Predicted Effects of Restoration Efforts on Water Quality in Lake Elsinore: Model Development and Results

Final Report

Submitted to:

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12 March 2006

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Executive Summary

A model was developed to predict the impacts on water quality in Lake Elsinore from recycled water addition, groundwater inputs, aeration, lake level stabilization, and carp removal. Results from recent nutrient budget and internal loading studies, fish population surveys, monitoring data collected over the past decade, and other studies were used to develop and parameterize the model. The differential form of the model was found to accurately reproduce observed water quality over the period from 1993-2004. Monte Carlo techniques were incorporated into a steady-state form of the model to develop probabilistic descriptions of predicted water quality and estimates of uncertainty in model predictions.

The steady-state form of the model was used to assess the impacts of recycled water and groundwater inputs to the lake. Assuming Lake Elsinore received average rainfall and runoff each year, the lake level was predicted to stabilize at 1222.7 ft above MSL, have a mean depth <1 m, and TP concentrations near 1 mg/L. Against this shallow, hypereutrophic lake condition, addition of supplemental water increased surface elevation and mean depth and decreased predicted TP concentrations. About 6000 af/yr of supplemental water was needed maintain the lake level at 1240 ft. The source of the supplemental water affected the predicted TP levels; 6,000 af/yr inputs from EVMWD, EMWD or the island wells yielded median predicted TP concentrations in the lake of 0.49, 0.28 and 0.28 mg/L, respectively. Chemical phosphorus removal at the EVMWD plant to 0.5 mg/L lowered the median predicted TP concentration in Lake Elsinore to 0.35 mg/L. Even under conditions of high annual inflows with high TP concentrations, internal loading dominated the predicted P budget for the lake.

Other restoration efforts also affected the predicted steady-state water quality in the lake. Aeration was found to very favorably reduce median predicted TP and chlorophyll concentrations; for example, aeration sufficient to reduce the internal recycling rate by 30% was predicted to lower the TP concentration from 0.377 mg/L to 0.148 mg/L and reduce chlorophyll levels from 395 μg/L to 102 μg/L. Carp removal was less effective than aeration at improving water quality (e.g., a 75% reduction in carp biomass lowered the median predicted TP concentration to 0.26 mg/L).

Introduction

Lake Elsinore is a shallow, polymictic and highly eutrophic lake that is subject to wide variations in lake elevation, nutrient concentrations, DO levels and algal productivity. Recent studies have demonstrated the importance of internal sources of nutrients to the overall nutrient budget in Lake Elsinore (Montgomery-Watson, 1997; Anderson, 2001). For example, a nutrient budget developed for 2001 with limited runoff (2520 af) indicated that external loading accounted for only 626 kg of total P inputs as compared with 33,160 kg of P input to the water column in the dissolved form based upon core-flux measurements (Anderson, 2001).

The mass of particulate P recovered in sediment traps exceeded the known inputs, however, and thus implicated resuspension as an additional and important source of P that delivered an estimated 50,606 kg of P to the water column. The shallow mean depth of the lake at the time of these measurements (4.3 m) suggests that wind energy may have resuspended sediments during periods of high winds (Carper and Bachman, 1984; Scheffer, 1998). Moreover, the intense evaporative demand in the region removes an average of 1.43 m of water each year from the lake; thus during droughts and other periods of low total runoff to the lake, the lake experiences rapid declines in lake elevation and mean depth. Declining lake level results in exposure of new sediments to wind resuspension as the wave-mixed layer advances lower into the lake. Thus it follows that lake elevation is an important factor in understanding the importance of wind-driven sediment resuspension.

In addition to sediment resuspension by wind, bioturbation by benthivorous fish is thought to be important. A recent fish population survey puts carp levels at about 700,000 – 1.5 million, or 1 fish for every 8 - 17 m^2 of lake bottom (EIP, 2004). Carp and other benthivorous fish are widely recognized for their ability to disturb and resuspend substantial amounts of sediment during feeding (Scheffer, 1998).

Several efforts are planned or underway to improve water quality in the lake, including lake level stabilization by supplemental recycled water and groundwater inputs, aeration via axial flow pumps to improve DO levels in the subsurface to minimize fish kills and reduce internal P recycling, and carp removal to reduce sediment resuspension and associated effects on internal nutrient recycling, remove P and biomass, and help balance the fishery.

The objective of this study was to evaluate the relative changes in water quality that can be expected from these and other lake management efforts through development and application of a water quality model for Lake Elsinore.

