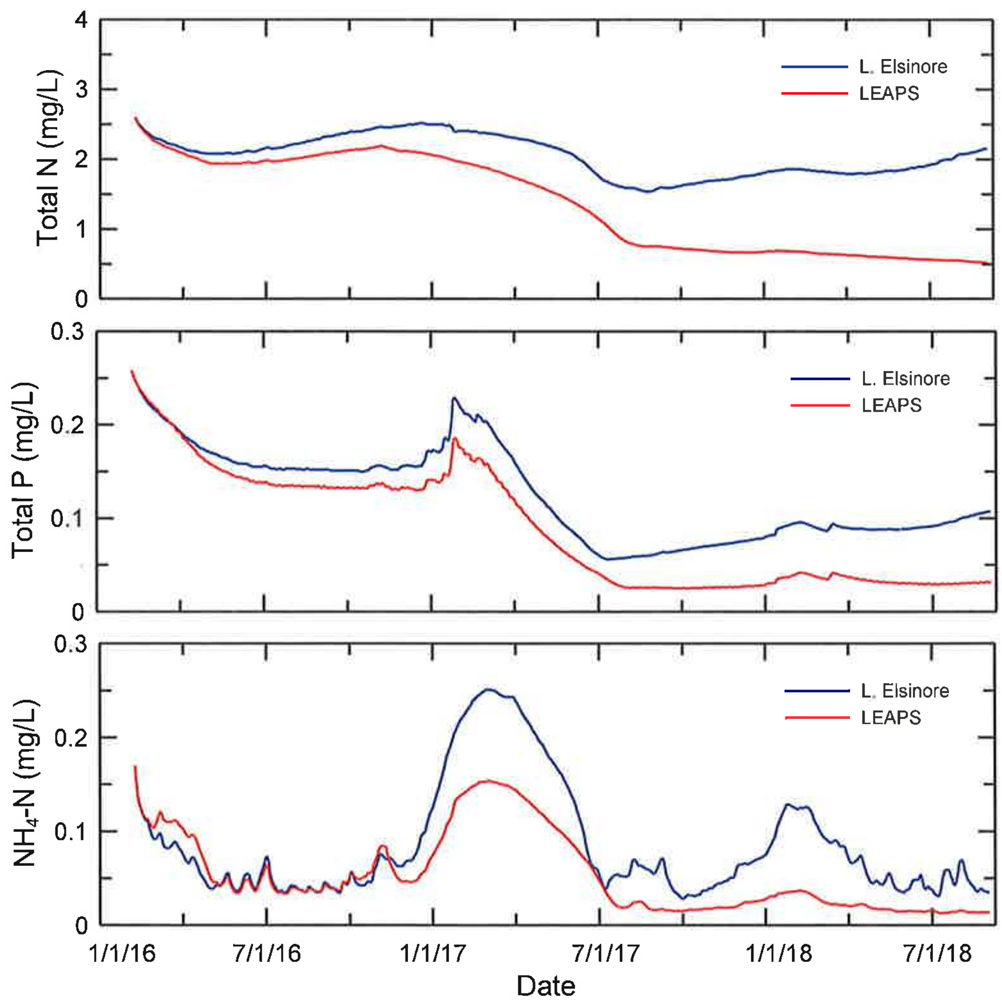

Fig. 39. Volume-averaged nutrient concentrations in Lake Elsinore over time: a) total N, b) total P and c) NH_4-N (Scenario 3).

vi. Chlorophyll a

The lower concentrations of nutrients in the lake were predicted to yield lower concentrations of chlorophyll a, reaching a maximum value of about 120 µg/L in summer 2016; this compares with values over 300 and 180 µg/L at this same time at lower initial lake levels (1235 and 1240 ft, respectively) (and higher nutrient concentrations). Operation of LEAPS had minimal impact on chlorophyll a in 2016, but operation was predicted to have a greater effect in 2017-2018, lowering predicted concentrations to 25-30 µg/L compared with concentrations reaching about 100 µg/L by August 2018 for the lake in its native condition (Fig. 40).

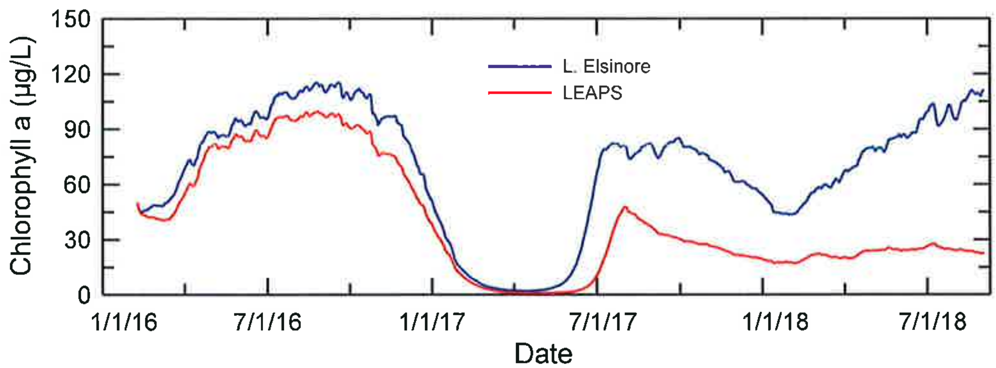

Fig. 40. Volume-weighted chlorophyll a concentrations over time (Scenario 3).

vii. Microcystin

Microcystin concentrations were predicted to be much lower than found in Scenarios 1 and 2, reaching only about 1.2-1.6 µg/L in summer 2016 and 2018, while being absent in 2017 (Fig. 41). Supplementation with SWP water and operation of LEAPS was predicted to further lower concentrations, with levels reaching a maximum of only about 0.2 µg/L in September 2016 and being essentially absent for the rest of the simulation (Fig. 41).

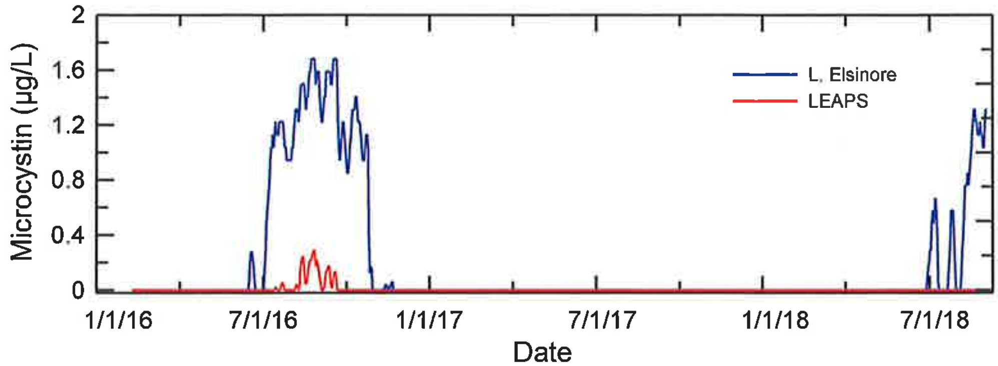

Fig. 41. Volume-weighted microcystin concentrations over time (Scenario 3).

Objective 2. Impacts of Transient Storage in Upper Reservoir on Water Quality

The data presented above in objective 1 focused on the influence of LEAPS on water quality in Lake Elsinore; this section considers the water quality and water column conditions in the Upper Reservoir and specifically addresses questions related to the effects of pumping of water from Lake Elsinore to the Upper Reservoir, transient storage in the Upper reservoir, and return of water to Lake Elsinore during hydropower generation. For this evaluation, the low lake level scenario (Scenario 1) was considered since it represents the poorest water quality when impacts of transfer and transient storage in the Upper Reservoir would be expected to be most evident. Simulations were conducted assuming the 50 m wide I/O and nighttime pumping/daytime generation (Schedule 1). As previously noted, the Upper Reservoir represents an additional surface from which water will evaporate, thus increasing somewhat evaporative losses and altering the water budget for the lake. Additional SWP water in the amount of approximately 300 af per year was added in the simulations to compensate for this additional evaporation to reflect water supply agreements associated with LEAPS.

The general water column conditions in the Upper Reservoir reflect to varying degrees the conditions in Lake Elsinore. This is unsurprising given the direct hydraulic linkage between the two water bodies and, except for slight differences in local air temperature and relative humidity, equivalent meteorological conditions. For example, under the nighttime pumping/daytime hydropower generation schedule, water column temperatures in the two water bodies track each other closely (Fig. 42). Thus, seasonal warming to maximum values near 30°C in the summer and cooling to winter minimum values near 10°C were witnessed in both water bodies. Total dissolved solids concentrations also tracked each other very closely (data not shown). Differences in lake level, both on the seasonal scale and shorter term due to withdrawal and return flows associated with pump-generation cycles, are the most notable differences between Lake Elsinore and the Upper Reservoir when considering basic water column conditions (level, temperature, TDS) (Fig. 42).

