

## Section 5

# Linkage Analysis

The primary function of a TMDL linkage analysis is to establish a link between pollutant loading from multiple sources and water quality in receiving waters. The linkage analysis serves as a key step in the use of a reference watershed approach to determine numeric targets for the Canyon Lake and Lake Elsinore nutrient TMDLs. This reference watershed approach and its use to establish numeric targets was presented in Section 3. This section provides the following information:

- *Linkage Analysis Approach (Section 5.1)* - This section describes the role of the linkage analysis in the estimation of numeric targets for Canyon Lake and Lake Elsinore using the reference watershed approach. The basis for the Linkage Analysis involves application of lake models to simulate the biogeochemical processes within each lake segment.
- *Lake Model Descriptions (Section 5.2)* - This section describes the lake models employed in developing the linkage analysis. This effort involved coupling of a biogeochemical model with a hydrodynamic model to evaluate spatially and temporally varying water quality in each lake segment. The rationale for selection of CAEDYM to simulate biogeochemical processes in both lakes and use of different hydrodynamic models for each lake (ELCOM for Canyon Lake; DYRESM for Lake Elsinore) is discussed in this section.
- *Application of lake models in Canyon Lake (Section 5.3) and Lake Elsinore (Section 5.4)* - These sections are organized in the same way to present the simulation periods, boundary conditions, input data, and key parameter estimates for the Canyon Lake and Lake Elsinore models. It is important to develop a scenario representing current inflows and outflows and associated nutrient loads, to facilitate calibration of models to generate a good fit of hydrologic and water quality results with data measurements. The calibrated models are then subjected to runoff and nutrient loading from a hypothetical reference watershed to serve as the linkage between allowable loading and receiving water quality. Lastly, comparisons of modeled lake water quality for current and reference watershed conditions are presented to illustrate expected benefits within each lake segment anticipated with TMDL implementation.

## 5.1 Linkage Analysis Approach

### 5.1.1 Role of Linkage Analysis in TMDL Revision

The linkage analysis estimates water quality response variables, chlorophyll-*a* and DO, for different level of external nutrient loading representing existing and reference watershed conditions. Results plotted as cumulative distribution frequencies (CDFs) allow for an assessment of the difference between existing and reference watershed conditions. The expectation is that with implementation of the TMDL, existing condition CDF curves will shift to be equal to or better than reference conditions, i.e., achieving the numeric targets (see Section 3 Numeric Targets).

Existing conditions approximate the current distribution of water quality in each of the three lake segments (Canyon Lake - Main Lake; Canyon Lake - East Bay; Lake Elsinore). A subset of the period of simulation for existing conditions is used to calibrate water quality model parameters to achieve a reasonable goodness-of-fit with measured data collected by the in-lake monitoring program. In the case of Lake Elsinore, the LEMP project was implemented to improve water quality by reducing the surface area of the lake and reclaimed water has been added to maintain water levels (see Section 2.2.2). These projects are accounted for as elements of the linkage analysis for existing conditions, but not as part of reference conditions.

The calibrated model developed for existing conditions is modified to evaluate the water quality response for the three lake segments for a hypothetical reference watershed condition scenario. The only physical structure included in the reference condition linkage analysis is Railroad Canyon Dam, because Canyon Lake would not exist without its presence. Simulation results for chlorophyll-*a* and DO, plotted as CDFs, serve as numeric targets for the revised TMDLs (see Section 3.4).

Lastly, the water quality models used to develop numeric targets for the lake segments will be used to test the potential benefits from existing and potential supplemental in-lake management strategies (see Section 7 Implementation).

### 5.1.2 Water Quality Model Development

The Problem Statement in Section 2 describes a unique condition for Lake Elsinore and Canyon Lake resulting from an El Nino-driven climate system within a drought-prone semi-arid region. For Lake Elsinore, climate and presence of upstream retention, including Canyon Lake, have created a natural cycle involving periods of complete lakebed desiccation. Numerical models were developed to characterize a full range of water quality responses for the greatest sources of variability, temporal in Lake Elsinore and spatial in Canyon Lake, as follows:

- *Lake Elsinore* – Lake models were developed to allow for multidecadal simulation periods needed to capture the full range of hydrologic conditions, including a period of known lakebed desiccation.
- *Canyon Lake* – Lake models were developed to allow for assessment of spatially variable water quality response, including vertical stratification and the presence of unique lake segments with limited mixing.

Numerical lake models leverage current scientific understanding of interactions among hydrology, nutrient loading, and resulting water quality in each lake. They also facilitate extrapolation of our current understanding out to hypothetical conditions in a reference watershed, or estimation of benefits from implementation of in-lake water quality control strategies. **Figure 5-1** provides a roadmap for the input data and model boundary conditions used to develop lake water quality models for Lake Elsinore and Canyon Lake.

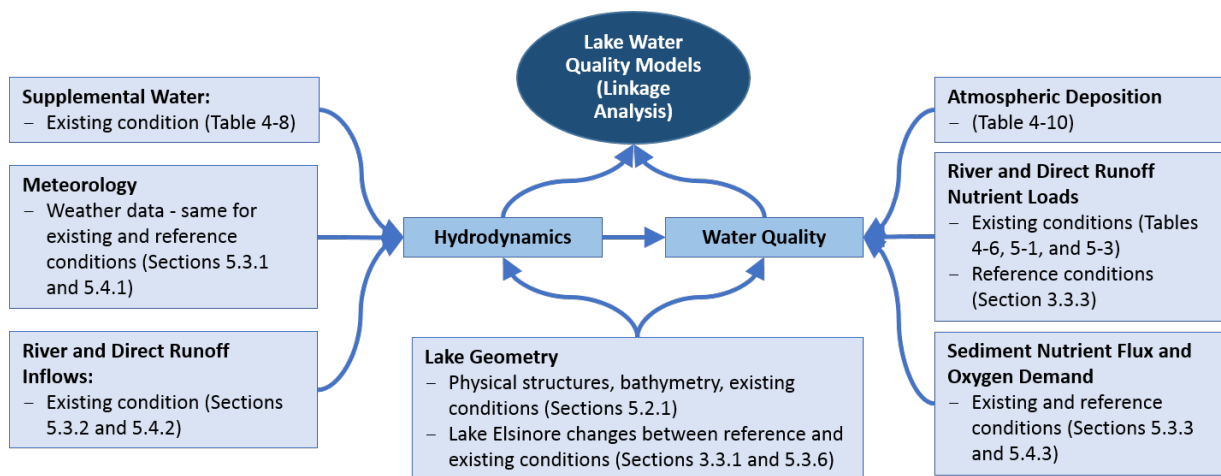


Figure 5-1. Document Location for Key Input Data and Boundary Conditions for Linkage Analysis

## 5.2 Essential Physical/Biogeochemical Processes and Model Selection

Water quality modeling involves evaluating both hydrodynamics and water quality.

Hydrodynamic lake models solve energy, momentum and water budget equations to calculate density stratification, mixing, flow and transport, as well as lake level. Water quality models typically couple with hydrodynamic models, so that they can simulate water quality responses to changes in hydrodynamics. Several models have been developed to simulate hydrodynamics and water quality in lakes and reservoirs, including CE-QUAL-W2, Environmental Fluids Dynamic Code (EFDC), DYRESM-CAEDYM, and ELCOM-CAEDYM. These models vary in sophistication, with varying levels of dimensions represented and water quality processes included. The level of sophistication needed to capture water quality in Lake Elsinore and Canyon Lake depends on the key physical and biogeochemical processes in the lakes, which is discussed in the following sections.

### 5.2.1 Physical Model Characteristics

Mathematical representation of a lake or reservoir can in some cases be as simple as a 0-D continuous stirred tank reactor (CSTR) model (Thomann and Mueller 1987; Chapra 1997), or as detailed as a finely resolved 3-D model. In the case of a 0-D model, the total volume of a waterbody is considered to exhibit instantaneous, full mixing vertically and horizontally. This can be appropriate for a waterbody that is both shallow enough to show uniform characteristics throughout the water column and also shows little variation in water quality parameters in the horizontal direction.

Lakes and reservoirs tend to be more complex systems than a 0-D model can represent; water column variations in temperature tend to result from light penetration, and this often results in a layering effect in most inland waterbodies. The dynamics of the upper, mixed layer and the deeper, dense layer below are important for hydrodynamic and water quality evaluation, because primary production (and thus oxygen generation, among other things) only occurs where light is present. Buoyant forces derived from the density gradient limit vertical mixing of the water

column, often resulting in an anoxic hypolimnion that is elevated in  $\text{NH}_4\text{-N}$  and  $\text{PO}_4\text{-P}$  and potentially also  $\text{Mn}^{2+}$ ,  $\text{Fe}^{2+}$  and  $\text{H}_2\text{S}$ .

In lakes with relatively simple geometry and little horizontal differences in temperature or water quality, a 1-D model is often utilized. 1-D thermodynamic / hydrodynamic models such as DYRESM thus explicitly assume that the primary gradient in properties is in the vertical direction and treat the waterbody as uniformly mixed laterally. The advantage of 1-D models is the low computational cost and high speed of simulations, thus allowing simulations of long periods of time and/or a large number of scenarios. As discussed below in more detail, this is the case with Lake Elsinore, which has simple enough geometry that lateral gradients in water quality parameters are not as important to water quality processes as capturing vertical variations.

For lakes and reservoirs with significant horizontal gradients in water column conditions, 2-D or 3-D representations are generally necessary. This is often the case with waterbodies that have complex geometry or spatial variations in water quality loadings. Geometric complexity of Canyon Lake, combined with its vertical stratification, require a 3-D model such as ELCOM to capture key processes of physical transport and vertical nutrient fluxes.

#### 5.2.1.1 Lake Elsinore

Lake Elsinore is a relatively large lake (approximately 3,000 surface acres at a nominal lake surface elevation of 1240' above mean sea level [MSL]) that, including the channelized part of the lake linking it to the San Jacinto River, possesses a simple geometry (13.5 mile of shoreline, shoreline development number,  $D_L$  of 3.5). The relationship between depth and lake surface area is provided in **Figure 5-2**. As shown in lake monitoring reports (and summarized in Section 2),<sup>1</sup> measurements of temperature, DO, and TDS generally demonstrate limited lateral variation but stronger variation in the vertical direction. Satellite imagery sometimes demonstrates lateral gradients in chlorophyll-*a* concentrations that result from development and wind movement of algal blooms, but averaging over several days typically damps out short-term variability in chlorophyll-*a* concentrations. With the exception of relatively rare large runoff events, pronounced lateral gradients in nutrients, TDS and other water quality properties are generally absent.

While strong lateral gradients are generally not persistent in Lake Elsinore, it is subject to extreme fluctuations in lake level and water quality over annual, decadal, and multidecadal scales (see Section 2 Problem Statement for history of lakebed desiccation). Thus, a long-term simulation that reflects several decades of hydrologic and meteorologic variability is essential in representing the dynamics of lake water quality. The extreme fluctuations in lake volume also make calibration of a 3-D model difficult, as the model domain of a 3-D model itself would vary significantly as the lake volume changed. Because of the lake's limited horizontal gradients, significant vertical gradients, and extreme response to decade-scale forcings, the 1-D DYRESM Model v.4 for Lake Elsinore was adopted. DYRESM uses a Lagrangian approach in which the thickness of the vertical layer is calculated dynamically based upon heat inputs/losses at each time step, buoyancy/density differences between layers and available mixing energy that allows segregation or combination of adjacent layers.

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<sup>1</sup> <http://www.sawpa.org/collaboration/projects/lake-elsinore-canyon-lake-tmdl-task-force/>

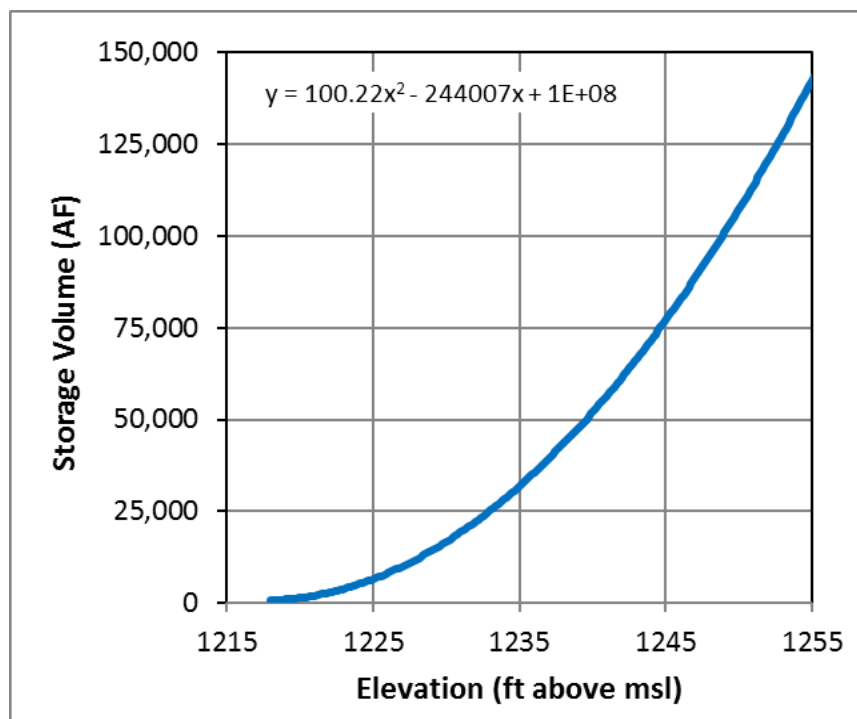


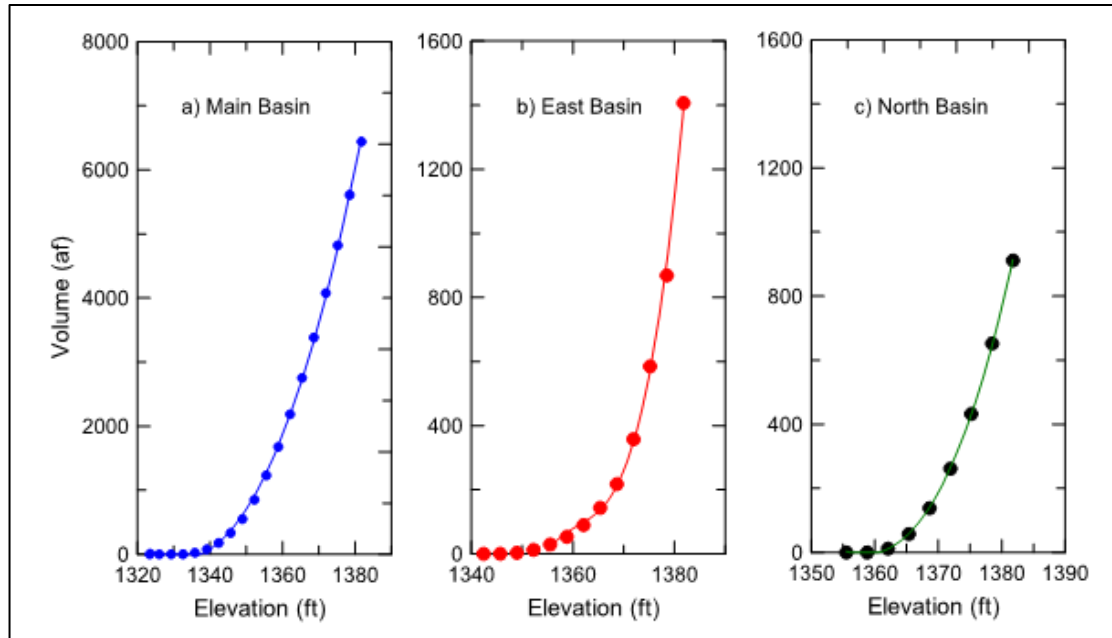
Figure 5-2. Lake Elsinore Elevation-Storage Volume Relationship

### 5.2.1.2 Canyon Lake

The 3-D ELCOM model v.3 was adopted for use in Canyon Lake because of its complex, sinuous morphology ( $D_L=13.5$ ). Strong gradients in properties exist in both vertical and lateral dimensions necessitating a 3-D model for the lake. A 20 m x 20 m lateral grid with 0.3 m vertical layers was developed for the model yielding 247 x 203 horizontal grid with 4,712 horizontal “wet” cells and 92,721 total cells in the simulation domain. A 40-second timestep was used for the simulations. Limitations on availability of USGS streamflow gage data above Canyon Lake and the intensive computational demand of a 3-D hydrodynamic/water quality model restricted the simulation to a 5-year time period. The period from 2007-2011 was selected based upon the wide range of hydrologic conditions and relatively complete water quality dataset over this period.