Approach

A model has been developed that incorporates known processes affecting total P (TP) levels and resulting chlorophyll and transparency levels in the lake (Fig. 1). The model is an extension of a preliminary model originally developed for the lake as part of an agreement with CH2M-Hill that has also been used in recent TMDL efforts.

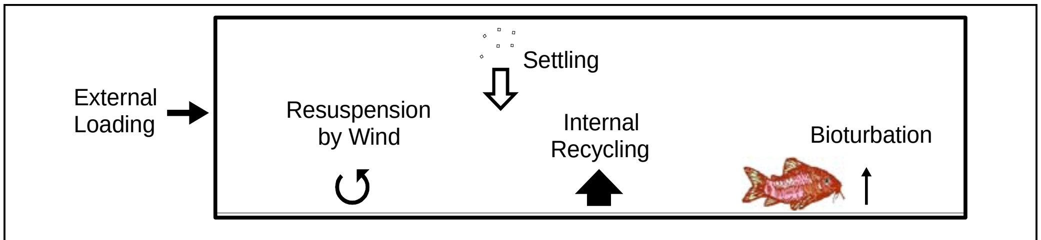

Fig. 1. Processes affecting nutrient levels in Lake Elsinore.

External loading, internal recycling, and resuspension of bottom sediments by wind and by bioturbation are all source terms that deliver dissolved and/or particulate forms of P to the water column. Since the lake has not released water through outflow since the El Nino of 1995, settling is functionally the only process by which P is lost from the water column. Normalized to unit surface area, the P cycle can be described as a series of fluxes that, when at steady-state, requires that the sources are balanced by losses. Thus, it can be written that:

$$J_S = J_{EL} + J_W + J_{IL} + J_B \tag{1}$$

where J_S is the settling flux rate, J_{EL} is the external loading rate normalized to lake area, J_W is the resuspension flux due to wind, J_{IL} is the internal loading rate, and J_B is the flux due to bioturbation by fish (principally carp). All flux rates are expressed in units of mg/m^2/d.

External loading (J_{EL})

The external loading of nutrients to the lake arises from inputs from the watershed, which vary strongly depending upon precipitation inputs to the region. El Nino events have typically produced substantial runoff, while average and below average rainfall (La

Nina years) yield very little runoff that is largely restricted to the local watershed. In an effort to provide a more consistent source of water to the lake and thus limit net evaporative losses during periods of low rainfall, recycled water addition was started in July 2002. Groundwater inputs from the island wells have also recently begun. The areally-normalized loading of P from these external sources, J_{EL}, can be written as:

$$J_{EL} = \frac{\sum_{i=1}^{n} Q\ C_i + PA_w RC_w}{A} \qquad (2)$$

where Q_i and C_i are the daily flow (m^3/d) and average TP concentration (mg/m^3) of external source *i* (taken here as recycled water from EVMWD, recycled water from EMWD, or groundwater from the island wells), P is the precipitation rate (m/d), A_w is the area of the local watershed (m^2), R is the runoff coefficient representing the fraction of rainfall that runs off into the lake from the local watershed (m/m), and C_w is the concentration of P in the runoff (mg/m^3). The upper watershed, defined here as the watershed upstream of the Canyon Lake dam, is accounted for in these calculations, through the San Jacinto River flow reported at the USGS gage #11070500. This contribution is especially important during El Nino years, while supplemental water from recycled water and groundwater sources would be less needed.

The total annual flow of recycled water over the period July 2002 – June 2003 from EVMWD was 2018 af, while EMWD released 1498 af over this period (Veiga-Nascimento and Anderson, 2004). A target flow from EVMWD of up to 4500 af/yr is planned, although volumes will be pumped as available. The average TP concentration over this period was 1.99±1.03 mg/L, with all phosphorus in the dissolved form; adding additional treatment to lower these levels is under consideration (CH2M, 2004).

The volume-weighted average runoff concentration of TP was 0.20 mg/L during the 2001 water year (Anderson, 2001). The atmospheric deposition rate for P is not known, but is thought to be very low compared with other external and internal sources.

The runoff coefficient for the local watershed was estimated from lake elevation changes following precipitation events. Rainfall and lake elevation changes were evaluated for 3 storms in 2001 and used to derive runoff coefficients for each of these events (Table 1).