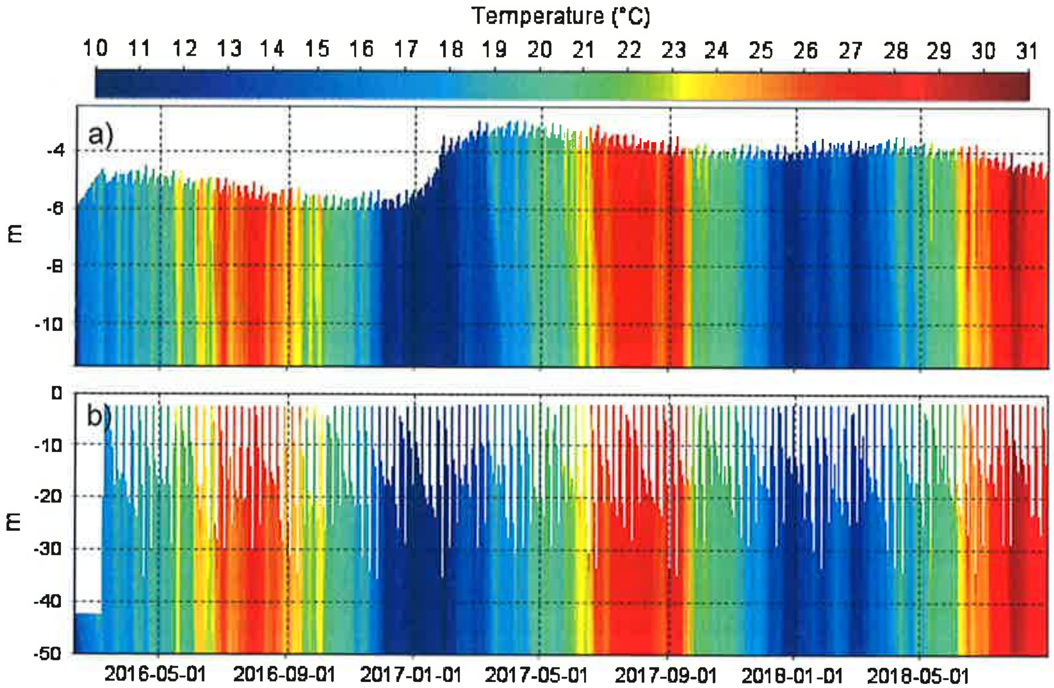

Fig. 42 LEAPS temperature profiles over time comparing the Upper Reservoir with Lake Elsinore: a) Lake Elsinore and b) Upper Reservoir with nighttime pumping/daytime generation and 50 m I/O (Scenario 1).

The conformance between the two water bodies is weaker when considering concentrations of DO (Fig. 43). Lake Elsinore often exhibits marked gradient in DO with depth, with high concentrations near the surface due to photosynthetic production of O_2 and much lower concentrations deeper in the water column resulting from large net oxygen demand (Fig. 43a). In comparison, DO concentrations were generally somewhat lower and more uniform vertically in the Upper Reservoir, although seasonal trends of higher DO levels in the spring and lower concentrations in the summer were present in both water bodies (Fig. 43b).

The differences in DO concentrations and profiles can be attributed to several factors. First of all, while pumping-generation transfers only a small proportion of the total volume of Lake Elsinore each day (even at the very low lake levels present in 2016, on the order of 7-10% per day), a much larger volume of the Upper Reservoir is exchanged (about 50% per day under the nighttime pumping-daytime hydropower generation schedule).

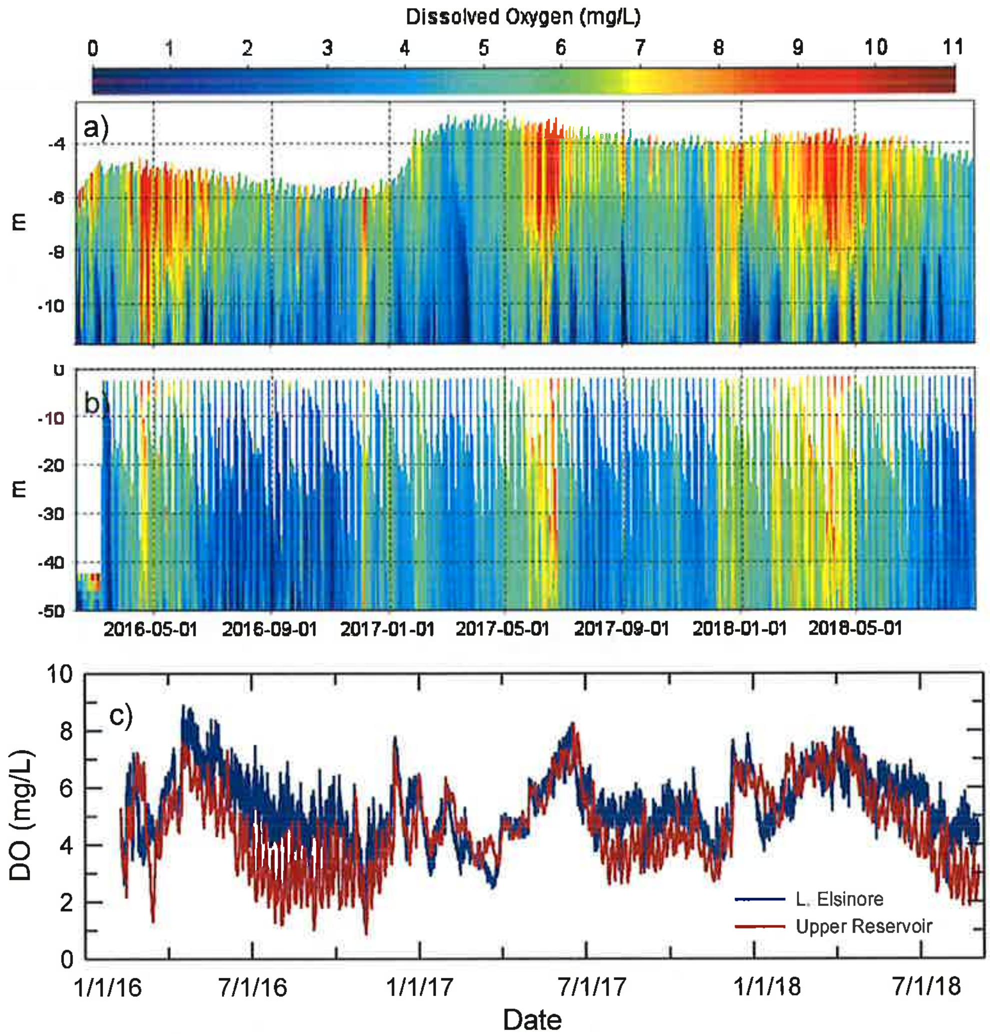

Fig. 43. LEAPS DO profiles over time comparing the Upper Reservoir with Lake Elsinore: a) Lake Elsinore, b) Upper Reservoir, and c) volume-weighted DO concentrations (Scenario 1).

Thus the Upper Reservoir is strongly affected on a daily basis during operation, with the properties of the incoming water (temperature, DO, etc.) and the large TKE input associated with very high flow rates into a small volume dominating conditions in the Upper Reservoir. The large TKE input into the small volume of the Upper Reservoir during pumping ensures relatively well-mixed conditions. A second factor affecting DO concentrations in the Upper Reservoir has to do with withdrawal depth from Lake Elsinore. Although the I/O is located near shore, excavation provides deeper water there (Fig. 1); thus water withdrawn from the lake includes the lower DO water near deeper bottom sediments. High DO surface water is mixed with lower DO and transferred to the Upper Reservoir during pumping. The timing of the delivery of water to the Upper Reservoir also plays a role, with pumping at night transferring water from Lake Elsinore that is lower in DO when compared with daytime concentrations during active photosynthesis. A final factor is the considerable depth of the Upper Reservoir, up to almost 50 m, in which only a very small volume near the surface would comprise the photic zone capable of net O_2 production during photosynthesis, while most of the water column, even when drawn down, would be subject to high rates of net algal respiration. The consequence of all these factors is a typically lower DO concentration in the Upper Reservoir when compared with Lake Elsinore. This will be considered further later in this section.

The concentrations of nutrients and chlorophyll a in the Upper Reservoir tended to track quite closely their concentrations in Lake Elsinore (Fig. 44) due to the large and rapid volume exchange within the Upper Reservoir during LEAPS operation. Total N and NH_4-N concentrations in the two water bodies were nearly always very similar, while some difference were noted for total P during winter 2017 due to presumably to particulate-P settling. Chlorophyll a and microcystin concentrations were slightly lower in the Upper Reservoir when compared with Lake Elsinore throughout the simulation as well (Fig. 45).