Canyon Lake is a smaller reservoir (436 acres, 19.7 mile shoreline) with a much more complex, sinuous morphology ( $D_L=13.5$ ) reflecting impoundment of the San Jacinto River (to the north) near its confluence with Salt Creek (to the east). Lake bathymetry and geometry suggest that strong gradients in properties may exist in both vertical and lateral dimensions, necessitating a 3-D model for the lake (**Figure 5-3**). Thus, the 3-D ELCOM Model v.3 was adopted for use.

The TMDL revision includes separate allocations for Canyon Lake Main Lake and Canyon Lake East Bay. These lake segments have very different tributary drainage areas with San Jacinto River flowing to Main Lake and Salt Creek flowing to East Bay. There is minimal exchange between these two segments of Canyon Lake during dry weather conditions. They also have very different bathymetric characteristics as illustrated in the relationship between depth and lake surface area provided in Figure 5-3.



**Figure 5-3. Canyon Lake Elevation-Volume Relationship for (a) Main Basin (Main Lake), (b) East Basin (East Bay), and (c) North Basin (North Ski Area)**

### 5.2.2 Water Quality Model Characteristics

Water quality modeling can take many forms, from simple passive scalar transport to eutrophication models involving interactive kinetics and algal growth. A linked biogeochemical-ecological model can include a large number of interacting state variables, as described in Hipsey et al. 2006.

For a scientifically defensible linkage analysis to support the development of numeric targets and estimation of nutrient reduction offset credits in Canyon Lake and Lake Elsinore, a eutrophication model is needed to simulate the relationships between nutrients, algae and DO. Nutrient fluxes into the water column from lake bottom sediments in both Canyon Lake and Lake Elsinore have been shown as an important source for water column concentrations (See Section 3.3 for discussion of Internal Sources). It is also critical that sediment fluxes be represented in the water quality model selected.

CAEDYM includes full eutrophication kinetics and can adequately represent water column water quality dynamics in both lakes. Water quality in Lake Elsinore and Canyon Lake was simulated using CAEDYM v.3. This model can be linked to both DYRESM and ELCOM, allowing for a consistent water quality solution between Canyon Lake and Lake Elsinore, while the hydrodynamics are tailored to the specific systems being modeled.

## 5.3 Lake Elsinore Model Configuration, Calibration and Scenario Simulations

### 5.3.1 Meteorological Input Data

Meteorological inputs include the shortwave solar heat flux (300-3,000 nanometers [nm]) that includes photosynthetically available radiation (Photosynthetically Active Radiation [PAR], 400-700 nm), as well as near-ultraviolet (UV) (300-400 nm) and near-infrared (IR) and IR (700-3,000 nm), air temperature and windspeed.

Meteorological conditions for the calibration period were taken from the California Irrigation Management Information System (CIMIS) station #44 at UCR (**Figure 5-4**), which provided shortwave solar heat flux (300-3,000 nm) (Figure 5-4a), air temperature (Figure 5-4b) and windspeed (Figure 5-4c). Values are represented as daily average values in the model. A strong seasonal trend in solar shortwave heat flux is evident in the figure, with daily average shortwave flux values of about 350 watts/square meter ( $W/m^2$ ) in the summer and 50-100  $W/m^2$  during the winter (Figure 5-4a). Daily average air temperatures exhibit a similar seasonal pattern, with daily-averaged summer temperatures near 30°C and daily average winter temperatures generally 7-10°C (Figure 5-4b). Daily average windspeeds averaged near 2 meters/second (m/s) and exhibited some seasonality as did daily rainfall rates that also showed annual variability (Figure 5-4c, d).

### 5.3.2 Hydrologic Input Data

In addition to direct precipitation on the lake surface, water delivered to the lake included San Jacinto River flows, runoff from the local watershed, and supplemental water that includes recycled water from EVMWD and water pumped from island wells in 2003-2004 (collectively represented as recycled water in the model). Lake outflows include a lake outlet channel to downstream Temescal Creek.

The San Jacinto River is the primary watershed runoff inflow to Lake Elsinore and includes all overflow volume from Canyon Lake. Continuous flow data recorded at USGS Station 11070500 are input to the lake models. Daily runoff from the local watershed has been estimated in previous studies (Anderson 2015a), and yields are comparable to long-term average annual volume inflows (see Section 4, e.g., Table 4-4). Recycled water discharge to Lake Elsinore has been documented by EVMWD since inception. All modeled inflows are shown in **Figure 5-5**.

A limited number of large runoff events delivered most of the flows from the San Jacinto River during this time period, including the very large runoff events at the beginning of 2005, that included daily flow exceeding 8,000 acre-feet. Shorter duration high flow runoff events were also present in January 2010 and December 2011. Precipitation generated runoff from the local watershed as well, although daily flows were much smaller than the very large runoff events noted in 2005, 2010 and 2011. Daily rates of recycled water flow are much lower than periods with wet weather runoff from the watershed. Presented as cumulative flows however, we see that recycled water inputs exceeded that of local runoff and contributed about 50,000 acre-feet since inputs began in late 2002 (**Figure 5-6**). Based upon these values, a total of 187,926 acre-feet of water was delivered to Lake Elsinore over this 2000-2014 period, with approximately 53% derived from San Jacinto River flows, 20% from local runoff and 27% from recycled water.

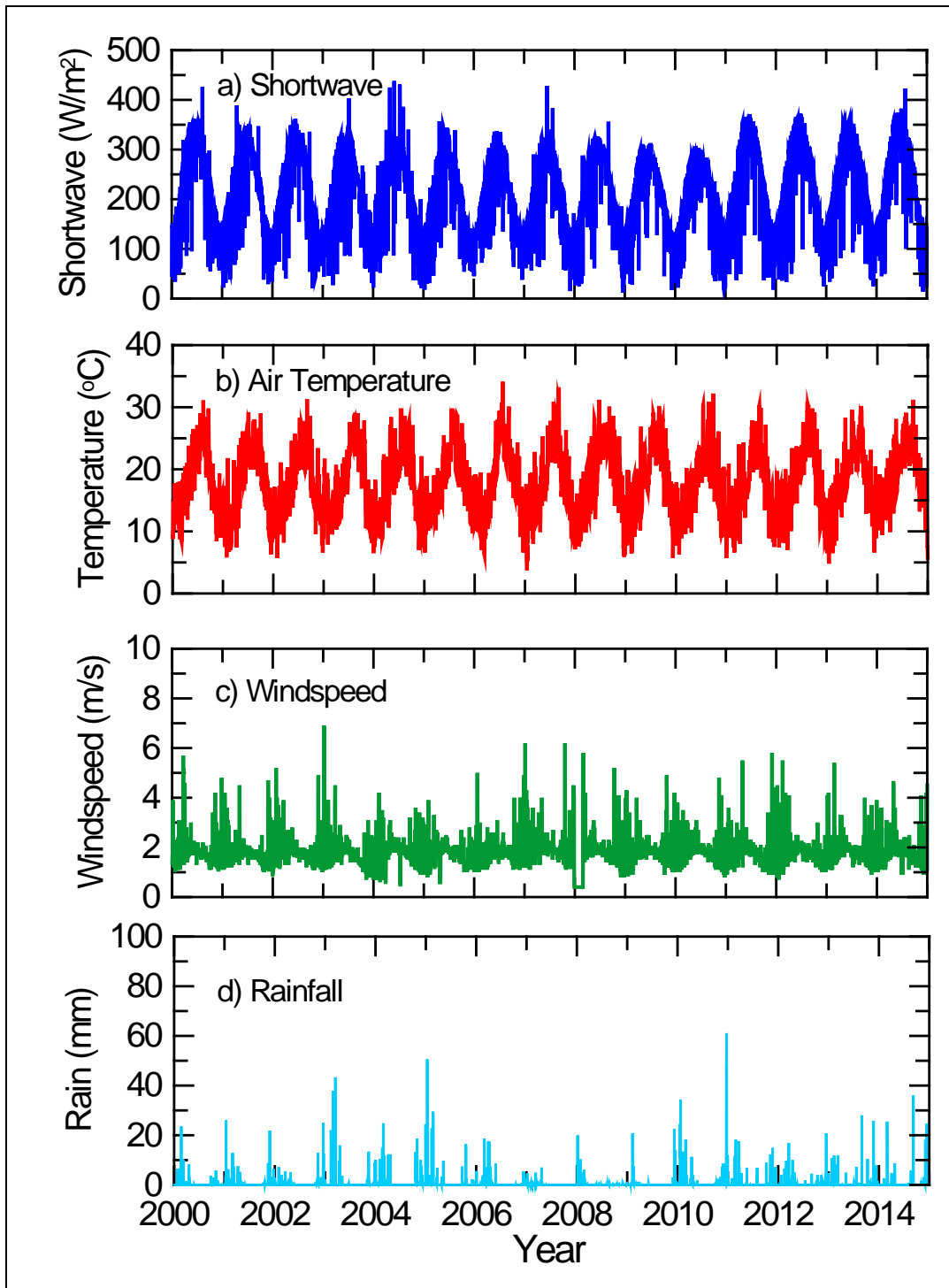


Figure 5-4. Daily Average (a) Shortwave Radiation, (b) Air Temperature, (c) Windspeed and (d) Rainfall Used in Model Simulations for the Calibration Period 2000-2014



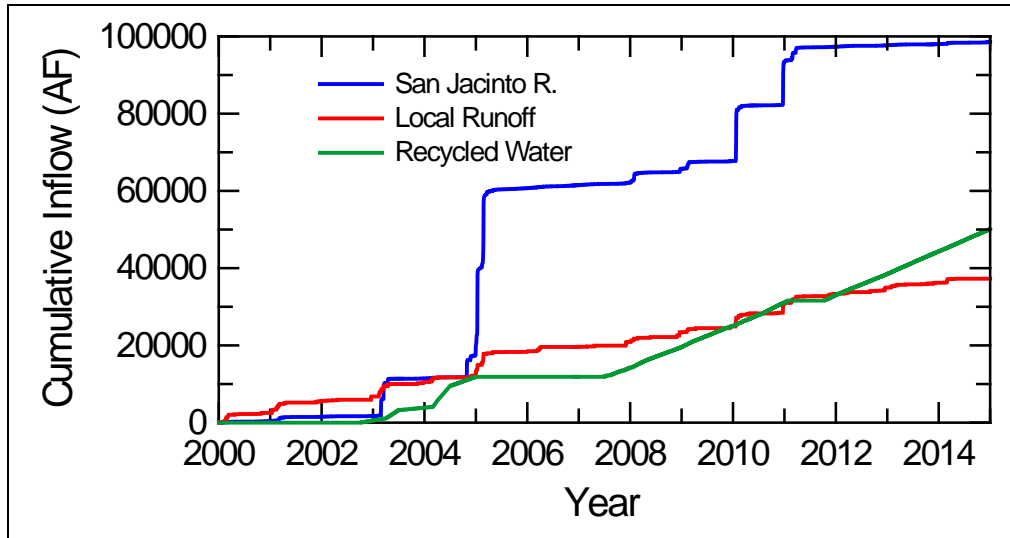


Figure 5-5. Daily Inflows to Lake Elsinore for the Calibration Period 2000-2014

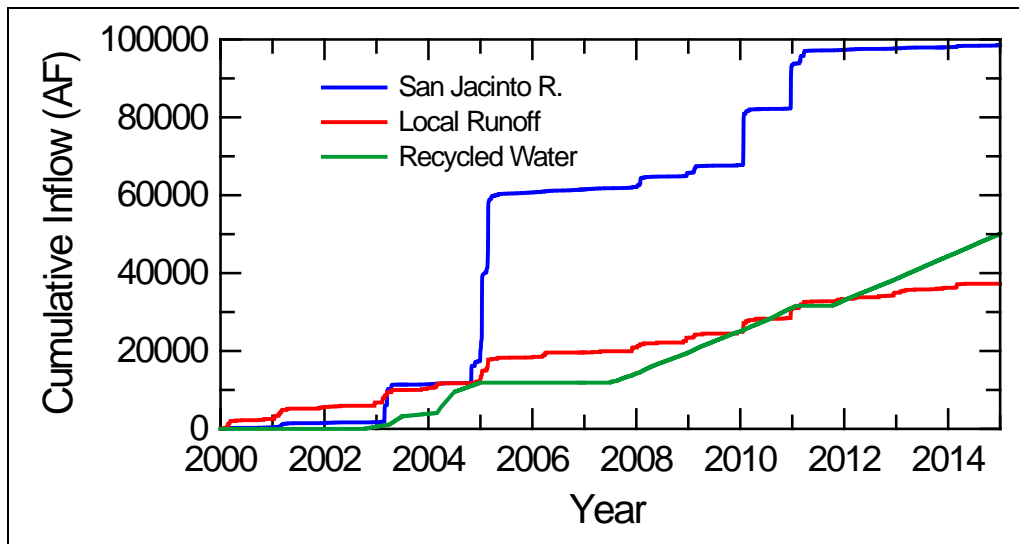


Figure 5-6. Cumulative Inflow to Lake Elsinore from the San Jacinto River, Local Runoff and Recycled Water for the Calibration Period 2000-2014

### 5.3.3 Nutrient Water Quality

Concentrations of nutrients in these inflows vary depending upon a number of factors, including intensity and duration of storms, interval of time between storms and other factors (including treatment plant operation for recycled water inputs). Average concentration values derived from runoff sampling within the watershed and treatment plant data were used in model simulations (Table 5-1). Total external nutrient loading over the calibration period was calculated from flow data (Figure 5-5) and nutrient concentrations (Table 5-1).