The lake elevation increased by 2.26 – 3.36x the amount of precipitation falling directly upon the lake surface, indicating contributions from the local watershed. (These storms were not of sufficient intensity to generate any significant release from Canyon

Lake.) With a local watershed area of 54 km^2 draining directly to the lake (TetraTech, 2003), one calculates total rainfall volumes of 1655, 2329 and 477 af that fell within the local watershed during those 3 events (excluding that falling directly on the lake surface). With an average lake elevation of 1241.75 ft, the lake surface area is very close to 3000 acres; thus the increases in lake elevation due to local watershed inputs for those 3 events correspond to 468, 676 and 253 af, respectively, for a total of 1397 af for the 3 storms (very close to the median annual value reported by CH2M-Hill for 1991-2000 (CH2M, 2004)). The runoff coefficients, defined simply as the volume of runoff to the volume of rainfall striking the local watershed (assuming negligible subsequent flow from the back basin), were thus calculated as 0.28, 0.29 and 0.53 (Table 1). As is expected, the runoff coefficient increased over the 3 storms, reflecting both increased antecedent moisture levels in the soils and shorter time intervals between the storms.

Table 1. Rainfall, lake elevation and derived runoff coefficients for 3 storms in 2001.

Date (2001)	**Rainfall (mm)**	**△Lake Elev (mm)**	**△Lake Elev/Rainfall**	**Runoff Coeff**
Feb. 12-14	37.8	85.3	2.26	0.28
Feb. 23-28	53.2	121.9	2.29	0.29
Mar. 6-10	10.9	36.6	3.36	0.53

Bioturbation (J_B)

A number of researchers have demonstrated a linear increase in suspended solids concentrations and resuspension flux with increased density of benthivorous fish (e.g., Meijer et al. 1989; Breukelaar et al., 1994). In the study by Breukelaar et al. (1994), experimental data for a series of small ponds with varying densities of benthivorous carp were well fit by a line through 0 with a slope of 0.24 $g/m^2/d$ per kg/ha. Assuming this relationship also reasonably holds for carp in Lake Elsinore, and further assuming a total population of 700,000 – 1,500,000 carp in about 3000 acres for a density of 233 - 500 fish per acre (or 576-1235 per ha), at a median mass of about 860 g (EIP, 2004), which corresponds to 496 – 1062 kg/ha, one calculates a predicted bioturbation rate of 119 – 255 $g/m^2/d$. A previous assessment of labile inorganic P following Rydin and Welch (1999) found 273±32 μg/g sediment against a total sediment P content of 916±73 μg/g in the fine organic type III sediments was potentially labile, although the short lifetime of suspended sediment in the water column would limit the net amount released from the particles. A smaller amount was loosely sorbed (85 μg/g). A linear partition coefficient

(K_d) describing the amount of loosely sorbed P (S) in local equilibrium with the porewater at concentration (C) is given by:

$$K_d = S/C \qquad (3)$$

where K_d is the partition coefficient (L/g), S is the sorbed concentration (mg/g) and C is the solution concentration (mg/L). At a loosely sorbed concentration of 85 μg/g or 0.085 mg/g and a porewater concentration of 6.0 mg/L, one calculates a K_d of 0.014 L/g.

Peeper measurements have shown that the porewater dissolved P (and NH_4-N) concentrations increase with depth beneath the sediment surface, from quite low values (often 0.1 – 0.2 mg/L) to 2 – 6 mg/L approximately 10 cm deep into the sediments (Anderson, 2001). Since most benthic organisms are located in the upper few cm of sediment, we will assume that carp typically forage and disturb the uppermost 2-3 cm of sediment. By linear extrapolation of porewater profiles from a number of sites and throughout 2000-2001, the SRP concentration averaged 0.71±0.35 mg/L in the uppermost 2-3 cm. With an average sediment dry-weight of approximately 0.12 g/g, this corresponds to 7.3 mL of porewater released per dry-weight g sediment disturbed; with a typical porewater concentration of 0.71 mg/L, this corresponds to release of 0.0052 mg porewater SRP per g resuspended sediment (0.0052 mg/g). Further assuming that essentially all of the loosely sorbed P, calculated from eq 3 using the K_d of 0.014 L/g and C of 0.71 mg/L (0.010 mg/g), would also be released due to the mixing into the low concentration overlying water, one calculates a total of 0.015 mg P is released upon resuspension of 1 g of sediment (dry-weight) (0.015 mg P/g sediment).

With information about carp population density, the findings of Breukelaar et al. (1994), and the bioavailable fraction of P released upon disturbance, one can propose that the flux of bioavailable P due to bioturbation can be written as:

$$J_B = fPMB \qquad (4)$$

where J_B is the bioturbation flux rate of bioavailable P (mg/m^2/d), f is the normalized sediment resuspension flux rate of 0.24 g/m^2/d per kg carp/ha (Breukelaar et al., 1994), P is the fish population in the lake (fish/ha), M is the average mass of the carp (kg/fish), and B represents the bioavailable P fraction of the resuspended sediment (mg/g).