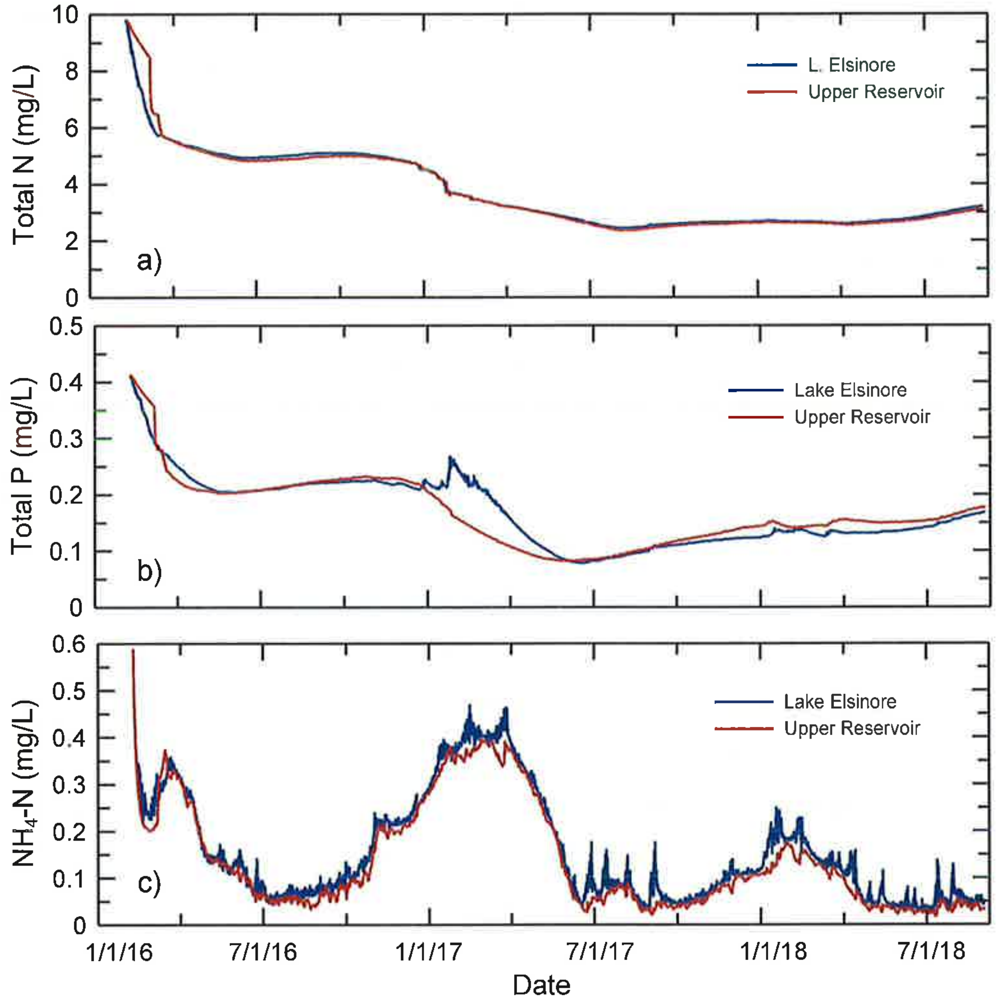

Fig. 44. Volume-weighted concentrations over time comparing the Upper Reservoir with Lake Elsinore: a) total N, b) total P and c) NH_4-N (Scenario 1).

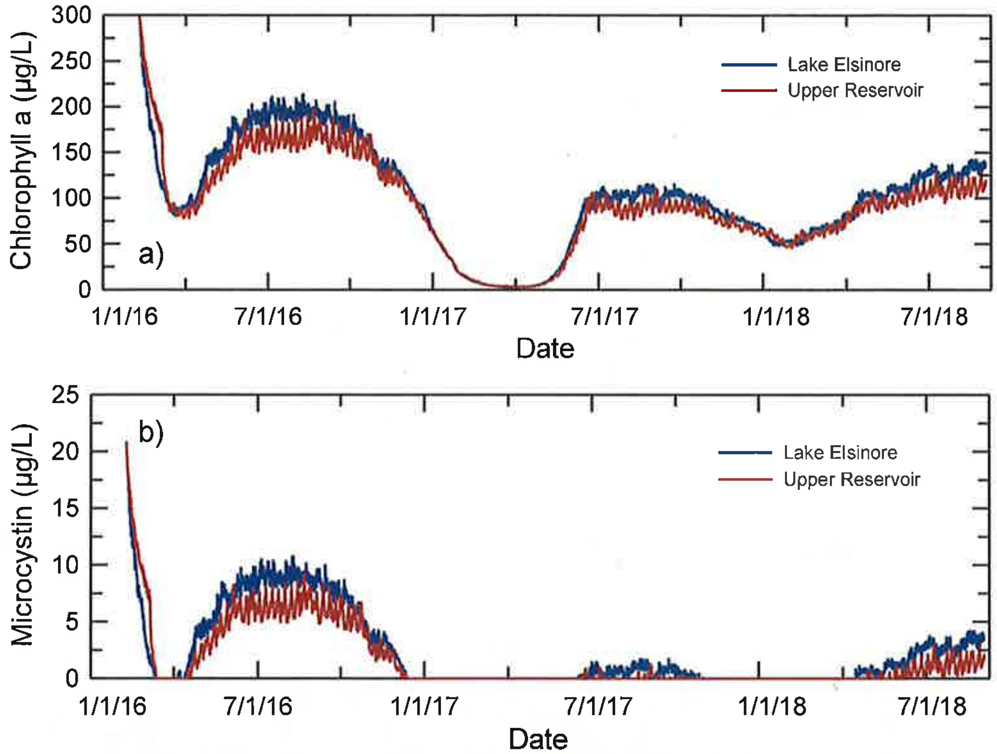

Fig. 45. Volume-weighted concentrations over time comparing the Upper Reservoir with Lake Elsinore: a) chlorophyll a and b) microcystin (Scenario 1).

The retention time (or residence time) of water provides a convenient way to quantify the frequency and extent of volumetric exchange between the two water bodies. Here the retention time is set at 0 at the start of the simulation and effectively counts the time in which a parcel of water remains in Lake Elsinore (Fig. 46a) and in the Upper Reservoir (Fig. 46b). Each time water is added, that water enters with a residence time of 0 so dilutes water already present in the water body. For reference, the basin-wide retention time in Lake Elsinore without LEAPS ranged from 0 at the start of the simulation on Feb. 9, 2016, increased to almost 300 days by the end of December 2016, decreased to about 200 days following winter 2017 inflows, and reached 667 days at the end of the simulation in August 2018. Operation of LEAPS dramatically shortened the residence time at site E2 in Lake Elsinore to a maximum value of only about 40 days for some bottom water near the deepest part of the lake following the increase in lake volume and reduction in retention time with the runoff inflows of winter of 2017.

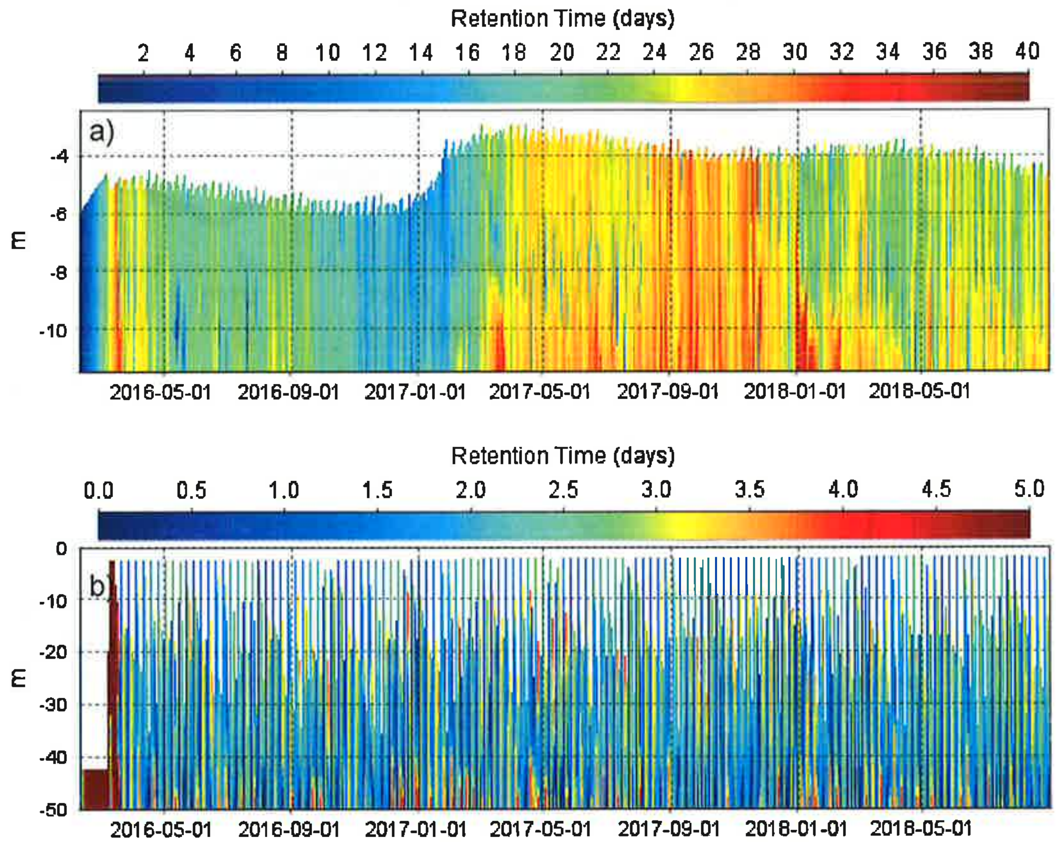

Fig. 46. Retention time over time comparing the Upper Reservoir with Lake Elsinore: a) Lake Elsinore and b) Upper Reservoir (Scenario 1).

The retention times in the Upper Reservoir were reduced another order of magnitude to values of 0.5 – 5 days (Fig. 46b) following the initial fill in March 2016 (the model required some water be present in the Upper Reservoir at the start of filling to properly simulate hydrodynamics). Thus, it is clear that under a regular weekday pump-generation schedule that the water in the Upper Reservoir will remain there for only a very short period of time.

Effects of Duration of Storage

The retention time and the water column conditions in the Upper Reservoir are thus dependent upon the frequency and duration of hydropower production. Recognizing that operation of LEAPS 5 days a week may not necessarily always be the case, a simulation explored conditions with increasing intervals of non-operation.