**Table 5-1. Nutrient Concentrations (mg/L) of Inflows to Lake Elsinore Used in Model Simulations**

Source	PO <sub>4</sub> -P	Total P	NH <sub>4</sub> -N	NO <sub>3</sub> -N	Total N
San Jacinto River	0.28	0.51	0.22	0.57	1.89
Local Runoff	0.20	0.48	0.22	0.80	1.82
Recycled Water <sup>1</sup>	0.32	0.41	0.36	1.62	2.87

<sup>1</sup> Recycled water concentrations for EVMWD 2007-present

For internal water quality processes, default water quality parameters were used in CAEDYM (Hipsey et al. 2006) in CAEDYM except for key parameters for bioavailable nutrient (soluble reactive phosphorus [SRP] and NH<sub>4</sub>) fluxes and sediment oxygen demand (SOD), as follows:

- Internal loading of nutrients, i.e., the bioavailable nutrient flux from lake bottom sediment, is recognized as a very important process in Lake Elsinore, accounting for approximately 85 percent of long-term nutrient load (see Section 4). Measurements of internal loading have been conducted periodically at the lake using the core-flux method (Anderson 2001, 2010). Internal loading rates exhibit significant spatial and temporal variation based on core-flux estimates, largely driven by the non-uniformity of large rainfall events and settling of particulates to the lake bottom. For the TMDL revision, the average flux rates from previously collected core samples (100 milligrams/square meter/day [mg/m<sup>2</sup>/d] NH<sub>4</sub>-N and 10 mg/m<sup>2</sup>/d SRP) were assumed to approximate long-term average internal loading (see Section 4.3.1). The long-term average sediment nutrient flux rate is a constant input to CAEDYM for simulated nutrients for standard conditions. CAEDYM estimates a daily flux of dissolved nutrients as a function of dynamic changes in water temperature, DO, and pH.
- SOD is also high for this eutrophic lake (Anderson 2010); an average value of 0.8 grams/square meter/day (g/m<sup>2</sup>/d) was used in the model calibration. To accommodate time constraints on modeling efforts, a static internal loading model was used in these simulations that allows internal loading rates to vary with temperature and DO, but does not explicitly simulate sediment deposition and associated biogeochemical changes resulting in nutrient recycling and efflux from sediments.

### 5.3.4 Model Calibration

The Lake Elsinore coupled DYRESM-CAEDYM model was calibrated against available data for 2000-2014. Model calibration was focused on assessing model-data agreement on an annual to decadal scale. For this reason, diurnal fluctuations in hydrodynamic and water quality parameters are not the focus of this calibration effort. The adequate representation of long-term trends implies representation of short-term trends for the purposes of this long-term TMDL study.

#### 5.3.4.1 Lake Surface Elevation

**Figure 5-7** contains a time series comparison between measured and modeled lake surface elevations during the calibration time period. Observations indicate a marked decline in elevation over the years 2000 through 2003, 2005 through 2010, and 2011 through 2014. A dramatic increase in elevation occurs at the end of 2004 and in early 2005. Modeled water surface

elevations reflect all of these observed trends and also match closely in magnitude. Absolute model results match observations within a foot for the majority of the simulation.

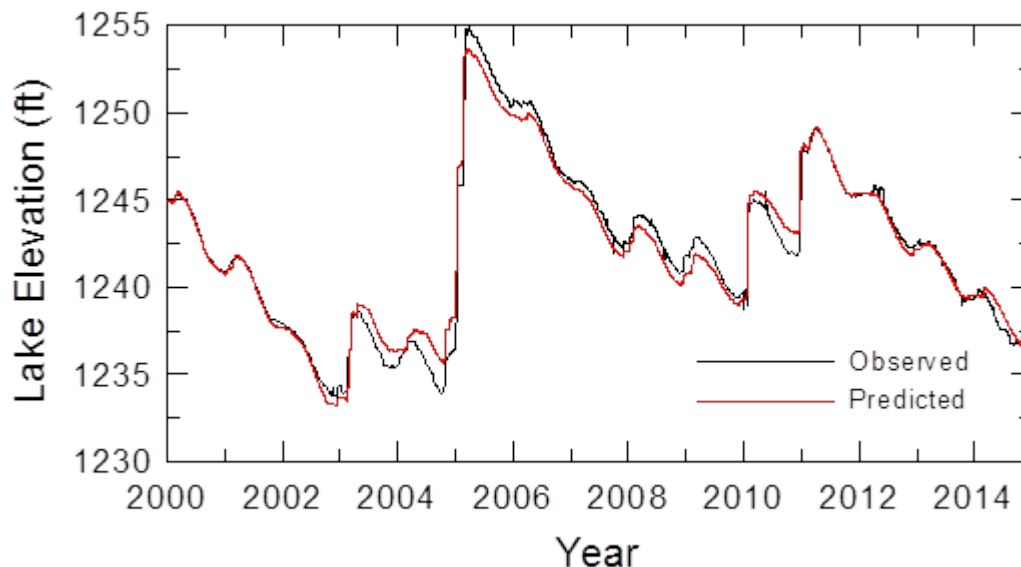


Figure 5-7. Predicted and Observed Lake Surface Elevation for the Calibration Period 2000-2014

#### 5.3.4.2 Salinity

Salinity in the lake varied from approximately 700 – 2,600 mg/L TDS, with low concentrations following the very large runoff in winter 2005 (**Figure 5-8**, solid circles). The model captured trends in TDS reasonably well, including the high TDS concentrations measured in late fall 2002 and the marked decline in TDS in 2005 (Figure 5-8, line). The only discrepancy was found in 2014, when the model over-predicted TDS in the lake.

#### 5.3.4.3 Temperature

The model reasonably captured measured temperature values in Lake Elsinore (**Figure 5-9**). The model correctly predicted strong seasonal trends in water column temperature that reflects seasonal trends in solar shortwave heat flux (see Figure 5-4a) and air temperature (see Figure 5-4b). The model predicted summer values near 27°C and winter minimum values near 10°C, with little difference between depths reflecting weak stratification or mixed conditions commonly present in the lake (Figure 5-9).

#### 5.3.4.4 Dissolved Oxygen

DO in the lake varied seasonally and with depth (**Figure 5-10**). The temperature effect on oxygen solubility was evident in model predictions for the 2-meter depth, with DO values generally near 10 mg/L in the winter and 7-8 mg/L in the summer (Figure 5-10a). At the same time, supersaturation was periodically predicted (e.g., in spring 2011 when concentrations reached 17 mg/L). The model predicted DO concentrations deeper in the water column to be often quite similar to near-surface values, but did also correctly predict periods of anoxia in the summer of 2003, 2004, 2006 and 2010 (Figure 5-10b).

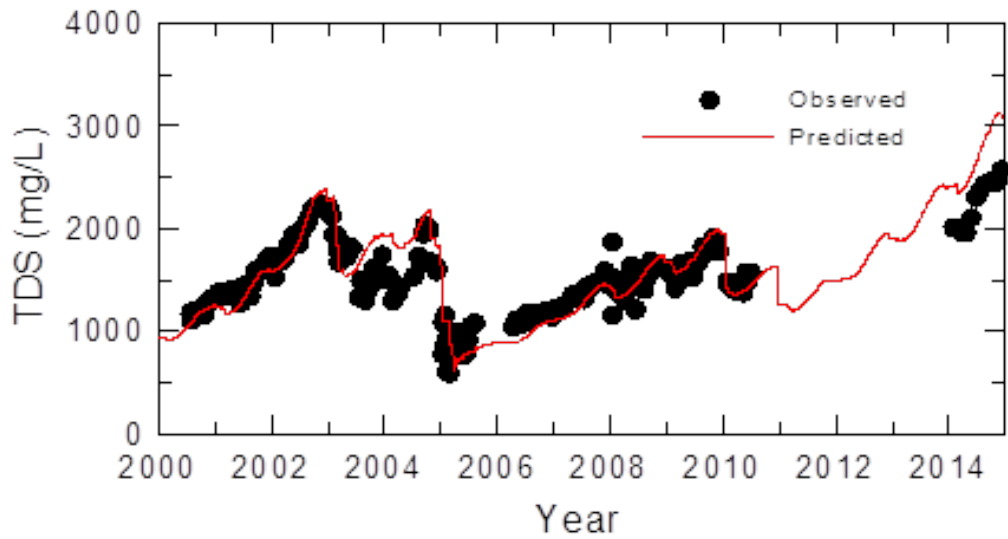


Figure 5-8. Predicted and Observed TDS Concentrations for the Calibration Period 2000-2014

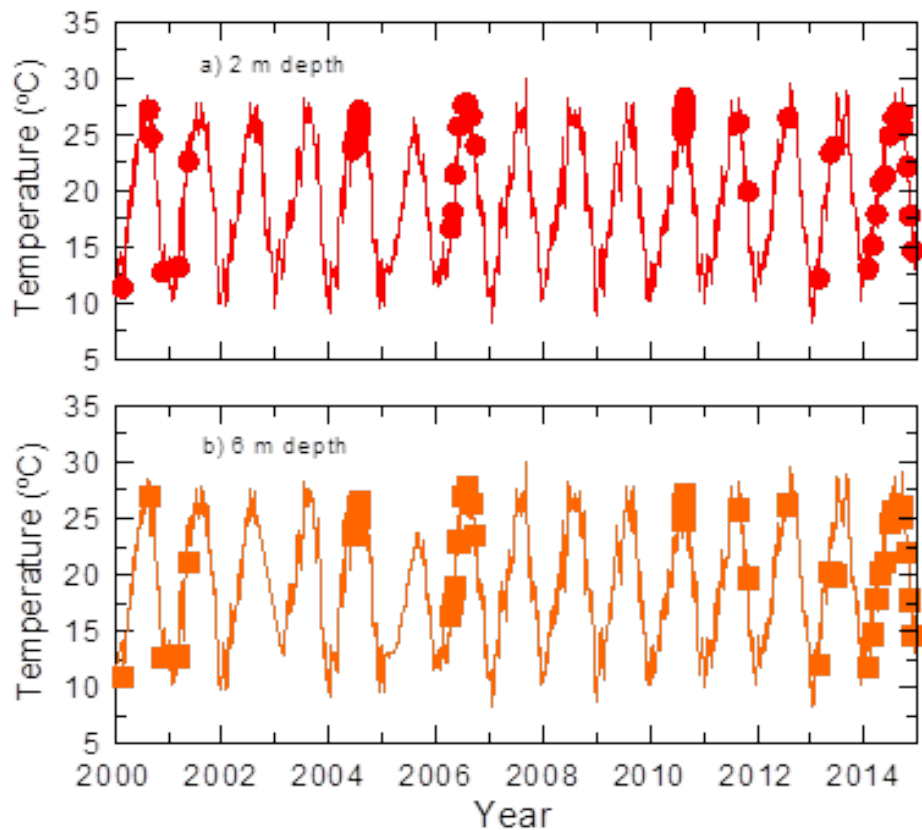


Figure 5-9. Predicted and Observed Temperature at (a) 2-meter and (b) 6-meter Depths for the Calibration Period 2000-2014

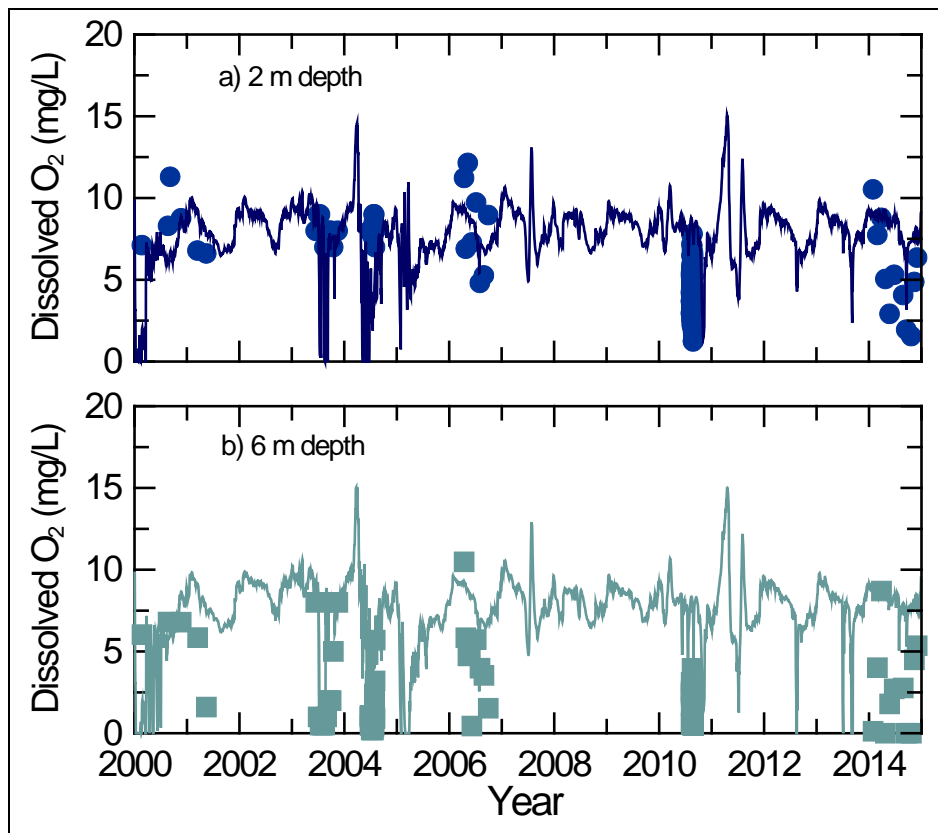


Figure 5-10. Predicted and Observed Dissolved Oxygen Concentrations at (a) 2-meter and (b) 6-meter Depths for the Calibration Period 2000-2014

#### 5.3.4.5 Total Nitrogen

The model did a fair job of capturing the dramatic trends in concentrations of TN in the lake between 2000 and 2015 (**Figure 5-11**). Concentrations increased from about 2 mg/L in 2000 to greater than 8 mg/L by late 2004, and then declined sharply with the very large runoff volumes delivered in winter of 2005 that quadrupled the volume of the lake. TN concentrations then edged up over several years before declining slightly in 2010 (**Figure 5-11**). While the model captured trends reasonably well, it did not reproduce the more significant apparent swings observed, e.g., in 2008, when reported concentrations over the period of a few months ranged from < 1 to > 8 mg/L. It may be that sampling bias or analytical challenges crept into the time series data, exaggerating short term trends.