Wind Resuspension (J_W)

The potential for wind-driven resuspension of bottom sediments at Lake Elsinore is high given its shallow depth, relatively long fetch and periodic strong winds. For example, elevated non-algal turbidity levels were found in the late fall of 2003 that coincided with an increase in the square of the windspeed, which is directly proportional to wind shear acting on the lake surface (Fig. 2). Non-algal turbidity was estimated from total suspended solids (TSS) concentrations using measured chlorophyll values assuming that chlorophyll comprises about 1.5% dry-weight of an algal cell (APHA, 1998).

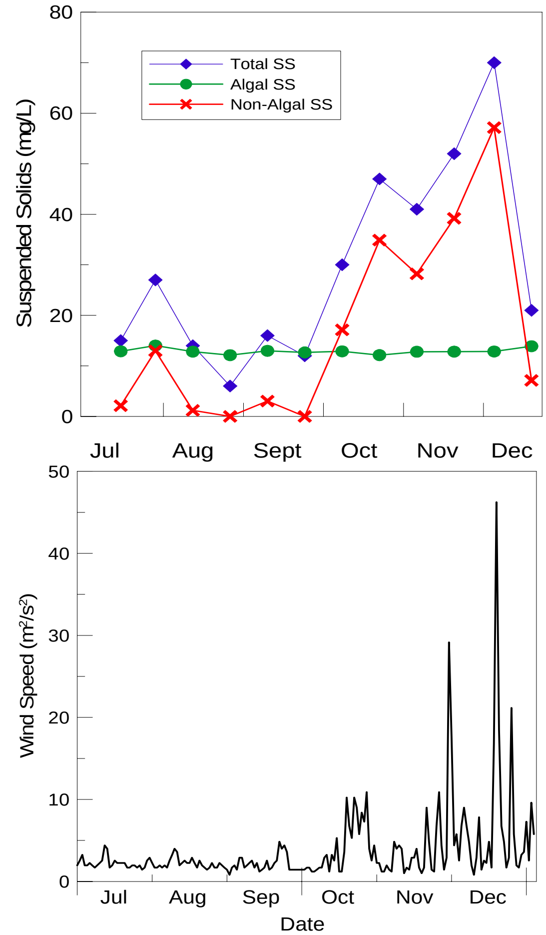

Fig. 2. Temporal trends in a) suspended solids concentrations and b) square of windspeed.

The potential for wind resuspension can be evaluated theoretically as well. Specifically, resuspension can occur when deep-water waves enter water shallower than one-half the wave length (Bloesch, 1995). The wavelength, L, of a deepwater wave is related to its period, T, by the relation:

$$L = \frac{gT^2}{2\pi} \quad (5)$$

where g is the gravitational constant (Martin and McCutcheon, 1999). A wave's period can be estimated using the empirical equation developed by the US Army Coastal Engineering Research Center (Carper and Bachmann, 1984) that states:

$$T = \frac{2.4\pi U \tanh\left[0.077\left(\frac{gF}{U^2}\right)^{0.25}\right]}{g} \quad (6)$$

where U is the wind speed and F is the fetch.

Using these relationships, the wind-mixed depth, taken as one-half the wavelength, *L* (Martin and McCutcheon, 1999) was calculated for wind speeds of 2.5, 5 and 10 m/s assuming a 3 km fetch (Table 2).

Table 2. Predicted wave properties as a function of windspeed for a 3 km fetch.

Windspeed (m/s)	**Wave Period (s)**	**Wavelength (m)**	**Wind-Mixed Depth (m)**
2.5	1.08	1.8	0.9
5.0	1.63	4.1	2.1
10.0	2.37	8.8	4.4

These lower wind speeds represent the range in typical daily average wind speeds often found at the lake, while the higher wind speed of 10 m/s is a relatively frequently observed sustained wind speed during storms, Santa Ana winds, and other meteorological conditions. As one can see, under relatively low wind speeds, resuspension due to oscillatory horizontal motion immediately above the sediments at a wind speed of 2.5 m/s is expected only at depths <1 m. This increases to 2.1 m at wind speeds of 5 m/s, and to a depth of 4.4 m when sustained wind speeds reach 10 m/s. The relationship between fetch, windspeed and critical or mixing depth can be seen more fully in Fig. 3.