For this simulation, initial conditions were set at those predicted for July 3, 2016 and evaluated a week of regular schedule with pump-generation 5 days a week and 2 days (weekend) with no operation, then 5 days a week operation with 1- and 2-weeks non-operation, although 1-2 weeks without operation is highly unlikely (Fig. 47a).

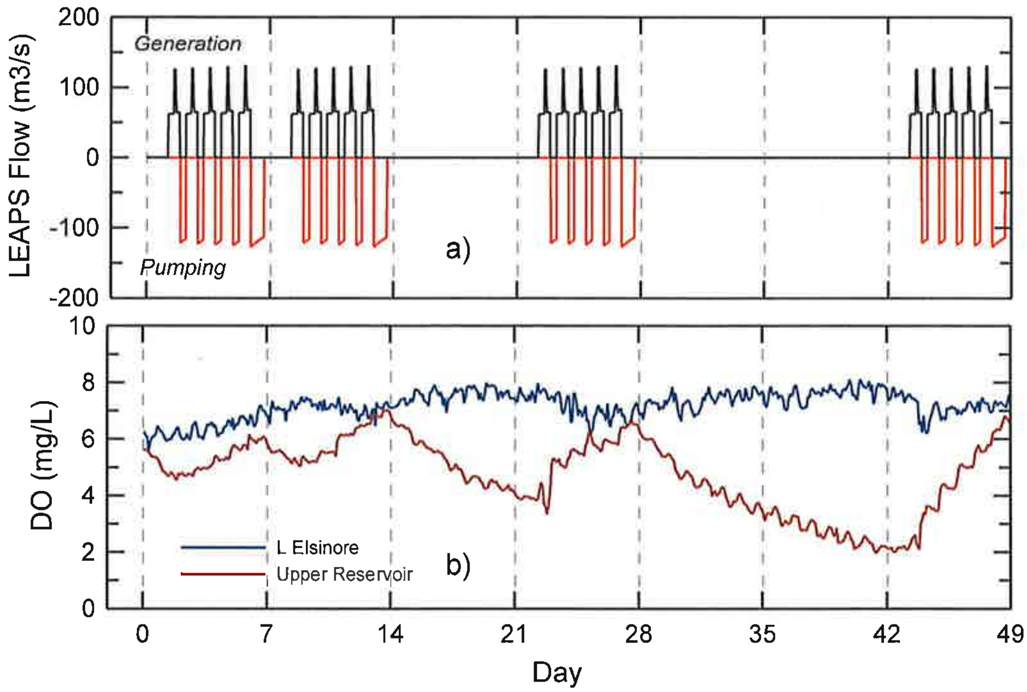

Fig. 47. Effect of hydropower production schedule: a) operational schedule with increasing storage time in the Upper Reservoir, and b) depth-averaged DO concentrations over time comparing Lake Elsinore and the Upper Reservoir.

The depth-averaged concentrations of DO in the water column of the Upper Reservoir varied strongly with the duration of storage, with relatively high and increasing DO levels under regular schedule with daily pumping-generation during the week, and reductions on the order of 0.4-0.5 mg/L per day during periods of weekend storage (Fig. 47b). Concentrations of DO dropped about 1 mg/L during about 2 day (weekend) storage, about 3 mg/L during storage over 8 days (from day 14-22), and 4 mg/L over 2 weeks without operation (Fig. 47b). Delivery of water to Lake Elsinore after storage of 1-2 weeks in the Upper Reservoir yielded reductions of 1-2 mg/L in volume-averaged DO concentrations, with further transient reductions near the I/O following restart of hydropower generation, although average concentrations in Lake Elsinore remained high throughout the simulation (Fig. 47b).

Short-term oscillations in average DO concentrations in both Lake Elsinore and in the Upper Reservoir resulted from daytime photosynthetic production of O_2 and nighttime respiration. The vertical trends within the water column of the Upper Reservoir indicate daytime surface DO production but deletion of DO in most of the water column (Fig. 48). Biological oxygen demand in the Upper Reservoir over this period ranged from 13-16 mg/L resulting from high rates of algal respiration.

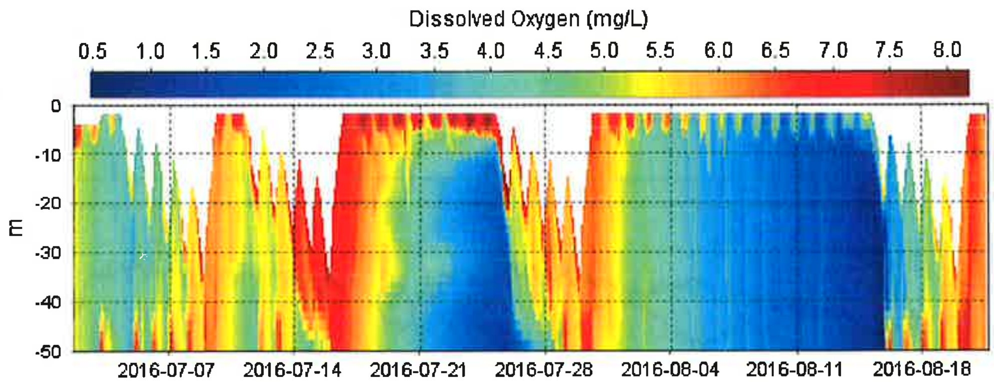

Fig. 48. DO concentration profiles over time in the Upper Reservoir with increasing intervals of storage.

Transferring algae from a shallow relatively well-mixed water column (Lake Elsinore) to the much deeper Upper Reservoir lowers the overall availability of photosynthetically-available radiation (PAR) to phytoplankton and reduces vertical mixing, leading to marked declines in predicted chlorophyll levels over time (Fig. 49).

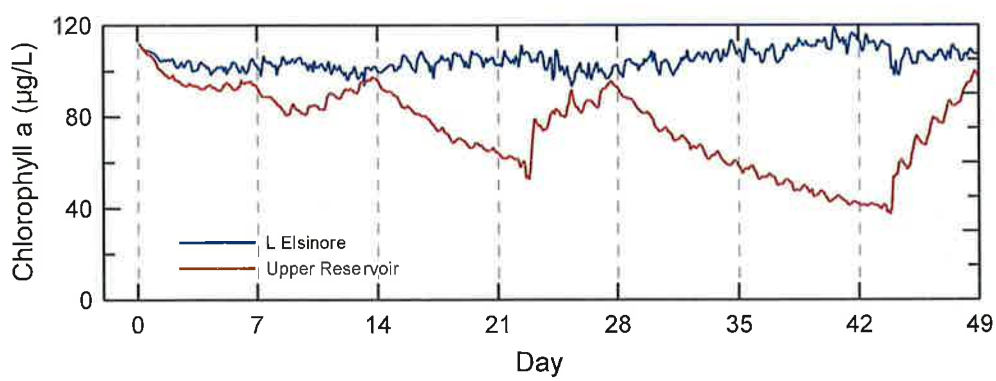

Fig. 49. Comparison of chlorophyll a concentrations over time in the Upper Reservoir and Lake Elsinore with increasing intervals of storage.

The delivery of this water after 2 weeks storage in the Upper Reservoir reduced volume-averaged chlorophyll a concentrations in Lake Elsinore by about 10 µg/L (Fig. 49); more substantial reductions were predicted near the I/O during restart of hydropower generation as the stored water enters the lake.

The average concentrations of total N and total P in the Upper Reservoir followed relatively closely the concentrations in Lake Elsinore and were less strongly influenced by duration of storage in the Upper Reservoir than chlorophyll a or DO, with concentrations generally slightly higher than present in Lake Elsinore (Fig. 50).

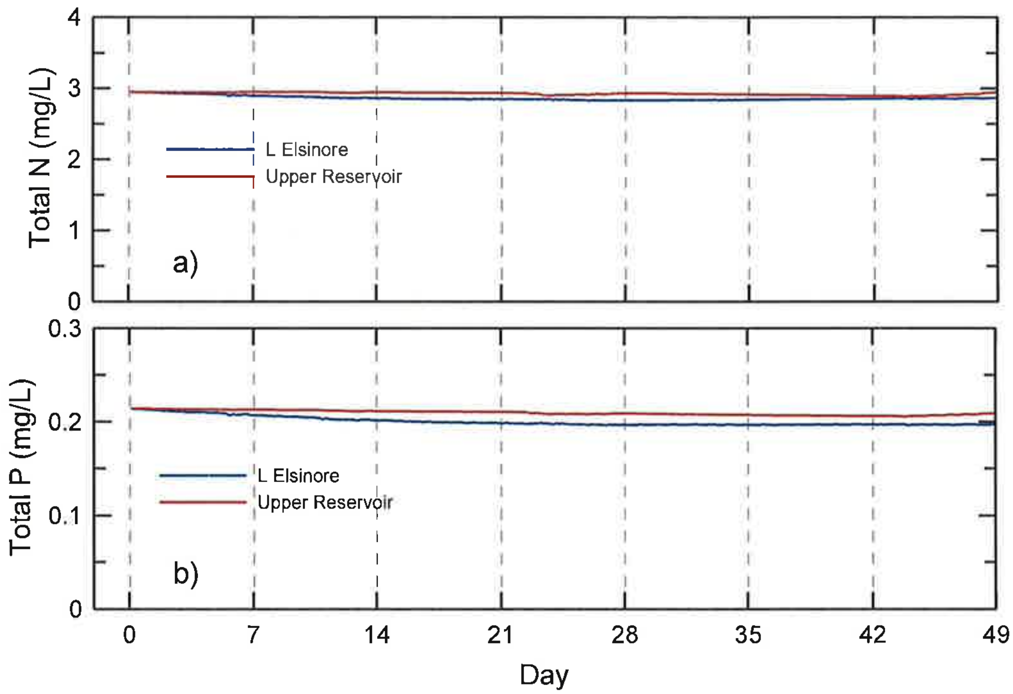

Fig. 50. Comparison of nutrient concentrations in the Upper Reservoir and Lake Elsinore with increasing intervals of storage: a) total N and b) total P (note scales).