#### 5.3.4.6 Total Phosphorus

Total P concentrations also varied quite dramatically over this calibration period, from about 0.1 mg/L in 2,000 to > 0.6 mg/L in late 2004 before declining to a value near 0.2 mg/L (**Figure 5-12**). The model generally captured trends but under predicted concentrations somewhat in 2003-2004, although it did predict a maximum value of about 0.6 mg/L in late 2004 (**Figure 5-12**).

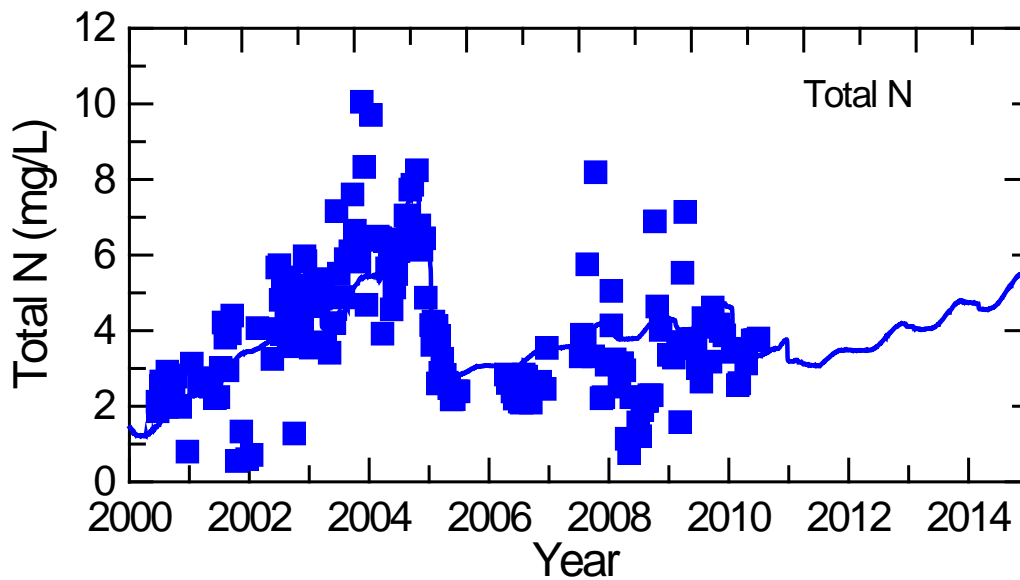


Figure 5-11. Predicted and Observed Total Nitrogen Concentrations for the Calibration Period 2000-2014 (Note: additional data points for 2013-2015 to be added)

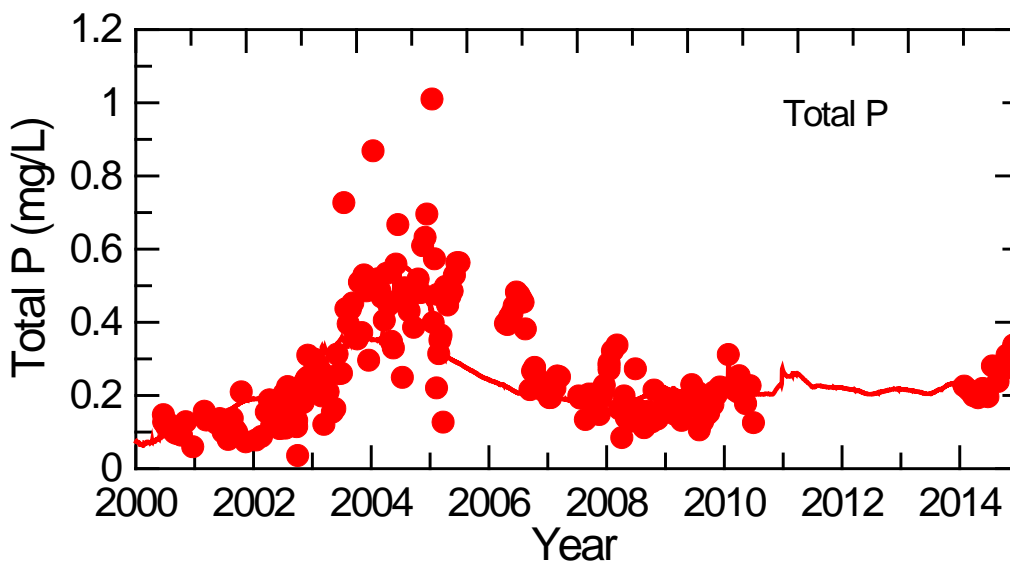


Figure 5-12. Predicted and Observed Total Phosphorus Concentrations for the Calibration Period 2000-2014

#### 5.3.4.7 Chlorophyll-*a*

Measured chlorophyll-*a* concentrations exhibited pronounced seasonal and inter-annual variability, ranging from  $< 10 \mu\text{g/L}$  in some winters to  $> 300 \mu\text{g/L}$  in 2002, 2004 and 2014 (Figure 5-13, solid symbols). The model did a fair job overall in reproducing these complex trends and correctly predicted summer maximum chlorophyll-*a* concentrations in 2000-2004 (Figure 5-13, line). The model did not do as well predicting the winter minimum values however,

and missed the particularly high concentrations observed in 2014 (Figure 5-13).

Notwithstanding, the agreement between predicted and observed concentrations was considered acceptable given the highly dynamic algal community in the lake and the complex dependence of chlorophyll-*a* concentrations on nutrient availability and ecosystem structure.

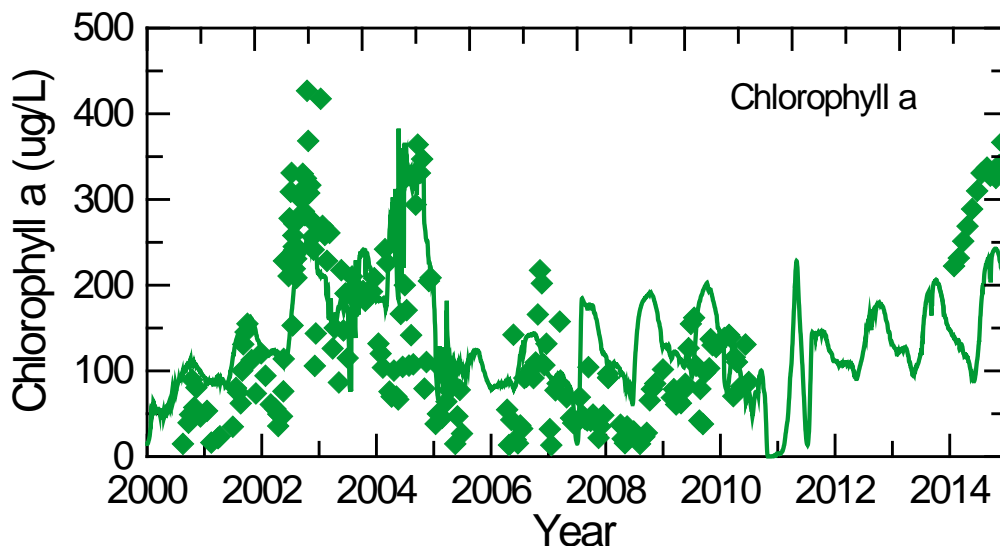


Figure 5-13. Predicted and Observed Chlorophyll-*a* Concentrations for the Calibration Period 2000-2014

### 5.3.5 Water Quality Model Summary Statistics

The overall goodness-of-fit of the model results to measured concentrations of TN, TP and chlorophyll-*a* was assessed using the relative percent error between predicted and observed average concentrations (Table 5-2). TN averaged 3.98 mg/L over this period, while the model yielded an average value of 3.88 mg/L, representing a 2.5% underestimate (Table 5-2). The average observed TP concentration over this period was 0.265 mg/L while the predicted average concentration was 0.235 mg/L, an 11.3% underestimate. Predicted and observed chlorophyll-*a* concentrations were 130 and 137  $\mu\text{g/L}$ , corresponding to a % Relative Error (%RE) of 5.4%. Given the extreme range in conditions experienced at the lake over this 2000-2014 period, the model reasonably predicted water quality in Lake Elsinore under a wide range of hydrologic, chemical and ecological conditions, allowing for comparison of water quality under different conditions and scenarios.

Table 5-2. Mean Observed and Predicted Values of Key Water Quality Parameters for Calibration Period (2000-2014) for Lake Elsinore

Variable	Observed	Predicted	% Error
TN mg-N/L	3.98	3.88	-2.5
TP mg-P/L	0.265	0.235	-11.3
Chlorophyll- <i>a</i> $\mu\text{g/L}$	130	137	+5.4

### 5.3.6 Reference Condition Scenario Evaluation

The linkage analysis was used to evaluate the water quality conditions in Lake Elsinore for a scenario where external loads are reduced to levels representative of a reference watershed condition to develop numeric targets for response variables, ammonia-N, DO and chlorophyll-*a*. Section 3.3 describes the water quality input data and lakebed characteristics that define the reference condition for estimating numeric targets. This scenario was developed for a 99-year (2016-2015) simulation period coinciding with available daily flow data for the San Jacinto River near Elsinore USGS gauge 11070500. Watershed runoff from 90 percent of the Lake Elsinore watershed, including all Canyon Lake overflows, are recorded by this gauge. Rainfall records for Lake Elsinore (RCFC&WCD Station# 067) also go back to 1916, facilitating estimation of daily runoff from the local Lake Elsinore watershed by applying a runoff coefficient model for this same period (Anderson 2015). Reference watershed nutrient concentrations are assumed to occur in the total (USGS gauge + local runoff model) daily inflow volume to Lake Elsinore.

A 1-D model allows simulation of conditions in the lake over long time periods due to relatively modest computational demands. A minimum layer thickness of 0.25 m and maximum layer thickness of 1.0 m was used for these simulations, with a 2-hr timestep. As discussed in Section 3, the LEMP involved construction of a levee to separate the main lake from the back basin, reducing the lake surface area from about 6,000 to 3,000 acres, thereby reducing evaporative losses and internal loading, and in turn improving water quality. This project is not included in the reference condition for Lake Elsinore, and therefore a much larger lake basin is used for the reference condition simulation. A different elevation volume relationship for the reference condition lake basin is included in the plot of current conditions in Figure 5-2 above. **Figure 5-14** shows the footprint of the lake without the levee.

Results of the reference condition model for Lake Elsinore are plotted as time series in **Figure 5-15** for lake level, TDS, TP, TN, ammonia-N, DO and chlorophyll-*a*. The results for water quality response variables ammonia-N, DO, and chlorophyll-*a* are plotted as CDFs and serve as the basis for numeric targets (see Figures 3-6 through 3-8). The plots clearly show the impact of multidecadal trends in lake level upon TDS and nutrients, and in turn, upon response variables chlorophyll-*a* and DO for a naturally occurring reference watershed condition. While seasonal variability can be detected in the response variables, it is much less significant than longer-term trends, with highly productive periods (as indicated by rising chlorophyll-*a* concentrations and greater diurnal fluctuations in DO) persisting for multiple years or decades.

## 5.4 Canyon Lake Model Configuration, Calibration and Scenario Simulations

Limitations on availability of USGS streamflow gage data above Canyon Lake and the intensive computational demand of a 3-D hydrodynamic/water-quality model restricted the simulation to a 5-year time period for calibration. The 2007-2011 period was selected based upon the wide range of hydrologic conditions and relatively complete water quality dataset over this period of time.



### 5.4.1 Meteorological Input Data

The model requires sufficient meteorological data to calculate instantaneous heat budgets for the lake and mixing due to wind shear and convective processes. Hourly meteorological data from the CIMIS station located near UCR, with correction for elevation difference, was used to drive the hydrodynamic-thermodynamic model. A wind-sheltering factor of 0.4 was applied for East Bay to account for the effects of steep topography on wind speed there. The model also requires information for inflows and withdrawals to account for turbulent kinetic energy inputs to the water column via these mechanisms. Flow data for the calibration period were taken from the USGS gaging stations on the San Jacinto River at Goetz Road (USGS gage #11070365) and on Salt Creek (USGS gage #11070465). Water quality measurements for the (limited) flows entering the lake over this period were not available, so average values from previous sampling conducted on the San Jacinto River and Salt Creek were used as inputs (Dyal and Anderson 2003). Information on volumetric withdrawals from the lake over this period were provided by EVMWD (J. Ma, personal communication).