Using bathymetric data, one can then estimate the area of lake bottom sediments that could potentially be mobilized by wind (Carper and Bachmann, 1984). For example, using the bathymetric data developed at lake elevation near 1242 ft and with winds principally out of the WSW, the eastern shore possesses the greatest potential for sediment resuspension, with sediment as deep as approximately 4 m potentially being resuspended. Under these conditions, it is estimated that about 4% of the lake bottom will occur within the wind-mixed region. Recognizing that the finer, organic sediments are the most readily mobilized, it is also instructive to consider that proportion of the lake bottom comprised of type II or III sediments (Anderson, 2001) that may be actively resuspended (i.e., occur within the wind-mixed region) (Fig. 4). The percent area of

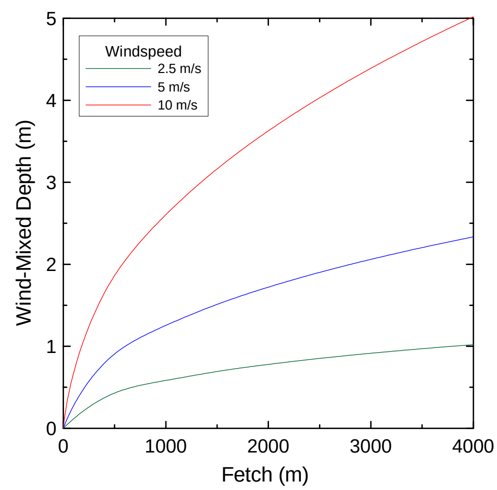

Fig. 3. Mixed layer depth as function of windspeed and fetch.

bottom sediments potentially resuspended will vary strongly depending upon lake elevation, with relatively little resuspension at high elevations and high mean depths and extensive resuspension potential at low surface elevations and depths (Fig. 4). Bengtsson and Hellstrom (1992) have previously demonstrated a linear increase in suspended solids concentrations with % area of erosion.

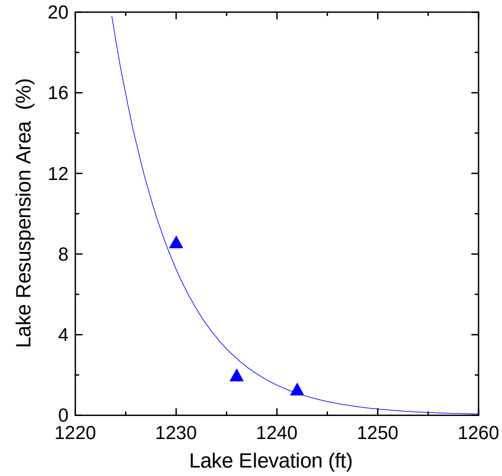

Fig. 4. Area of lake bottom prone to resuspension (type II and III sediments only) as a function of lake elevation.

The data in Fig. 4 were thus used, in conjunction with sediment trap and other nutrient budget data (Anderson, 2001), to estimate an elevation-dependent resuspension flux rate. The non-linear equation from Fig. 4 was used to predict the area of resuspension within the lake; following Bengtsson and Hellstrom (1992) we assume that the areally-averaged whole-lake resuspension flux rate varies linearly with increasing area prone to resuspension (A). Using the data from Fig. 4, the area prone to wind resuspension (A_r) is given by:

$$A_r = 9.27x10^{82}e^{-0.1574*Elev} \quad (7)$$

The data from Bengtsson and Hellstrom (1992), then, allows an empirical relationship to be written that relates the wind-driven bioavailable P resuspension flux rate (J_w) (mg P/m^2/d) to the sediment resuspension rate (w_r), the lake area subject to wind-resuspension (A_r) that varies with surface elevation (Fig. 4), and bioavailable sediment P fraction (B):

$$J_W = w_r A_r B \quad (8)$$

The sediment resuspension rate (w_r) was calculated using the TP sediment resuspension rate estimated from the nutrient budget developed for 2000-2001 (Anderson, 2001). Specifically, 50,606 kg of TP was taken as the annual mass resuspended, with 30% of that bioavailable. This resuspended mass for this period corresponds to a total areally-averaged rate of 3.3 mg P/m^2/d that is attributed to both bioturbation (J_B) and wind (J_w). Assuming a carp population density for 2000-2001 similar to that reported by EIP (2004), one can calculate the flux of bioavailable P due to carp foraging for this period was approximately 2.8 mg/m^2/d bioturbation (eq 4). Since the total resuspension flux rate was estimated at 3.3 mg/m^2/d, this suggests that the wind-driven resuspension flux rate of bioavailable P for 2000-2001 was 0.5 mg/m^2/d. At an average elevation near 1241 ft, about 1.1% of the lake area would have been subject to wind resuspension following eq 7. With a *B* value of 0.015 mg P/g sediment, w_r is about 3030 g sediment/m^2/d, equivalent to wind resuspending the top 2.7 mm of sediment.