As indicated by trends in chlorophyll a, the greater depth of the Upper Reservoir limiting the relative volume of the photic zone combined with duration of storage hinders phytoplankton growth and reproduction and increases settling and senescence. Phytoplankton senescence and death results in cell lysis and excretion of organic N and P compounds that can undergo deamination and dephosphorylation reactions; the consequence of this is an increase in NH_4-N and PO_4-P concentrations in the Upper Reservoir, with the magnitude of increase governed by the duration of storage (Fig. 51). These increases are quickly reversed when LEAPS is operated however (Fig. 47a). The delivery of this water to Lake Elsinore had only subtle effects on NH_4-N and PO_4-P levels, with concentrations edging up minimally (Fig. 51, note scales). The magnitude of changes in overall water quality in Lake Elsinore reflect the volumes of water exchanged; at the low lake levels in place during this simulation (about 1235.3 ft), daily operation of LEAPS would exchange not more than about 10% of the volume of Lake Elsinore (less at higher lake levels), with some re-entrainment and recirculation of water between Lake Elsinore and the Upper Reservoir.

Using the algal toxin subroutine in AEM3D, microcystin concentrations were predicted to increase from an initial assumed concentration of 1 µg/L to about 5 µg/L in Lake Elsinore and about 4 µg/L in the Upper Reservoir reflecting rapid approach to approximate steady-state (Fig. 52). This indicates a modest reduction in microcystin concentrations resulting from storage in the Upper Reservoir (Fig. 52). Concentrations of microcystin decreased slightly in Lake Elsinore following pumping-generation while levels increased slightly in the Upper Reservoir as higher concentration waters from Lake Elsinore were pumped up (e.g., between days 22-28 and 43-49) (Fig. 47a).

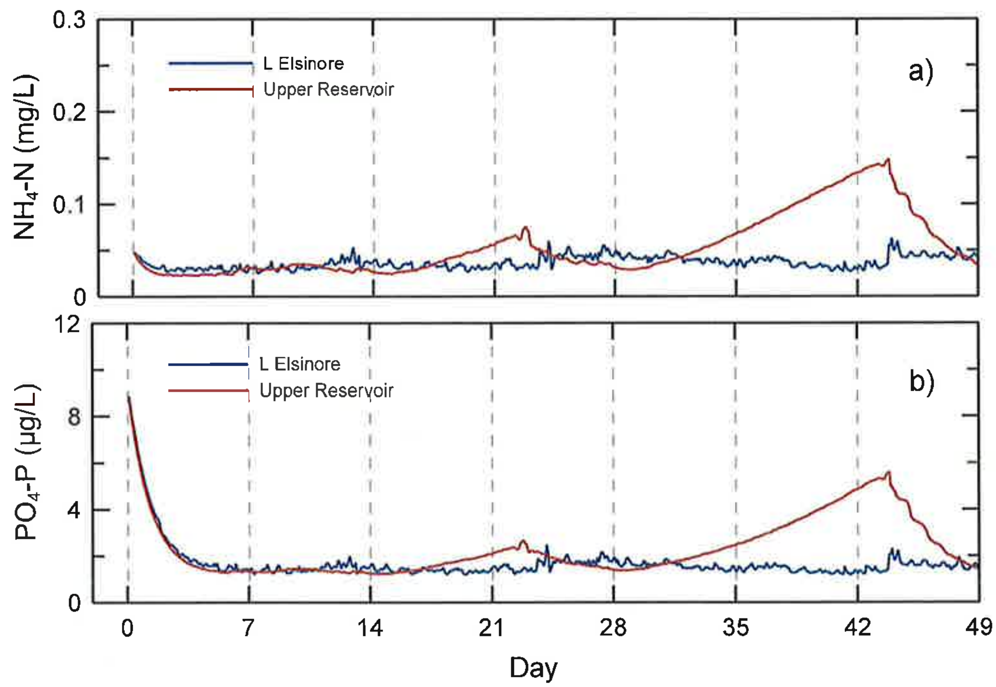

Fig. 51. Comparison of nutrient concentrations in the Upper Reservoir and Lake Elsinore with increasing intervals of storage: a) NH_4-N, and b) PO_4- P (note scale, here in µg/L).

Microcystin levels predicted from regression with chlorophyll a concentrations would yield much lower values than the AEM3D subroutine, which is not thought to be appropriate given the importance of cell senescence and lysis which are included in AEM3D. Microcystin concentrations are considered further in the Discussion and Conclusions section later in this report.

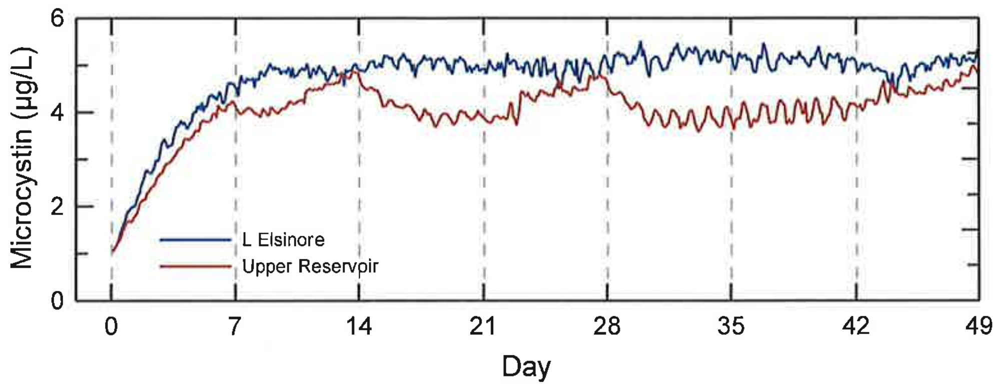

Fig. 52. Comparison of microcystin concentrations in the Upper Reservoir and Lake Elsinore with increasing intervals of storage.

Objective 3. Harnessing LEAPS to Improve Water Quality

The final objective of this study was to assess how LEAPS may be able to improve water quality in Lake Elsinore compared with current conditions. The focus of the initial design and operational plan of LEAPS was to minimize the negative impacts of its operation on conditions, use and water quality at Lake Elsinore. As previously noted, considerable effort has been expended over the past 15 years to enhance mixing and distribute DO throughout the water column to reduce fish kills and improve water quality. Despite these efforts, intervals of low DO and corresponding fish kills have continued to periodically recur. The design, installation and operation of LEAPS presents an opportunity to potentially enhance conditions in Lake Elsinore, e.g., increase mixing, improve DO levels, reduce fish kills, lower internal recycling of phosphorus from bottom sediments and reduce chlorophyll a concentrations. Temperature, nutrient availability and mixing regime are all known to affect blue-green algae abundance and production of microcystin and other algal toxins (Visser et al., 1996; Buford, 2006, Walls et al., 2018).

The supplementation of Lake Elsinore with high quality SWP water has been shown to substantially benefit the lake. Narrowing the width of the I/O from approximately 150 m in the original LEAPS design to about 50 m was found to improve slightly the mixing and distribution of DO. Nonetheless, hypoxic or anoxic conditions were often predicted to continue near bottom sediments and periodically throughout much of the water column. Maintenance of adequate DO in the lake is considered a key water quality objective and was the primary basis for installation of the axial flow pumps and diffused aeration systems. While both systems provide mixing energy to help distribute DO throughout the water column, they often struggle to maintain DO during periods of limited natural mixing and intense oxygen demand, e.g., as witnessed in 2016.

The operation of LEAPS introduces additional mixing energy to the lake and provides, in partnership with LESWJA and stakeholders in the watershed, an opportunity to increase DO levels in Lake Elsinore by the addition of oxygen into the return flows from the Upper Reservoir during hydropower generation. This was evaluated for the period 2016-2018 when the lake was at extremely low surface elevations, chlorophyll a concentrations exceeded 300 µg/L, predicted BOD concentrations reached 60 mg/L, and chronically low DO concentrations were present. For this analysis, the DO concentration in water delivered to Lake Elsinore during hydropower generation was increased from that at the Upper Reservoir I/O to 10 mg/L, e.g., by injection of liquid oxygen. The amount of O_2 needed to reach 10 mg/L will vary depending upon background concentrations of DO; at a nominal concentration near 5 mg/L, about 20,000 kg O_2 per day would be needed. This compares with, e.g., 7,260 kg per day delivered to Comanche Reservoir, so a system on the order of 3x the capacity of Comanche Reservoir would likely be needed. The O_2 could be produced on-site with industrial-scale pressure-swing gas adsorption or with regular delivery of liquid O_2. Installation of aerating turbines may alternatively be considered.