Figure 5-14. Comparison of Current Lake Elsinore Hydrography with Approximate Pre-LEMP Hydrography (shapefile from NHD)

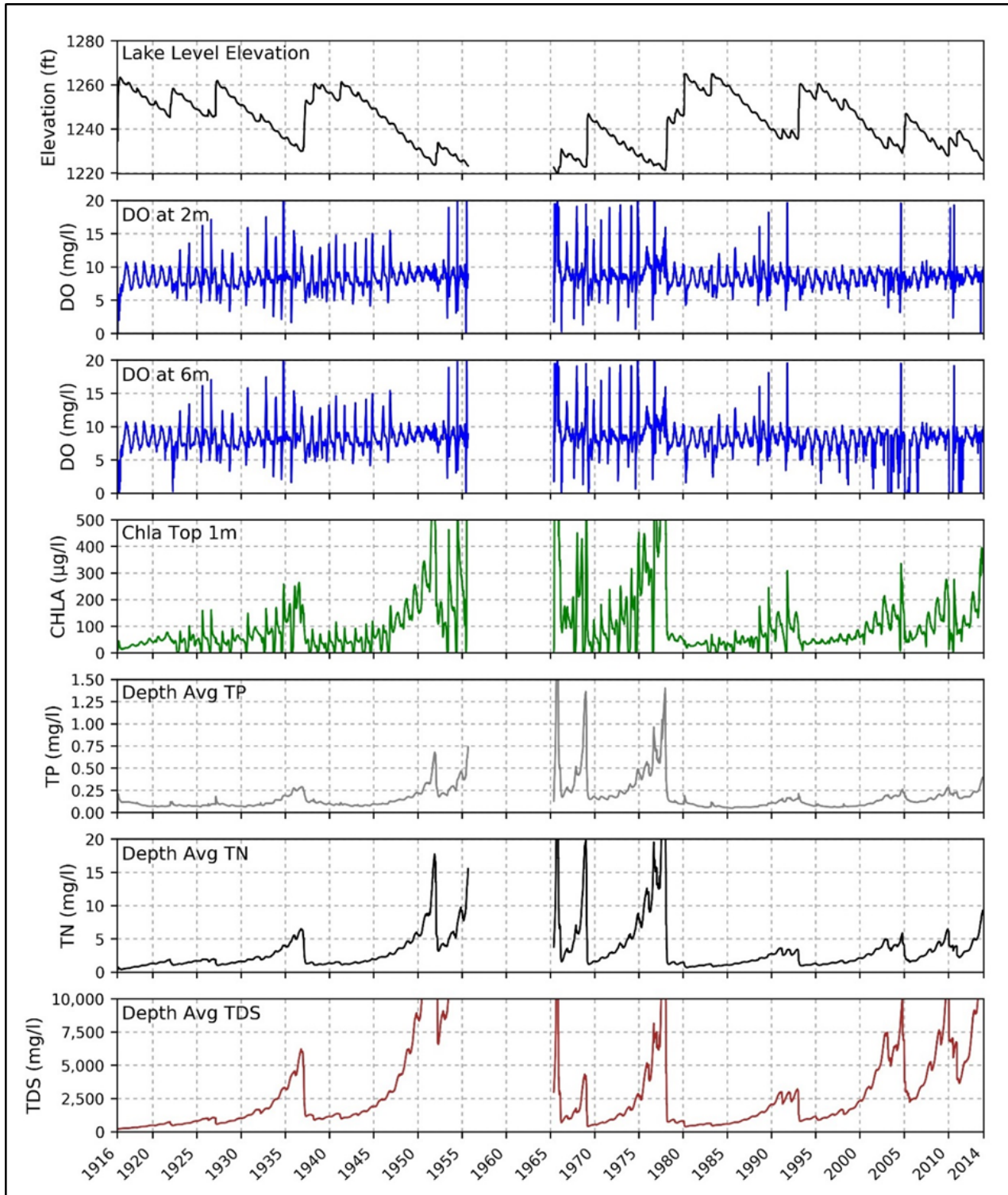
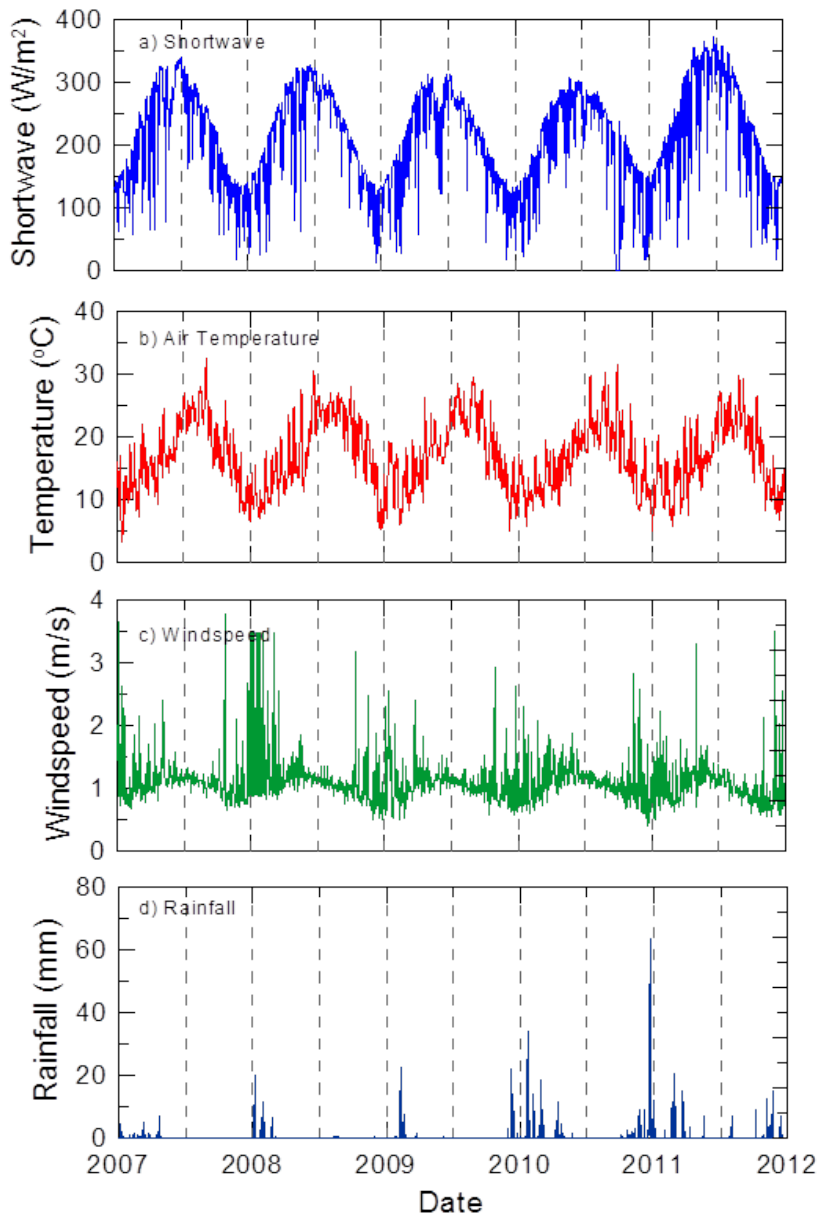


Figure 5-15. Time Series Output of Water Quality Parameters for Reference Condition Simulation for Lake Elsinore (1916-2015)

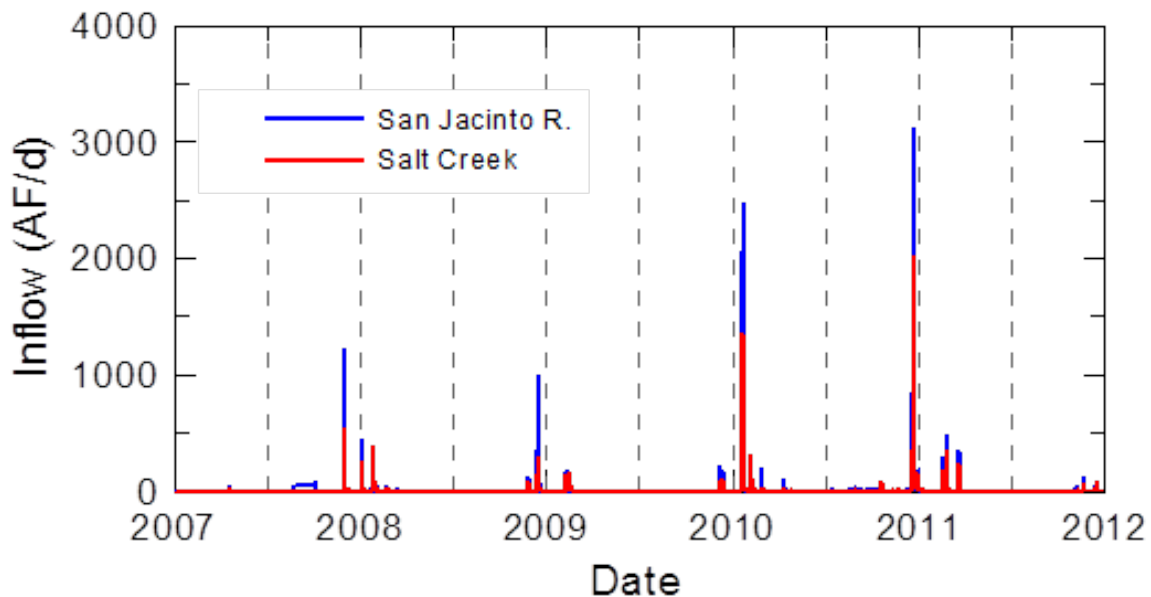
Daily average meteorological data were calculated from hourly data and presented in **Figure 5-16**. As previously seen for Lake Elsinore, clear seasonal trends are evident in critical meteorological parameters. Daily solar shortwave radiation was low during the winter, with cloud cover during winter storms lowering the daily average flux to  $< 50 \text{ W/m}^2$  on numerous occasions (Figure 5-16a). Daily shortwave flux reached maximum values of  $> 300 \text{ W/m}^2$  in the early summer months (Figure 5-16a), although we note that maximum daily air temperatures were reached later in the summer (Figure 5-16b). Daily average wind speeds, while variable, were also generally stronger during the winter months (Figure 5-16c), which in many cases coincided with rainfall events (Figure 5-16d).



**Figure 5-16. Daily Average (a) Shortwave Radiation, (b) Air Temperature, (c) Windspeed and (d) Rainfall Used in Model Simulations for the Calibration Period 2007-2011**

### 5.4.2 Hydrologic Input Data

The majority of inflows for the Canyon Lake hydrologic budget involves runoff from the San Jacinto River and Salt Creek (**Figure 5-17**). Inflow data for the calibration period are taken from two USGS gauges; the San Jacinto River at Goetz Rd (Sta#11070365) and Salt Creek at Murrieta Road (Sta#11070465). These gauges measure runoff from 90 percent of the Canyon Lake drainage area, thus a scaling factor of 1.1 was applied to account for flows from the local Canyon Lake watershed (from lakeshore and Meadowbrook and Quail Valley tributaries). Generally, no flow is present during dry weather conditions as measured by USGS gauges. Rainfall driven runoff occurs in the wet season, and volume is dominated by few extremely large events (e.g., with > 2,000 and > 3,000 acre-feet/day (af/d) flows in Salt Creek and San Jacinto River in late December 2010) (Figure 5-17). It was previously noted that these extreme events are responsible for much of the external nutrient loading in a year, with large runoff years in turn dominating loading from the watershed for several years or more (Anderson 2012).



**Figure 5-17. Daily Inflows to Canyon Lake for the Calibration Period 2007-2011**

### 5.4.3 Nutrient Water Quality

Concentrations of nutrients in watershed runoff inflows vary depending upon a number of factors, including intensity and duration of storms, interval of time between storms and other factors (including retention in upstream lakes or channels). Average concentration values derived from runoff sampling within the watershed were used in model simulations (**Table 5-3**). Total external nutrient loading over the calibration period was calculated from flow data (see Figure 5-17) and nutrient concentrations (Table 5-3).

**Table 5-3. Nutrient Concentrations (mg/L) of Inflows to Canyon Lake Used in Model Simulations**

Source	PO <sub>4</sub> -P	Total P	NH <sub>4</sub> -N	NO <sub>3</sub> -N	Total N
San Jacinto River	0.35	0.71	0.31	0.77	2.57
Salt Creek	0.27	0.54	0.29	0.75	2.49

For internal water quality processes, default water quality parameters were used in CAEDYM (Hipsey et al. 2006) except for key parameters for bioavailable nutrient (SRP and NH<sub>4</sub>) fluxes and SOD, as follows:

- Rates of internal loading of nitrogen and phosphorus to the water column were separately measured in laboratory core-flux studies (Anderson 2001; Anderson 2007a). Samples collected prior to the commencement of alum addition in 2013, had average sediment nutrient flux rates of 43.3 mg/m<sup>2</sup>/d for NH<sub>4</sub>-N for the 3 main basin sites, with similar average flux rates also found for the two East Bay sites (45.0 mg/m<sup>2</sup>/d). Average SRP flux from the sediments was lower than that of N (15.3 and 16.0 mg/m<sup>2</sup>/d for the Main Lake and East Bay sites, respectively).
- SOD was determined based on Anderson (2001) and Anderson (2007a). Measurement conducted in July 2006 found SOD values of about 0.3 g/m<sup>2</sup>/d, with very little difference between any of the sites (Anderson 2007a). Additional measurements made in April 2007 found slightly higher short-term SOD values (0.36-0.38 g/m<sup>2</sup>/d), although longer-term SOD values were somewhat lower (0.22-0.25 g/m<sup>2</sup>/d). An average SOD value of 0.3 g/m<sup>2</sup>/d was used for the model calibration.

As with DYRESM, the ELCOM and CAEDYM models require a very large number of parameters; default values were used for almost all thermodynamic and chemical/biological/ecological values.

#### 5.4.4 Model Calibration

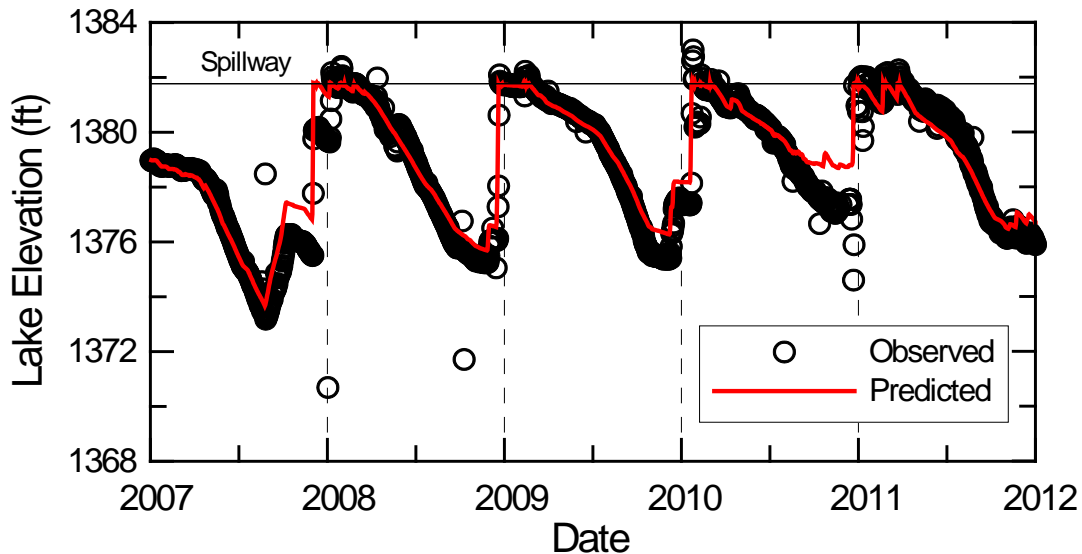
The model was calibrated against water column data collected at Canyon Lake from January 2007 – December 2011. Samples were collected at varying intervals but were generally collected monthly to bimonthly. Hydrolab casts were made at five sites on the lake, providing vertical profile measurements of temperature, DO, pH, electrical conductance, oxidation-reduction potential, and turbidity. Depth-integrated surface samples were analyzed for chlorophyll-*a*, total and dissolved nutrients, and other constituents. Discrete samples were also collected at the thermocline, and composited discrete samples from two to three depths within the hypolimnion were also collected (except during the winter when the water column was well-mixed vertically and only a single depth-integrated sample was collected from each site). Section 2 summarizes monitoring program results from Canyon Lake; key data from this program were used for calibration in this section.