Internal Loading (J_{IL})

Internal loading has previously been shown to be an important source of nutrients to Lake Elsinore, especially during periods of limited external loading. Thus, in addition the mathematical expressions to describe the contributions of external loading, wind resuspension and bioturbation, it is necessary to quantitatively describe internal recycling inputs of P. The rate of release of dissolved P was measured at up to 5 sites at the lake on 5 dates over the period Nov. 2000 – Aug. 2001 (Anderson, 2001); dissolved P flux ranged from <0.1 – 13.4±2.4 mg/m^2/d, and varied linearly with depth (e.g., Fig. 5). Temperature and dissolved oxygen (DO) levels were only weakly correlated with dissolved P release (R^2 values of 0.01 and 0.17, respectively) when compared with depth (R^2=0.54 when using all available flux data). The increase in SRP flux with depth reflects the net effect of resuspension and sediment focusing into deeper areas of the lake, as well as potential differences in DO levels and microbial activity.

Annual average SRP flux rates were developed in the nutrient budget study (Anderson, 2001) that considered both spatial and seasonal trends. That study also found that late summer (e.g., September) flux rates approximated the annual average values, with somewhat slower release during the winter and highest release in early summer. Thus flux measurements made in late summer 2002 and 2003 are thought to reasonably approximate annual average values (Fig. 6). Internal loading rates were separately estimated by Montgomery-Watson (1997) for 1995 and 1996 and also included in Fig. 6 that shows that the average P internal recycling rate varies linearly with

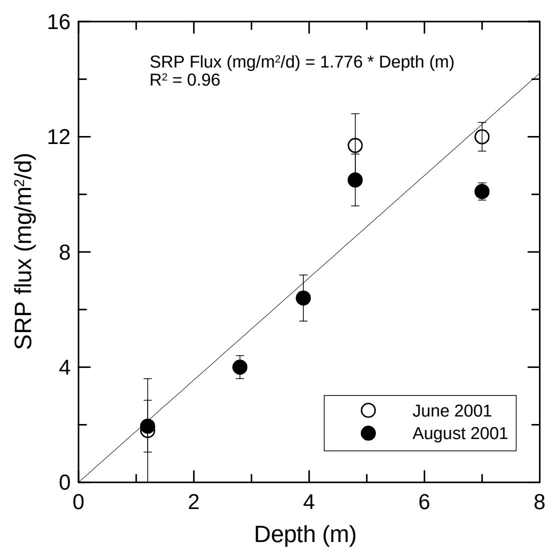

Fig. 5. SRP flux measured during summer of 2001 vs. depth.

the average TP concentration. This observation has practical importance for a number of reasons.

First of all, many models developed to describe the internal recycling of nutrients from sediment involve fairly elaborate descriptions of the sediment properties and their transformation reactions, even though the exact sediment properties and the forms of P involved in release are poorly understood. Parameterization of such models requires a great deal of information about the sediments and a number of assumptions about these transformation reactions.

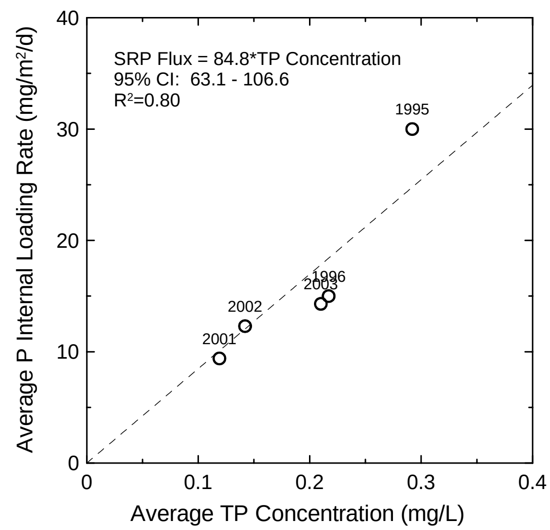

Fig. 6. Average P recycling rate vs. water column TP concentration.