The concentrations of DO directly above the sediments under natural conditions and with LEAPS operation with augmentation to flows to 10 mg/L DO are illustrated in Fig. 53. Focusing first on the left series of panels that depict Lake Elsinore under native conditions (i.e., without LEAPS, but with axial flow pumps and diffused aeration systems operating), one notes often high concentrations of bottom DO in shallow water near the lake margins, but low DO concentrations often <2-3 mg/L near the middle of the lake. This condition was present throughout much of the year, including summer and fall 2016, winter 2017 and summer 2018 as well (Fig. 53). With LEAPS operation+O_2 injection, consistently higher bottom DO concentrations were predicted (Fig. 53, right-hand side). The plume of 10 mg/L water can be seen particularly clearly in the July 2018 snapshot with the DO plume extending >1300 m to the north of the I/O and covering an area over 200 acres (Fig. 53). In addition, nearly all of the lake bottom possessed DO concentrations >5 mg/L.

The volume-averaged DO concentration in Lake Elsinore over time was significantly higher with LEAPS+O_2 injection when compared native concentrations or with normal LEAPS operation (Fig. 54). The mean volume-averaged DO concentration increased from 5.49 mg/L under native conditions to 5.63 mg/L with LEAPS and 6.99 mg/L with LEAPS+O_2, while minimum volume-averaged DO concentrations increased from 1.44 to 2.62 and 4.28 mg/L, respectively (Table 7).

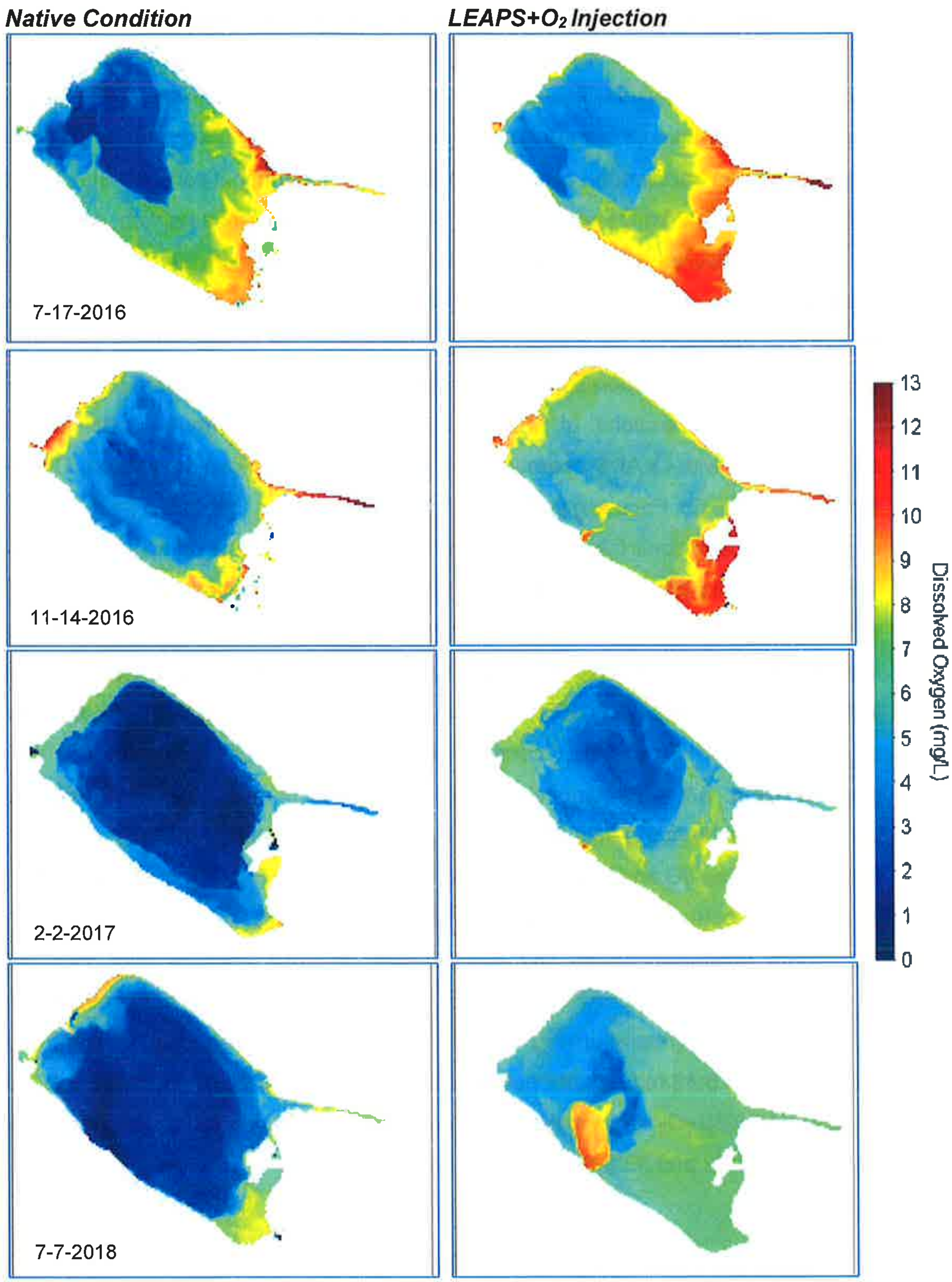

Fig. 53. Distribution of DO above bottom sediments: left=native; right=LEAPS+O_2.

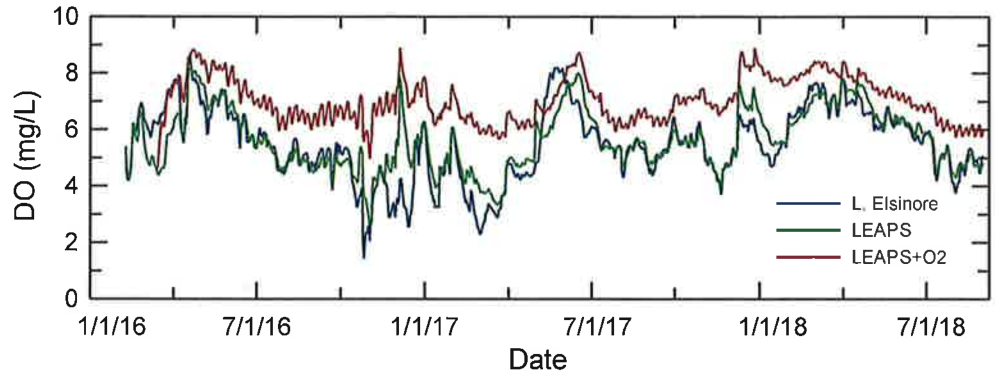

Fig. 54. *Effect of LEAPS with supplemental O_2 on volume-weighted DO concentrations (Scenario 1).*

Table 7. Summary of water quality for Lake Elsinore and with LEAPS and LEAPS with O_2 supplementation (Scenario 1)

Property	Value	L Elsinore	LEAPS	LEAPS+O_2
DO	Mean	5.49	5.63	6.99
	Median	5.49	5.49	6.9
	Minimum	1.44	2.62	4.28
	Maximum	8.62	8.18	8.89
Total P	Mean	0.26	0.15	0.12
	Median	0.24	0.13	0.08
	Minimum	0.15	0.06	0.05
	Maximum	0.43	0.38	0.38
Total N	Mean	5.45	3.4	2.99
	Median	5.49	2.84	2.91
	Minimum	1.44	2.09	1.15
	Maximum	8.62	9.5	9.1
Chlorophyll a	Mean	173	104	85.2
	Median	170	105	65.9
	Minimum	6.2	3.1	2.8
	Maximum	312	280	280
Microcystin	Mean	8.0	2.4	1.8
	Median	7.0	0.8	0
	Minimum	0	0	0
	Maximum	19.8	16.9	16.9

The number of days the volume-average DO was <5 mg/L in the lake decreased from 311 (33%) to 17 (2%) with LEAPS+O_2 (with all but 1 of those days during initial filling of the lake and prior to LEAPS operation). Fig. 55 reveals the improvements in DO throughout the water column near the deepest water in the center of the lake (TMDL site E2) and elsewhere resulting from injection of O_2 in generation flows. Intervals of complete hypoxia or anoxia that were evident in fall 2016 and winter 2017 were eliminated or, at the least, reduced to the lowermost 1-2 m directly above the bottom sediments (Fig. 55).

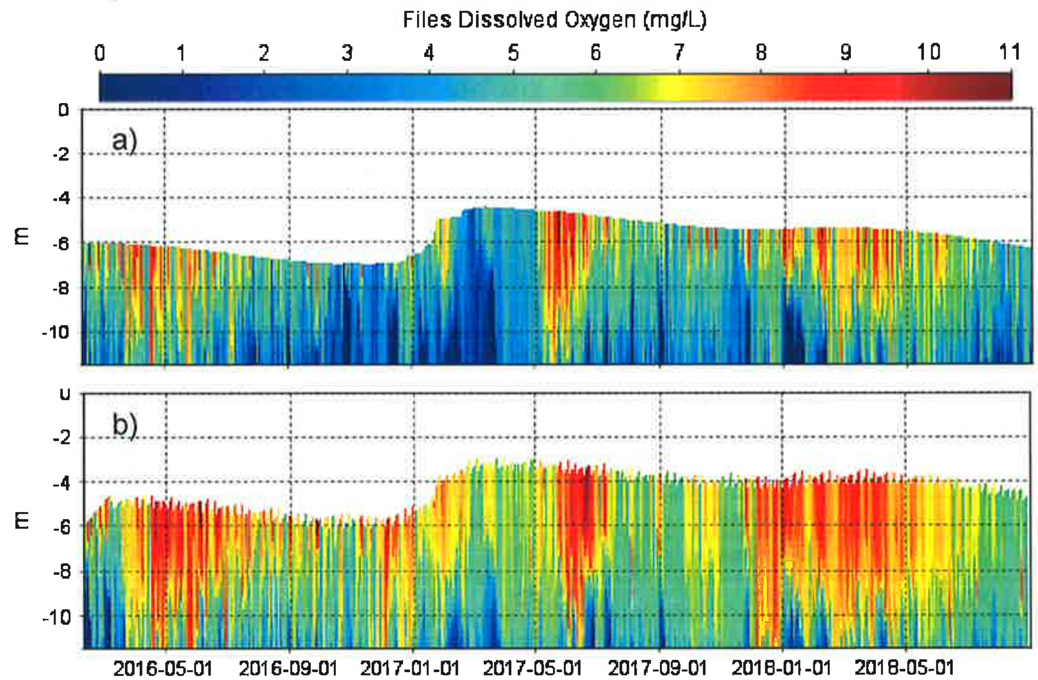

Fig. 55. Effect of LEAPS with O_2 supplementation on DO concentration profiles in Lake Elsinore: a) native condition; b) with LEAPS with O_2 supplementation.