A large number of model simulations were conducted for the period January 1, 2007 – December 31, 2011; default model parameters were used in initial simulations and compared visually with

observed data. Model parameters were varied to improve goodness-of-fit between observed and predicted values.

#### 5.4.4.1 Lake Surface Elevation

The reported lake surface elevations (symbols) were reasonably well-reproduced in the simulation (solid red line). The model captured the evaporation and drawdown of about six feet that occurred each summer as well as the generally very rapid increase in lake surface elevation each winter to the spillway elevation (**Figure 5-18**).



**Figure 5-18. Lake Surface Elevation for the Calibration Period 2007-2011**

Excluding some isolated outliers in the reported lake surface elevations, the only significant difference between observed and predicted values was found in late 2010, when the model predicted surface elevations near 1,379 feet (ft), while reported values were closer to 1,377 ft (Figure 5-18). The source of this discrepancy is not clear. Notwithstanding this anomaly, the model did a good job reproducing the average elevation over this period (1,378.71 vs. 1,378.79 ft, respectively), with %RE of 0.03%.

#### 5.4.4.2 Temperature

Temperature is an important property in lakes, regulating stratification and governing rates of chemical and biological reactions. Observed temperature values at depths of 2 meters (solid blue circles) and 12 meters (open orange circles) for site Main Lake site M1 (and other sites) were reasonably reproduced in the simulation (**Figure 5-19**). The model captured the rapid increase in near-surface (2 m) temperature from about 10-12 °C in the winter to nearly 30°C in the summer, as well as the rapid decline in the fall (Figure 5-19) due to reduced solar shortwave radiation inputs and lower air temperatures (Figure 5-16 above). The %RE between predicted and observed temperatures for 2-meter depth in the lake was 4.0% (n=80) with the mean predicted temperature of 21.3°C in good agreement with the observed mean value (21.5°C). The model (orange line) also reasonably reproduced temperatures at 12-meter depth (orange symbols) that

increased slowly during much of the year before increasing more dramatically in the fall during lake turnover (Figure 5-19). The model predicted a somewhat later turnover date in the fall of 2008 and 2010 compared with available temperature data, but reproduced turnover well in fall 2007 and 2009. The model discrepancy in fall 2010 was carried over somewhat in 2011, with the model predicting somewhat cooler conditions in the hypolimnion in the spring-summer of 2011 than observed. As a result, the %RE in temperature at 12-meter depth was slightly higher (%RE of 8.7%), with the mean predicted value (12.6°) slightly lower than the mean observed value (13.3°C).

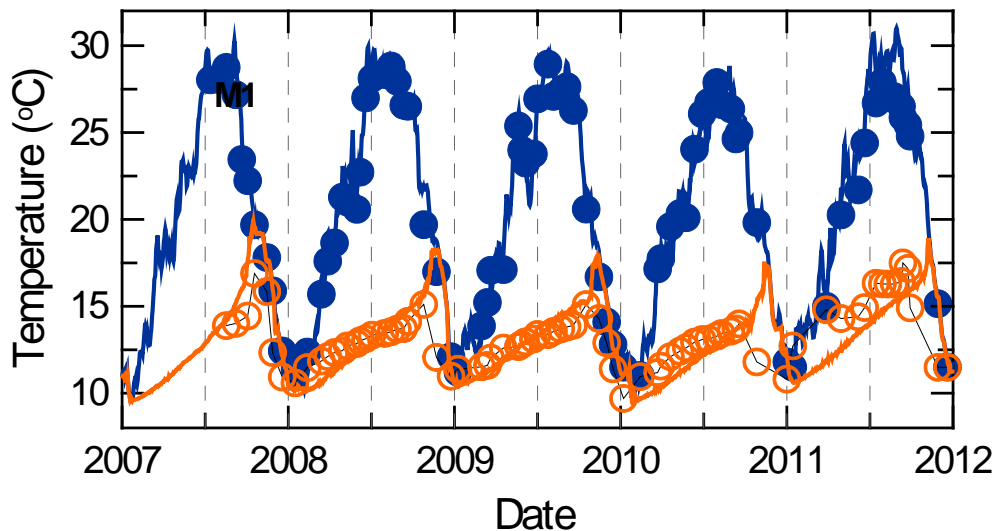
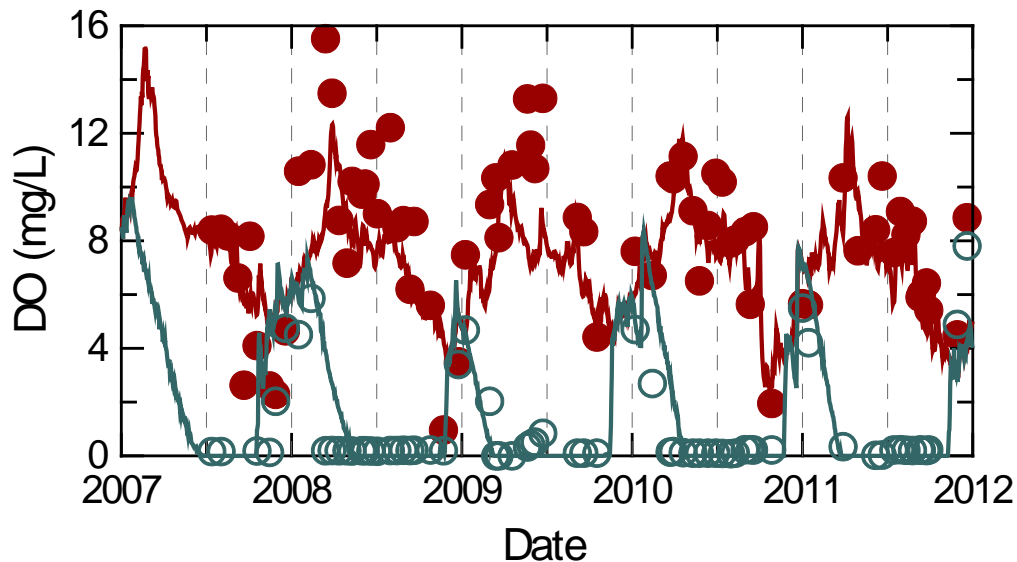


Figure 5-19. Measured and Model-Predicted Temperatures at 2-meter (dark blue solid circles) and 12-meter (orange open circles) Depths for the Calibration Period 2007-2011

#### 5.4.4.3 Dissolved Oxygen

Unlike temperature, which can be simulated using ELCOM alone, prediction of DO requires CAEDYM due to the regulation of DO by biological and chemical processes. DO is specifically a function of photosynthetic production and respiratory loss by algae, sediment oxygen demand, microbial respiration in the water column, chemical demand by reduced substances, and other processes. Dissolved oxygen in Canyon Lake is highly dynamic, with concentrations in the epilimnion often supersaturated in the spring and very low in the fall following turnover. This can be seen in Figure 5-19, where the DO concentration at 2-meter depth (solid symbols) reached nearly 14 mg/L or more in early spring in 2008 and late spring in 2009, but also dropped to 2 mg/L or lower in the surface water in the late fall following turnover (**Figure 5-20**). The model (dark red solid line) reproduced the trends reported in DO, with minima in the late fall and maximum values generally seen in the spring (Figure 5-20). The model did not always predict quite as high values in the summer as reported, and yielded a slightly lower mean predicted DO concentration at 2-meter depth value of 7.28 mg/L compared with the mean observed value of 8.14 mg/L, and a %RE of 22.7%. Considerable effort was dedicated to calibrating the model while also retaining available laboratory measurements of SOD, internal nutrient loading rates, and other factors.



**Figure 5-20. Measured and Model-Predicted Dissolved Oxygen at 2-meter (dark red solid circles) and 12-meter Depths (blue-green open circles) for the Calibration Period 2007-2011**

Dissolved oxygen at 12-meter depth also exhibited strong seasonal variation, with concentrations often approaching saturation during the winter months when the lake was well-mixed vertically, but declining rapidly in the early spring and being typically < 0.1 mg/L most of the summer (Figure 5-20). The model reproduced this trend quite well, and yielded a mean DO concentration at 12-meter depth of 1.27 mg/L, in pretty good agreement with the observed mean value of 0.99 mg/L, although %RE was quite high (75.6%) because of the large number of very low concentrations where even a modest difference yields a high relative error. Moreover, measurement values are also prone to error at very low concentrations, and concentrations near or below 1.0 mg/L exert similar biological effects so this model outcome is considered adequate.

#### 5.4.4.4 Total Nitrogen

The observed concentrations over time of TN at 2-meter depth and ammonium-N at 12-meter depth are presented in **Figure 5-21** (solid blue symbols= TN at 2 m; open purple triangles= $\text{NH}_4\text{-N}$  at 12 m). Most nitrogen in the hypolimnion during periods of stratification is expected to be in the ammonia-N form. TN concentrations in the epilimnion tended to range from about 1-3 mg/L, although values < 0.5 and > 4 mg/L were also reported (Figure 5-21). The data showed seasonal trends in epilimnetic TN involving higher concentrations in the fall following lake overturn and with subsequent external loads from wet season runoff, followed by lower concentrations later in the spring and summer. This trend was difficult to reproduce in the model, however (%RE of 45.3% in Main Lake), and the predicted mean concentration of 1.24 mg/L was lower than the observed value. Ammonium-N in the hypolimnion was negligible during the winter following overturn of the water column while  $\text{NH}_4\text{-N}$  increased each spring and summer as a result of internal recycling and accumulation in the bottom waters (Figure 5-21). The model captured these general trends and yielded a mean predicted concentration of 0.85 mg/L in good agreement with the measured values on the same dates (0.81 mg/L) (n=61). The %RE between predicted



and observed values across the available data for 2007-2011 was 41.3%. Observed nutrient concentrations were compared with predicted values at 12-meter depth, although actual depth of samples tended to vary somewhat, so some error is thought to arise from that assumption.

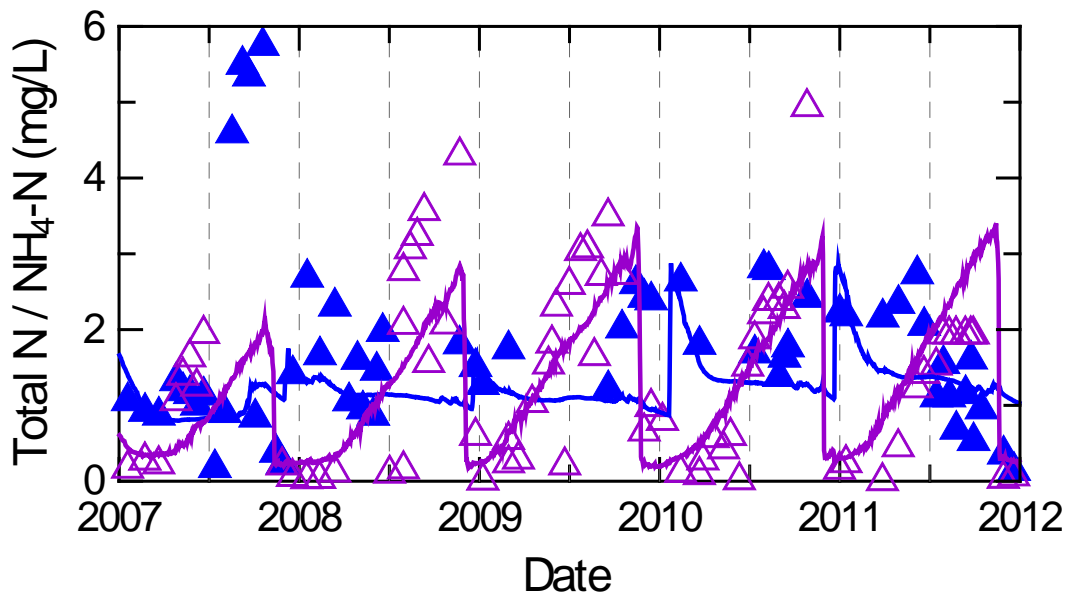


Figure 5-21. Measured and Model-Predicted Total Nitrogen at 2-meter (blue, solid triangles) and 12-meter Depths (purple, open triangle) for the Calibration Period 2007-2011

#### 5.4.4.5 Total Phosphorus

TP in the epilimnion (2 m depth) exhibited temporal differences although a clearly defined seasonal trend was not readily evident, with concentrations ranging from 0.07 – 1.74 mg/L and a mean of 0.64 mg/L (Figure 5-22, solid red circles). The model did a good job of reproducing the average concentration of TP (0.66 mg/L), but did not capture the variability present in the data (modeled range of 0.40 – 1.2 mg/L) (Figure 5-22), with a %RE of 39.5%. Dissolved  $\text{PO}_4\text{-P}$  concentrations at 12-meter depth exhibited clear seasonal trends similar to  $\text{NH}_4\text{-N}$ , with concentrations increasing each spring and summer to reach a maximum value in the fall immediately prior to turnover; concentrations often reached or exceeded 2 mg/L before fall sharply with mixing of the water column (Figure 5-22). The model predicted this seasonal trend but tended to underestimate the concentrations somewhat (mean measured and predicted concentrations of 1.15 and 0.71 mg/L, respectively, with a %RE of 47.2%). As with  $\text{NH}_4\text{-N}$ , some error between predicted and observed  $\text{PO}_4\text{-P}$  is also thought to arise from some differences in sampling depth.