The approach adopted here does not require specific assumptions about the sediment properties and transformation reactions; rather internal recycling rates are proposed to be a linear function of TP in the water column (Fig. 6). This is intuitively not unreasonable since the sedimentation flux rate, J_s, is proportional to the water column concentration, *i.e.*,

$$J_s = \nu C \tag{9}$$

where ν is the settling velocity of algal-derived TP (m/d) and C is the water column concentration (mg/m^3). Moreover, this is consistent with available historical data that shows directly measured (2001, 2002 and 2003) and estimated (1995, 1996) internal loading rates increased with increased average water column TP concentrations (Fig. 6). Since ν is 0.1 m/d (Anderson, 2002), the upward flux of dissolved nutrients (J_{IL}) averaged 84.8% of the downward total P flux (J_S). That is, 84.8% of the TP that settles out of the water column is recycled and only 15.2% is permanently lost from the water column. Thus P is sequestered only slowly in Lake Elsinore. For comparison, it is estimated that the P sequestration rate in the Salton Sea is 3x higher than Lake Elsinore, with 46% of the P that settles out of the water column of the Salton Sea permanently lost to the sediments.

On this basis then, the internal recycling rate of P (J_{IL}), is simply given by:

$$J_{IL} = iO\, C \tag{10}$$

where *i* represents the slope of the fitted line in Fig. 6 (84.8 mg/m^2/d per mg/L), *O* represents a scalar to account for oxygenation or aeration effects, and *C* is the TP concentration in the water column.

Substituting eqs 2, 4, 8, 9 and 10 into eq 1 and dividing both sides by mean depth (H) allows eq 1 to be rewritten as a differential equation describing the change in P concentration over time (dC/dt):

$$\frac{dC}{dt} = \frac{\sum_{i=1}^{n} Q_i C_i + PA_w RC_w}{V} + \frac{fP_r M_f B + w_r A_r B + iOC - \upsilon C}{H} \qquad (11)$$

Calculation of the change in TP concentration over time with eq 11 requires information about a number of variables. The model allows for external loading from both well-defined sources, and from non-point local runoff that will vary in response to precipitation inputs. Such inputs will also influence the volume of the lake (*V*), as well as the surface area and mean depth (*H*). As a result, it is necessary to also write a related hydrologic balance equation where the change in volume (V) over time (dV/dt) is:

$$\frac{dV}{dt} = \sum_i Q_i + PA + PRA_w - EA \qquad (12)$$

where Q_i reflects inputs due to San Jacinto River, recycled water and island well flows and losses due to channel outflow, *P* is the precipitation rate, *A* is the surface area of the lake, *R* is the runoff coefficient, A_w is the area of the local watershed, *E* is the evaporation rate, and *A* is lake surface area. As a result, *PA* reflects direct precipitation onto the lake surface, and PRA_w represents the contributions from local runoff. Loss of water from the lake is assumed to be chiefly due to evaporation from the lake surface (*EA*), although outflow at very high lake elevations (>1255') can also occur.

Table 3. Updated area-volume-elevation data.

Elevation (ft)	Area (acres)	Volume (af)
1218.0	0	0
1218.5	195	64
1220.1	537	664
1221.8	1024	1945
1223.4	1377	3914
1225.1	1561	6324
1228.3	1818	11866
1231.6	2056	18221
1234.9	2279	25333
1238.2	2518	33202
1241.5	3022	42290
1245.0	3319	54504
1250.0	3463	71443
1255.0	3606	89114

The original hypsographic data from Black and Veatch (1995) were updated to reflect bathymetric results from Anderson (2001) (Table 3), which found a lower minimum lake elevation (1218 ft) as compared with Black and Veatch's (1995) value of 1223 ft. Bathymetric data (Fig. 7) were then used to develop an empirical relationship between lake surface area (A, in acres) and volume (V, in acre-feet):

$$A = (V / 9.66x10^{-5})A^{0.4008} \qquad (13)$$

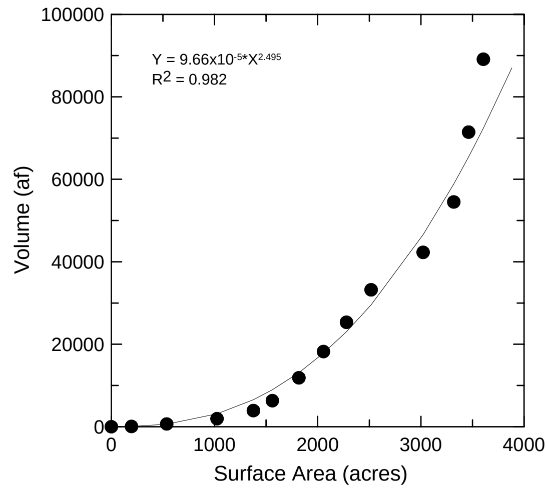

Fig. 7. Volume as function of lake surface area.