The increase in DO concentrations was also predicted to reduce nutrient and chlorophyll a concentrations in the latter half of the simulation (Fig. 56). Increased DO concentrations above bottom sediments (Fig. 53) would reduce the flux of PO_4-P by increasing sorption to ferric oxyhydroxides and/or slowing the reductive dissolution of Fe(OH)HPO_4-type solid phases within the sediments and favorably alter the N cycle; it appears that it took about 16 months for effects on the inventory of phosphorus and nitrogen.

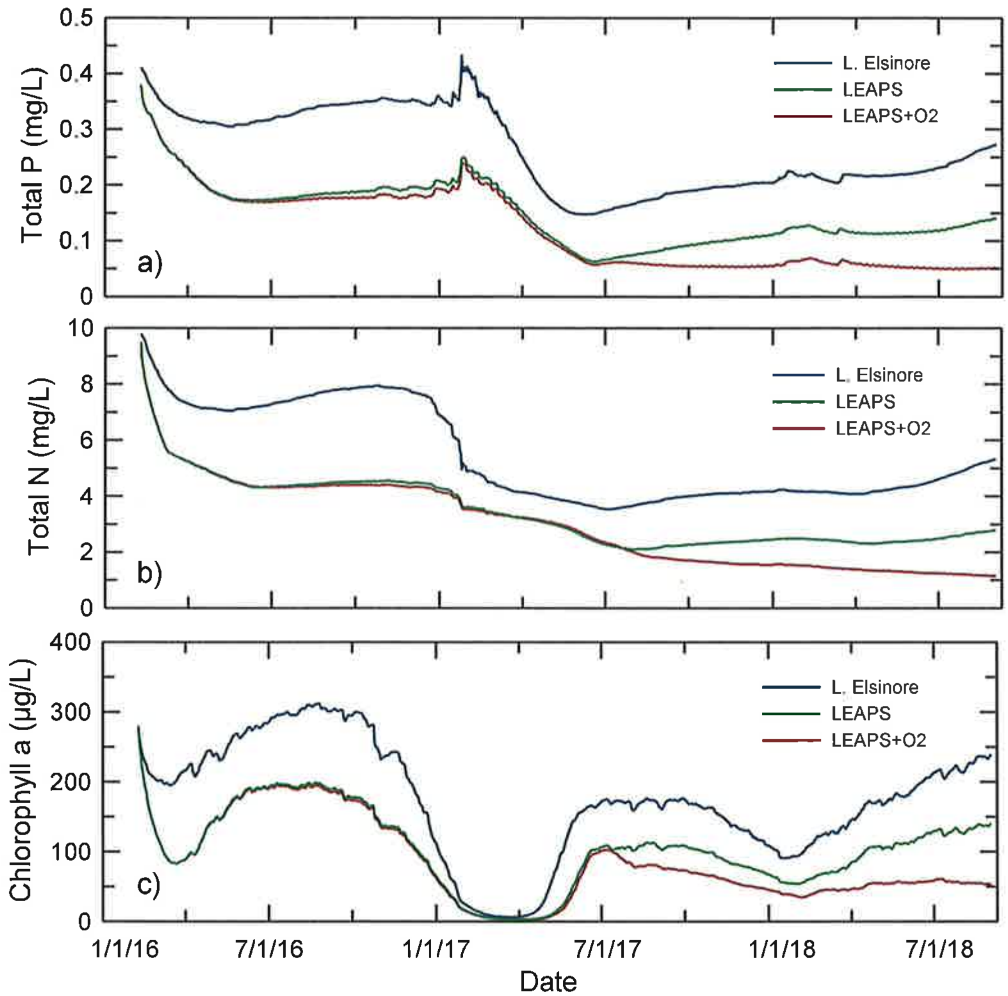

Fig. 56. Effects of LEAPS with O_2 supplementation on volume-weighted concentrations of: a) total P, b) total N, and c) chlorophyll a.

The increased DO concentrations also shifted the N cycle and lowered total N concentrations in the latter part of the simulation, when the lake elevation was significantly higher due to winter 2017 runoff, compared with LEAPS alone. The lowered total P and total N also yielded lower chlorophyll a and microcystin concentrations (Fig. 56, Table 7). Operation of LEAPS with the addition of O_2 into returns flows during hydropower generation lowered mean volume-averaged total P and chlorophyll a concentrations by 50% compared with native concentrations in the lake and represented further improvement in water quality beyond the predicted reductions achieved by LEAPS with SWP supplementation alone (Table 7). (The modest augmentation of the lake with SWP water to balance evaporative losses from the Upper Reservoir incorporated into LEAPS simulations are not of sufficient magnitude to meaningfully shift water quality.) Beyond reducing the occurrence and magnitude of fish kills, increasing DO concentrations in the lake has the prospect for favorably altering the longer-term nutrient cycling and biogeochemistry in the lake.

DISCUSSION AND CONCLUSIONS

Lake Elsinore is an important recreational and ecological resource for the region. Significant strides have been taken to improve water quality in the lake, including implementation of numerous best-management practices (BMPs) in the watershed, addition of recycled water to help maintain lake level, installation of axial flow pumps and a diffused aeration system, carp removal and stocking of piscivorous sport fish. Challenges remain however, as evidenced by the extremely poor conditions over the past several years, including very low lake levels, very high concentrations of TDS, nutrients, chlorophyll and algal toxins, and extended intervals of hypoxia or anoxia. Arguably the fundamental challenge confronting the overall health and viability of the lake is the availability of water sufficient to (i) meet evaporative losses that reach nearly 1.5 m per year and (ii) to periodically flush salts and nutrients out of the system. Long-term simulations previously confirmed historical observations of the lake going completely dry during periods of extended drought. The regular addition of recycled water has favorably shifted the water balance for the lake, although current deliveries of about 5,000 af per year fall short of total evaporative losses up to 15,000 af per year during intervals of extreme drought when annual rainfall can be as low as 2-4 inches producing negligible runoff to the lake.

Given the hydrological challenges for the lake, increased deliveries of recycled water and periodic supplementation from other sources will help maintain an adequate lake level and support all of the lake's beneficial uses, and also support operation of a pumped-storage hydroelectric plant. Current water supply agreements are in place to provide a one-time supplementation of 15,000 af of water to the lake, with about 45% of this volume used for filling of the Upper Reservoir while 55% of this volume will remain in Lake Elsinore. Water to offset increased evaporative losses from the Upper Reservoir will also be provided.

This study sought to build upon previous studies addressing potential impacts of LEAPS operation on Lake Elsinore, and specifically sought to improve understanding of the consequences for water quality of operating LEAPS at different lake levels and storage in the Upper Reservoir. Given the ongoing challenges related to water quality in the lake, the opportunity to improve water quality with LEAPS was also explored.

The operation of LEAPS with supplementation up to 15,000 af of SWP water increased the average lake level and reduced volume-weighted concentrations of TDS, nutrients, chlorophyll a and microcystin across a range of lake elevations, with the greatest relative improvement achieved at lowest initial surface elevation and poorest water quality (Table 8). The operation of LEAPS had minimal effect upon on averaged ΔT values but did increase slightly volume-averaged DO concentrations and improved to varying degrees vertical and horizontal distributions of DO. The operational schedule of LEAPS (nighttime pumping/daytime hydropower generation or morning pumping/afternoon-evening hydropower generation) and alternative widths of the lake I/O (50 m vs 150 m) were not found to significantly alter average water column conditions in Lake Elsinore. Model results support the operation of LEAPS across a lake surface elevation range of at least 1235-1253 ft.