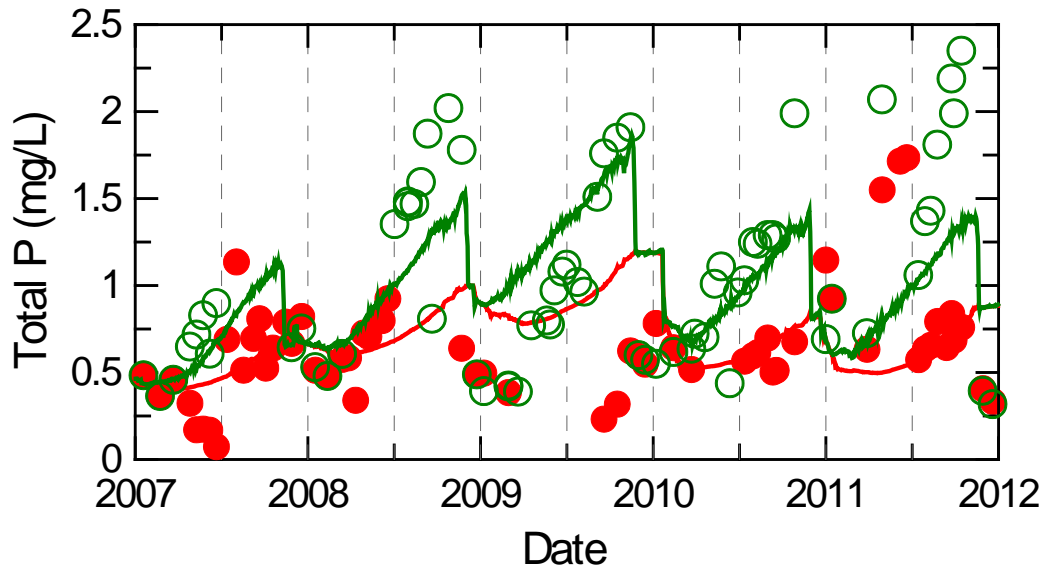


Figure 5-22. Measured and Model-Predicted Total Phosphorus at 2-meter (red solid circles) and 12-meter Depths (dark green open circles) for the Calibration Period 2007-2011

#### 5.4.4.6 Chlorophyll-*a*

Chlorophyll-*a* concentrations exhibited strong seasonal differences, with low measured concentrations during the winter and much higher concentrations during the summer (**Figure 5-23**, solid symbols). Sampling was limited to a few dates in the winter of 2008 and 2009, so sampling in 2010 and 2011 offered the most complete sets of annual trends in chlorophyll-*a*. Model predictions reflected these seasonal trends in chlorophyll-*a*, with temporally-averaged concentrations in relative agreement between observed and predicted values (31.2 and 38.8  $\mu\text{g/L}$ , respectively), similar minimum values (2.0 and 3.1  $\mu\text{g/L}$ , respectively), and similar maximum values as well (73.1 and 77.6  $\mu\text{g/L}$ , respectively). Notwithstanding, the timing of the phytoplankton blooms varied in some years, with a high %RE (66.8%). Given the complexity of reproducing the phytoplankton community in such a dynamic lake environment, the capacity to reproduce mean, minimum and maximum values suggests that the model can nonetheless be useful in describing water quality trends if not the specific timing of the blooms.

#### 5.4.5 Water Quality Model Summary Statistics

The model could be calibrated to reproduce water quality for a single year, but disparities between predicted and observed properties generally increased when using a five-year calibration period (2007-2011). The comparatively long simulation period (5-yrs) with markedly different hydrology created extra challenges in simulating water quality in the lake. However, when looking at the five year means for water quality parameters, model results matched well with observed data in both Canyon Lake Main Lake (M1) and Canyon Lake East Bay (E2) (**Table 5-4**). On average, the model predicted similar mean DO and temperature within the hypolimnion of the Main Lake, based on results collected from 12-meter depth below the lake surface.

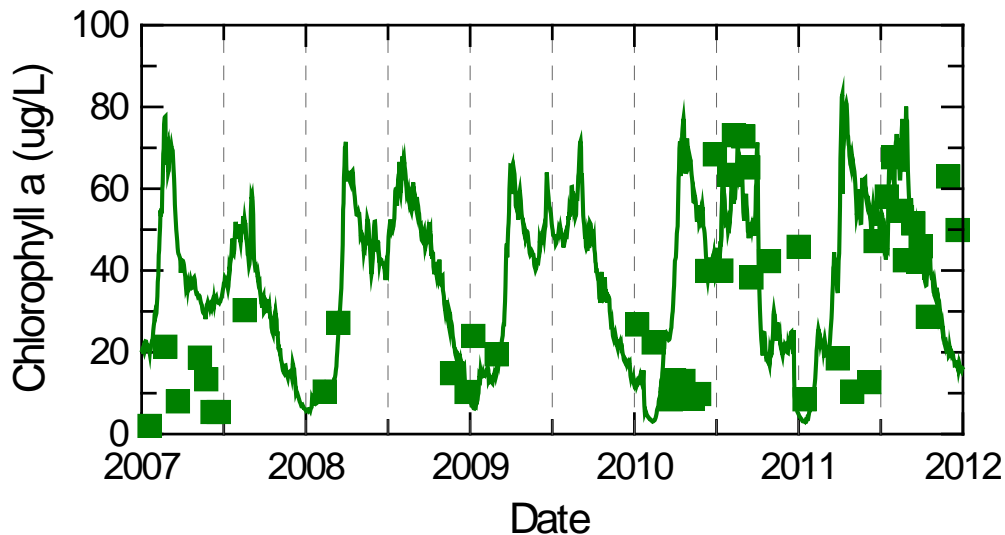


Figure 5-23. Measured and Model-Predicted Chlorophyll-*a* at 2-meter Depth (green solid squares) for the Calibration Period 2007-2011

Table 5-4. Mean Values for Observed and Predicted Water Quality Parameters (Observed / Predicted)

Site	Depth (m)	Temperature (°C)	DO (mg/L)	Chlorophyll- <i>a</i> (µg/L)	Total N (mg/L)	Total P (mg/L)
Main Lake (M1)	Epilimnion (2-meter)	21.5 / 21.3	8.1 / 7.3	31.2 / 38.8	1.82 / 1.24	0.64 / 0.66
	Hypolimnion (12-meter)	13.3 / 12.6	1.0 / 1.3	-	-	-
East Bay (E2)	Epilimnion (1 meter)	-	-	53.3 / 53.7	1.88 / 1.35	0.55 / 0.64

The goodness-of-fit for trends in water quality parameters was assessed by computing the RE of model results with observed data on days when water quality samples were collected for TN, TP and chlorophyll-*a*. The average of REs for all discrete pairs of modeled and measured results for all water quality parameters ranged from 22.7 to 75.6 percent (**Table 5-5**). Discussion is provided above related to the goodness-of-fit for each parameter. Overall, the lake model calibration is considered acceptable given that a reference watershed approach, and not the linkage analysis, is used for estimating allowable external nutrient loads in the revised TMDL.

#### 5.4.6 Reference Condition Scenario Evaluation

The linkage analysis was used to evaluate the water quality conditions in Canyon Lake for a scenario where external loads are reduced to be representative of a reference watershed condition to develop numeric targets for response variables, ammonia-N, DO and chlorophyll-*a*. Section 3.3.3 describes the water quality input data. This scenario was developed for a 15-year (2001-2016) simulation period coinciding with available daily flow data from USGS gauges for the San Jacinto River at Goetz Rd (Sta#11070365) and Salt Creek at Murrieta Road (Sta#11070465).

Reference watershed nutrient concentrations are assumed to occur in the total daily inflow volume to Canyon Lake.

**Table 5-5. Average of Percent Relative Errors Between Discrete pairs (sampled days) of Predicted and Observed Water Quality**

Site	Depth (m)	Temperature (% error)	DO (% error)	Chlorophyll- <i>a</i> (% error)	Total N (% error)	Total P (% error)
Main Lake (M1)	Epilimnion (2-meter)	4.0 (n=80)	22.7 (n=73)	66.8 (n=47)	45.3 (n=61)	39.5 (n=63)
	Hypolimnion (12-meter)	8.7 (n=77)	75.6 (n=68)	-	-	-
East Bay (E2)	Epilimnion (1 meter)	-	-	59.8 (n=66)	39.2 (n=73)	62.1 (n=73)

No changes were made to the Canyon Lake bathymetry or model resolution to run a reference condition scenario. Results of the reference condition model are plotted as time series in **Figure 5-24** for Canyon Lake Main Lake and **Figure 5-25** and Canyon Lake East Bay. Results include lake level, TDS, TP, TN, ammonia-N, DO and chlorophyll-*a*. The following observations were noted from these results:

- For both Main Lake and East Bay, algal productivity follows a seasonal pattern with an initial bloom toward the end of the wet season (February/March) that extends until the fall when days get shorter and wet weather provides some flushing of algae.
- Limited inter-annual variability exists in the magnitude of chlorophyll-*a* in both lake segments for a reference watershed condition.
- Apparent differences in nitrogen and phosphorus patterns can be attributed to both internal and external loading. Flux rates for nitrogen are about three times greater than for phosphorus, and this same proportion is reflected when comparing modeled depth average concentrations for nitrogen and phosphorus during dry seasons.
- Water column TP concentration resulting from sediment flux over the dry season is similar to the assumed concentration for external runoff inflows in a reference watershed condition; therefore, variability in phosphorus is much lower over the simulation period.
- Ammonia-N flux rates support a dry season depth average of about 0.5 mg/L, which is half of the TN assumed for external runoff inflows in a reference watershed condition. Therefore, external watershed runoff provides a considerable rise in water column TN concentration, especially for storm events with volumes in excess of the storage capacity (i.e., flushing the entire standing volume one or more times over a single storm).

**Figures 5-26 through 5-29** illustrate vertical profiles of the ELCOM-CAEDYM Model results for DO, nutrients, and chlorophyll-*a* comparing existing with reference watershed conditions based on output from Station M1 in the Main Lake. Note the difference in magnitudes for the color

ramps between existing and reference watershed conditions. One key observation is the persistence of low DO (below 5 mg/L) in the hypolimnion of Main Lake for both existing and reference watershed conditions. For nitrogen and phosphorus, the model estimated the greatest fluxes to occur in different years, with nitrogen fluxes increasing in dry seasons immediately following wet years and phosphorus fluxes greatest in drought years. Chlorophyll-*a* was found to grow deeper in the water column toward the end of the growing season which served to reduce the area of anoxia.

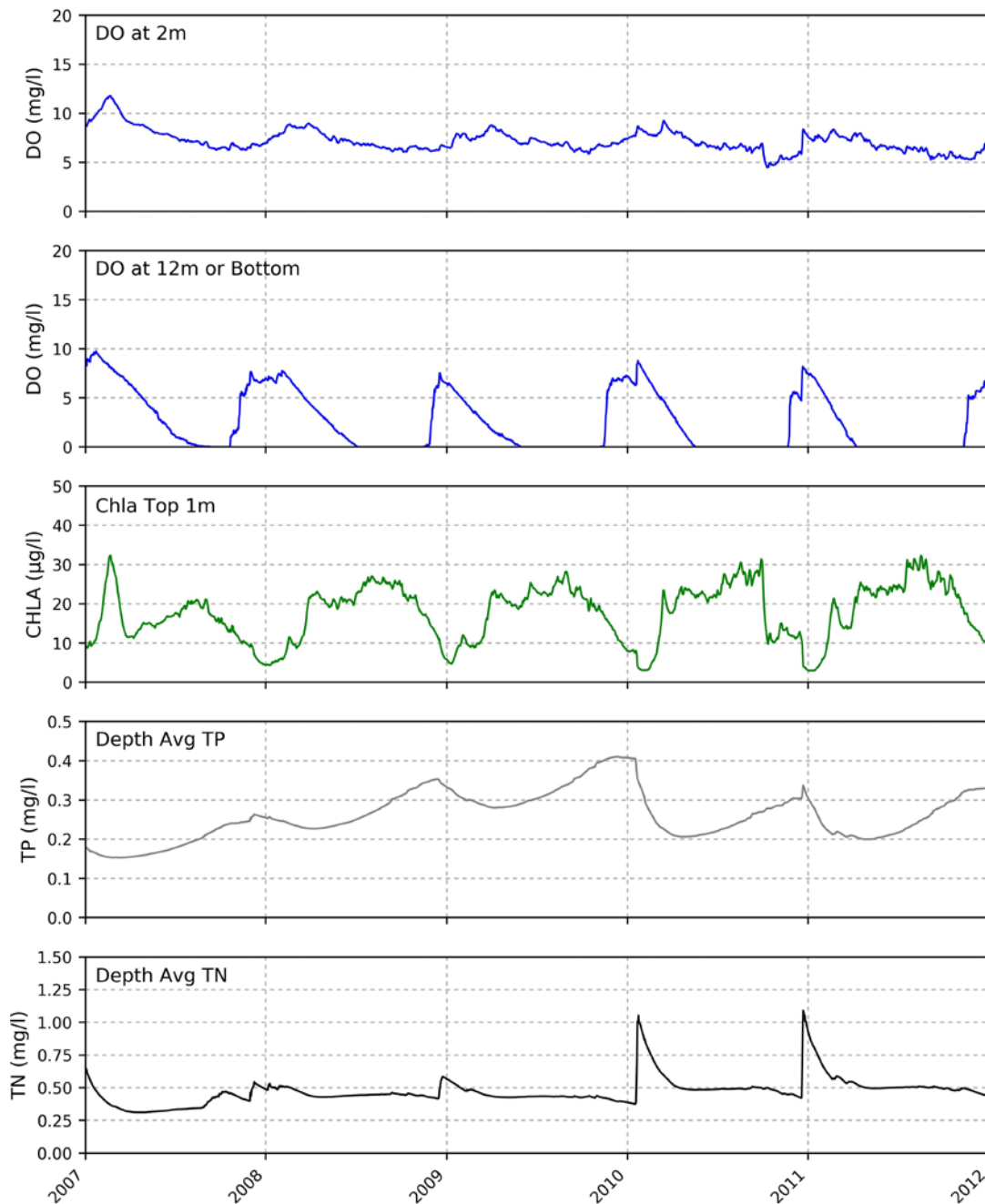


Figure 5-24. Time Series Output of Water Quality Parameters for Reference Condition Simulation for Canyon Lake Main Lake (2007 - 2011)