Table 8. Global volume- and time-averaged water column properties for Lake Elsinore under native conditions and with LEAPS operation for the 3 lake level scenarios (LE = native condition, LEAPS+O_2 = O_2 injection, LEAPS-sched = daytime pumping/afternoon-evening hydropower generation schedule; LEAPS I/O = 150 m I/O width)

	Scenario 1 (1235 ft)			**Scenario 2 (1240 ft)**				**Scenario 3 (1247 ft)**	
	LE	LEAPS	LEAPS +O_2	LE	LEAPS	LEAPS sched	LEAPS I/O	LE	LEAPS
Elev	1236.3	1240.6	1240.6	1239.8	1244.1	1244.1	1244.1	1246.9	1247.9
TDS	2742	1986	1985	2364	1810	1800	1810	1317	1222
ΔT	0.51	0.68	0.67	0.58	0.57	0.52	0.58	1.11	1.06
DO	5.49	5.63	6.99	7.8	7.6	7.6	7.6	7.87	7.81
Total N	5.45	3.40	2.99	2.89	2.16	2.15	2.16	2.05	1.36
Total P	0.26	0.15	0.12	0.16	0.12	0.12	0.12	0.12	0.09
Chl a	173	104	85	105	67	67	69	68	39
MC[a]	8.0	2.4	1.8	2.4	0.4	0.5	0.4	0.2	0.0

[a]MC = microcystin

Storage of water in the Upper Reservoir was found to have variable effects on water quality that increased with increasing retention time. With regular weekday operation, water in the Upper Reservoir would have a retention time of 1-2 days and water quality followed quite closely that of Lake Elsinore, with very similar temperatures and concentrations of TDS, nutrients, chlorophyll a and microcystin. The concentrations of DO did vary somewhat however, with concentrations in the Upper Reservoir generally lower than volume-averaged concentrations in Lake Elsinore.

Increasing the duration of storage in the Upper Reservoir from 1-2 days over the weekend under a typical schedule to 1, 2 or 3 weeks yielded more significant differences, especially with respect to concentrations of chlorophyll a and DO where substantial reductions were predicted.

The transfer of algae from a shallow relatively well-mixed lake to a much deeper less-mixed reservoir changes fundamentally the net balance of photosynthesis and respiration and reduces effective reaeration rates within the water column. A much larger fraction of the lake volume in the Upper Reservoir would be below the light-compensation level, and algae without sufficient light would slowly senesce and die. Moreover, the much smaller fetch and deeper water column would result in less wind-mixing, so algae could more settle more quickly during periods of non-operation. The model predicted reductions in chlorophyll a by about 50% within 2 weeks of storage. At the same time, loss of photosynthetic capability and high respiratory demands quickly depleted DO concentrations. With DO loss rates near 0.5 mg/L/d, a retention time of 2 weeks in the Upper Reservoir without cycling was predicted to lower the depth-averaged DO concentration from near 7 mg/L to almost 2 mg/L with strong anoxia throughout most of the water column. The altered light and mixing regime in the Upper Reservoir relative to Lake Elsinore also increased predicted NH_4-N and PO_4-P concentrations and lowered slightly microcystin levels at longer storage times. Transient storage on water quality in the Upper Reservoir thus provided some benefits (reductions in chlorophyll a and modest apparent reductions in microcystin and total nutrient concentrations) but also lowered DO concentrations and increased concentrations of NH_4-N and PO_4-P in stored water. At the same time, volume-averaged concentrations of nutrients, DO and microcystin in Lake Elsinore were only very modestly changed following flow of this water during restart of hydropower generation due to the small volume compared with that present in the lake.

The physical transfer of water from the Upper Reservoir to Lake Elsinore during hydropower generation will not alter total concentrations of nutrients, but algal cells and extracellular microcystin concentrations may be influenced by pressure and shear forces generated during hydropower production. While the model cannot explicitly simulate such processes, review of limited available data indicates that hydropower production at other facilities has not generally been found to increase microcystin concentrations. Evidence for this can be found in the Iron Gate Reservoir on the

Klamath River near the Oregon-California border. Microcystin concentrations in samples collected between July-October 2005-2014 were typically somewhat lower in the Klamath River immediately downstream of the powerplant compared with those upstream in Iron Gate Reservoir (Fig. 53) (excerpted figures from PacifiCorp, 2017).

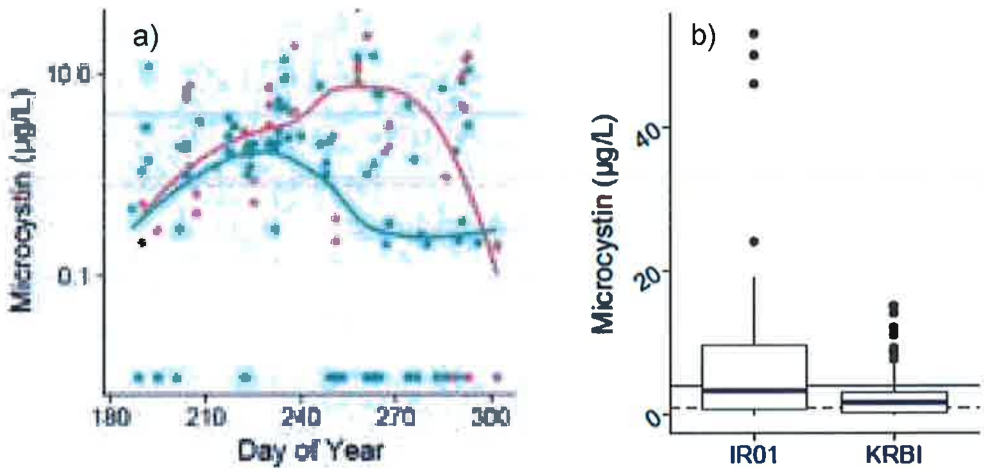

Fig. 57. Microcystin concentrations in Iron Gate Reservoir (IR01) and Klamath River below Iron Gate Reservoir (KRBI) between 2005-2014: a) microcystin concentrations plotted by day of year in Iron Gate Reservoir (light blue symbols and fitted line) and Klamath River below Iron Gate Reservoir (orange symbols and fitted line), and b) box-whisker plot for data presented in a) (PacifiCorp, 2017).

Addition of a curtain to exclude near-surface blue-green algae (typically *Microcystis aeruginosa*) from the intake to the powerplant was further found to reduce total microcystin concentrations on the intake side of the curtain by 40-80% from the upstream side of the curtain (PacifiCorp, 2017). As previously recommended, a filter curtain to restrict entrainment of zooplankton and fish should be installed near the I/O (Anderson, 2007b); the curtain can be designed to exclude surface blue-green algal scums or foams as well.

Low DO concentrations remain a difficult condition to reverse in Lake Elsinore. While LEAPS was shown to increase slightly volume-weighted concentrations of DO and concentrations in the lower water column especially at higher lake levels, greater improvements could be achieved with O_2-injection or use of aerating turbines during hydropower generation. For example, augmenting to 10 mg/L through O_2-injection substantially improved globally-averaged DO concentration, lowered nutrient, chlorophyll a and microcystin concentrations (Table 8), and with improved distribution of DO, would favorably shift biogeochemical cycling of nutrients, reduce fish kills and improve overall ecological health of the lake.

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APPENDIX

The AEM3D model is a highly sophisticated hydrodynamic-water quality-aquatic ecology model. As a result, the model includes an extremely large number of parameters that have been defined by the model authors and whose values have been adjusted for various applications. The AEM3D model package includes an example (Round Lake) which served as the default parameter set from which key model parameters were adjusted to improve fit to monitoring data (Table A1).

Table A1. Selected model parameters used in simulations (values in parentheses are provided when parameters were adjusted from values provided in Round Lake example, Hodges and Dallimore, 2016).

Physical-thermodynamic constants		
Mean albedo	-	0.077 (0.08)
Wind drag coefficient	-	0.0013
Sediment drag coefficient	-	0.005
Sediment reflectivity	-	0.9
Surface heat transfer coefficient	-	0.021 (0.0013)
Basic phytoplankton constants		Cyanobacteria / Diatoms
Phytoplankton optimum temperature	°C	28.0 / 25.0
Temperature multiplier	-	1.06 / 1.06
Maximum potential growth rate	/d	0.7 / 1.14
Ratio of C to Chl a	mg C/mg Chl_a	40 / 40
Respiration rate coefficient	/d	0.08 / 0.08
Respiration temperature multiplier	-	1.03 / 1.07
Fraction respiration to total metabolic loss	-	0.7 / 0.7
Internal toxin concentration at zero-growth	mg/L / mg Chl_a/L	0.04 / 0 (0.2 / 0)
Internal toxin concentration at max-growth	mg/L / mg Chl_a/L	0.4 / 0 (2.0 / 0)
Decay constant for toxins	/d	0.013 / 0 (0.01 / 0)
Nitrogen constants		
Half saturation constant for N	mg/L	0.045 / 0.05
Minimum internal N concentration	mg N/mg Chl_a	2.0 / 2.0
Maximum internal N concentration	mg N/mg Chl_a	4.0 / 4.0
Maximum rate of N uptake	mg N/mg Chl_a/d	0.75 / 0.75
NH_4-N release from sediments	$g/m^2/d$	0.061[a]
Maximum DON mineralization to NH_4-N	/d	0.0094 (0.01)
Nitrification rate	/d	0.10 (0.05)
Nitrification stoichiometry ratio of DO: N	-	3.43
Denitrification rate	/d	0.04 (0.01)
Phosphorus constants		
Half saturation constant for P	mg/L	0.005 / 0.005
Minimum internal P concentration	mg P/mg Chl_a	0.1 / 0.1
Maximum internal P concentration	mg/P mg Chl_a	0.6 / 0.6
Maximum rate of P uptake	mg P/mg Chl_a/d	0.1 / 0.1
PO_4-P release from sediments	$g/m^2/d$	0.005[a]
Maximum DOP mineralization to PO_4-P	/d	0.015 (0.01)

[a]Spatially-averaged NH_4-N and PO_4-P release from sediments estimated from data in Anderson (2001) with reference temperature of 20°C and temperature multiplier of 1.05.