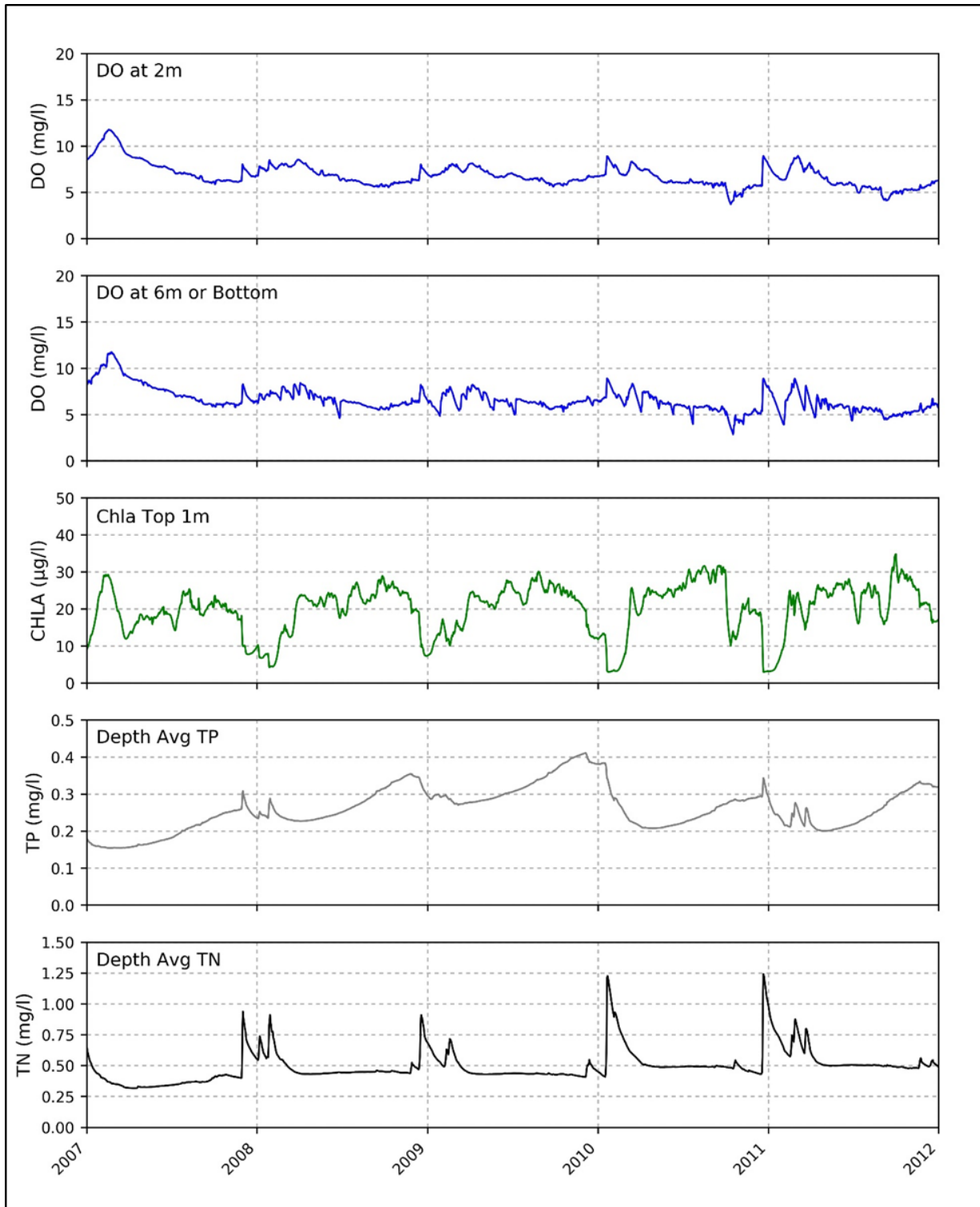
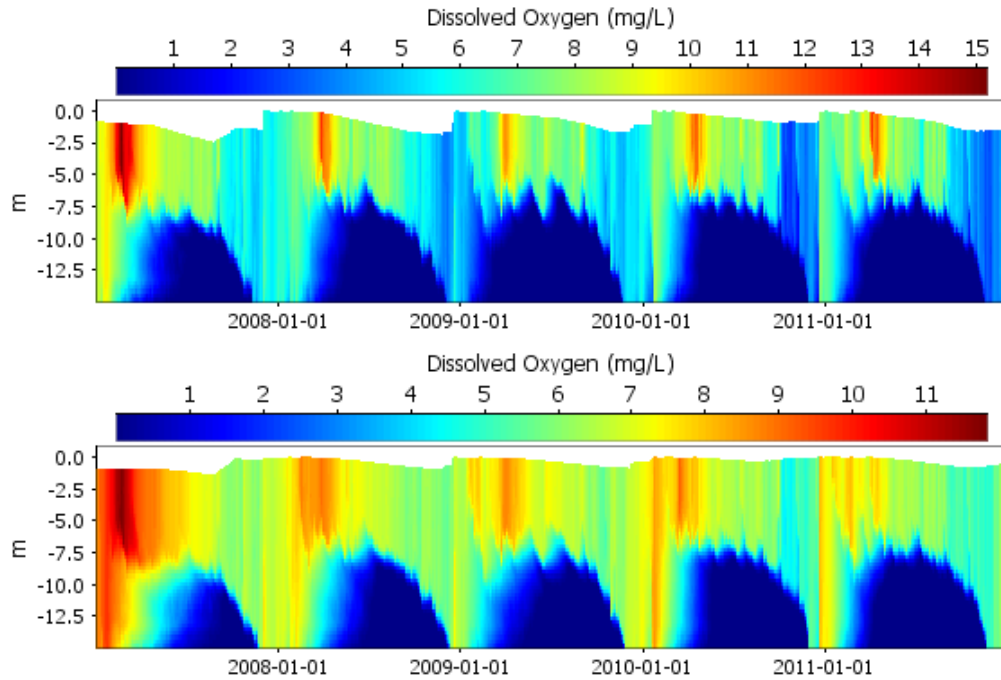
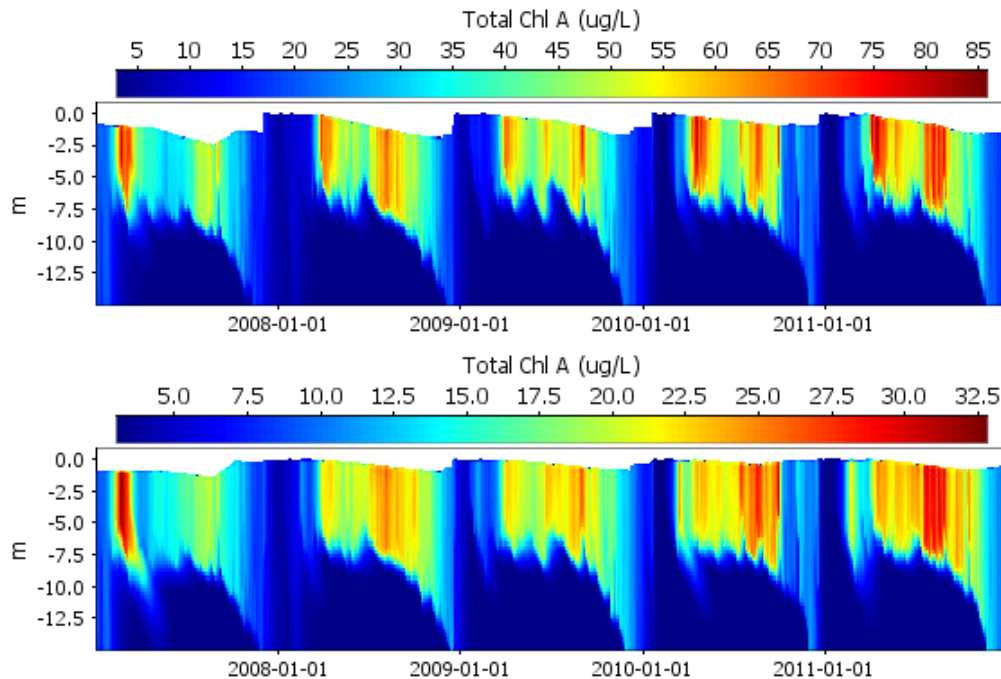


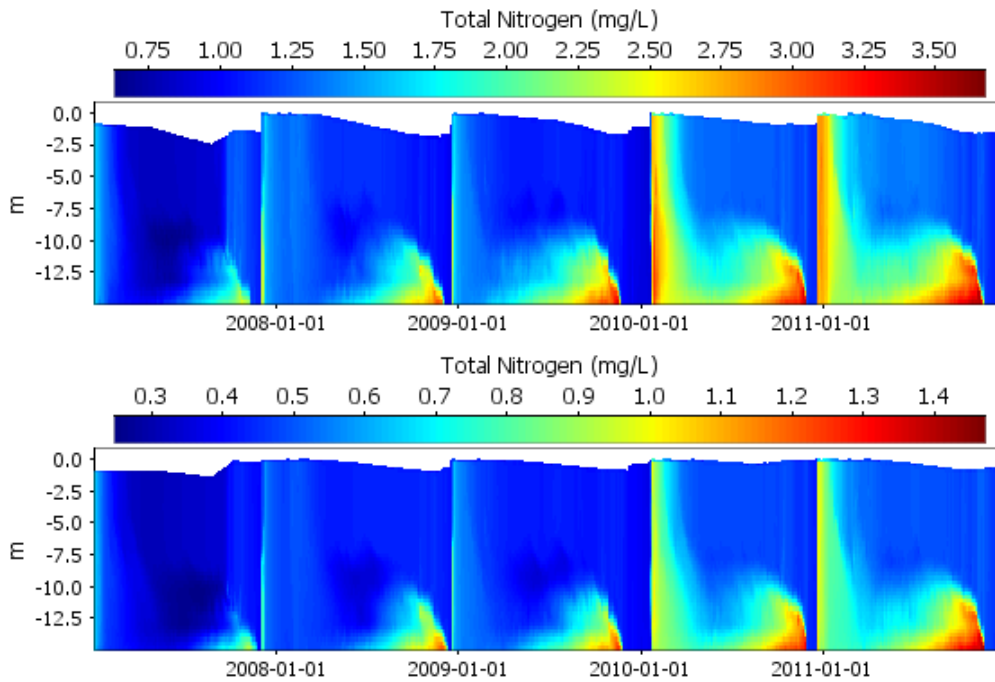
Figure 5-25. Time Series Output of Water Quality Parameters for Reference Condition Simulation for Canyon Lake East Bay (2007 - 2011)



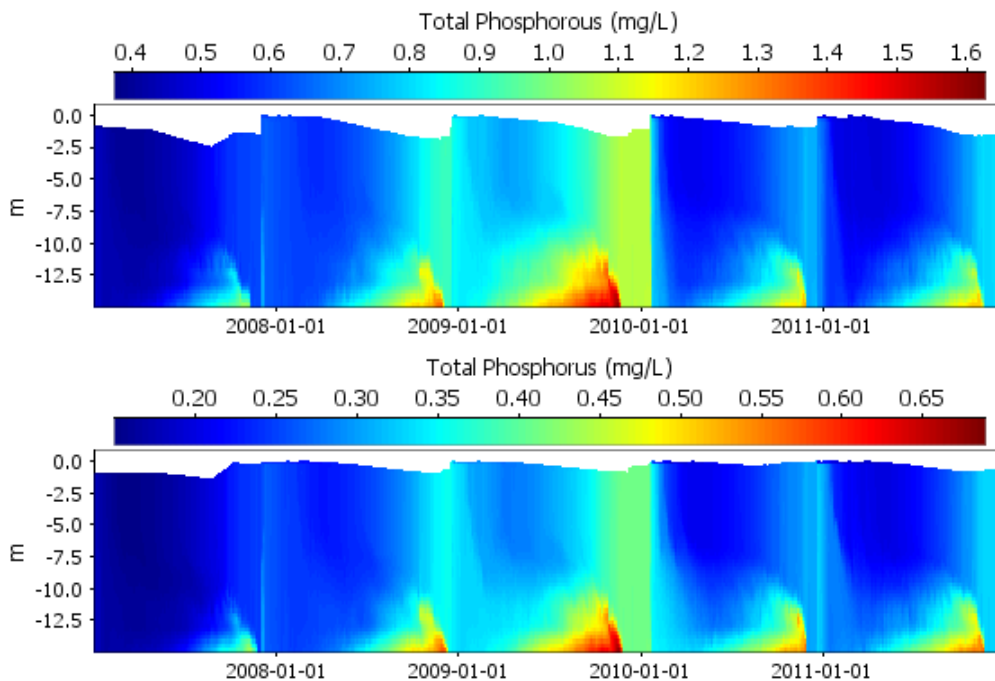
**Figure 5-26. Vertical Profiles of ELCOM-CAEDYM Model Results for Dissolved Oxygen Comparing Existing Conditions (top) with Reference Watershed Conditions (bottom) Based on Output from Station M1 in Main Lake of Canyon Lake**



**Figure 5-27. Vertical Profiles of ELCOM-CAEDYM Model Results for Chlorophyll-*a* Comparing Existing Conditions (top) with Reference Watershed Conditions (bottom) Based on Output from Station M1 in Main Lake of Canyon Lake**



**Figure 5-28. Vertical Profiles of ELCOM-CAEDYM Model Results for Total Nitrogen Comparing Existing Conditions (top) with Reference Watershed Conditions (bottom) Based on Output from Station M1 in Main Lake of Canyon Lake**



**Figure 5-29. Vertical Profiles of ELCOM-CAEDYM Model Results for Total Phosphorus Comparing Existing Conditions (top) with Reference Watershed Conditions (bottom) Based on Output from Station M1 in Main Lake of Canyon Lake**



## References – These will be added to Section 9 of TMDL Report

Anderson, M.A. 2015. *Surface Elevation and Salinity in Lake Elsinore: 1916-2014*. Technical Memorandum. Prepared for Santa Ana Watershed Project Authority (SAWPA) on behalf of the LECL Task Force. April 26, 2015.

Anderson, M.A. 2012. *Evaluation of Long-Term Reduction of Phosphorus Loads from Internal Recycling as a Result of Hypolimnetic Oxygenation in Canyon Lake*. Technical Memorandum: Task 2. Prepared for Santa Ana Watershed Project Authority (SAWPA) on behalf of the LECL Task Force. April 22, 2012.

Anderson, M.A. 2010. *Bathymetric, Sedimentological and Retrospective Water Quality Analysis to Evaluate Effectiveness of the Lake Elsinore Recycled Water Pipeline Project*. Final Report to the Lake Elsinore & San Jacinto Watershed Authority. 50 pp.

Anderson, M.A. 2007. *Predicted Effects of In-Lake Treatment on Water Quality in Canyon Lake*. Final Report submitted to the San Jacinto River Watershed Council. 31 pp. + Appendix.

Anderson, M.A. 2001. *Internal Loading and Nutrient Cycling in Lake Elsinore. Final Report*. Santa Ana Regional Water Quality Control Board. 52 pp.

Chapra, S. C. 1997. *Surface Water-Quality Modeling*, McGraw-Hill Series in Water Resources and Environmental Engineering.

Dyal, K. and M.A. Anderson. 2003. Unpublished data, UCR

Hipsey, M.R., J.R. Romero, J.R., J.P. Antenucci, J.P. and D. Hamilton. 2006. *Computational Aquatic Ecosystem Dynamics Model: CAEDYM v2. Science Manual v2.3*. Center for Water Research, University of Western Australia. January 16, 2006.

Thomann R. and J. Mueller. 1987. *Principles of Surface Water Quality Modeling and Control*. Manhattan College, HarperCollins Publishers, New York.

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