Steady-State Solution

Rearranging eq 11 and solving for the steady-state case (i.e., dC/dt = 0) yields the following expression:

$$C = \frac{\dfrac{H(\sum_i Q_i C_i + PRA_w C_w)}{V} + iOC + fP_f M_f B + w_r A_r B}{\nu} \qquad (14)$$

where C is the predicted steady-state concentration of TP in the water column. Inputs of dissolved and bioavailable P into the water column are assumed to be rapidly converted to algal biomass.

Similarly, eq 12 can be solved for the steady-state case where the lake area (A) can be calculated:

$$A = \frac{\sum_i Q_i + PRA_w}{E - P} \qquad (15)$$

The volume can in turn be estimated from the predicted lake surface area using eq 13.

Input Data

Solution to the differential equation describing the TP concentration in Lake Elsinore over time (eq 11) requires solution of the hydrologic balance equation (eq 12); as a result, meteorological and runoff data were used as input for model calculations (Fig. 8).

Precipitation data were averaged from the CIMIS stations at UCR and Temecula (Fig. 8a), while evaporation data were taken from the UCR station (Fig. 8b).

The large precipitation inputs to the region in 1993, 1995 and 1998 resulted in substantial runoff to the lake from the San Jacinto River (USGS gaging station #11070500) (Fig. 8c). Very little measurable runoff to the lake occurred over the period 1999 – 2002, although some significant runoff occurred in the spring of 2003 and in late 2004 (Fig. 8c). Recycled water and island well flows (Fig. 8d) were minor in relation to the very large flows from the upper watershed that occurred occasionally (note difference in y-axis values). Flows were used with measured or assumed concentrations (Fig. 9) to yield TP inputs for the 4 external nutrient inputs to the lake (i.e., the summation term on the rhs of eq 11). As implied by the very large differences in flows to the lake from the watershed during wet years relative to flows in 2002-2004 from other sources (e.g., recycled water), TP inputs also varied strongly between the San Jacinto River inputs during the high precipitation years and the drought years, as well as the other inputs (Fig. 10).

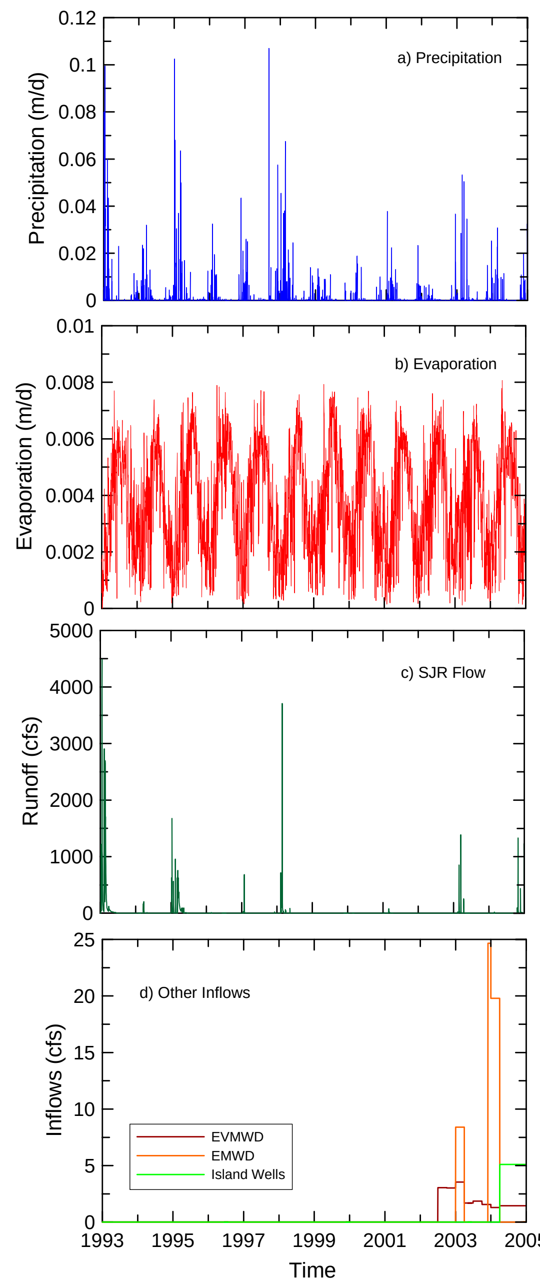

Fig. 8. Meteorological and hydrological conditions: a) precipitation, b) evaporation, c) San Jacinto R. flow and d) other inflows.

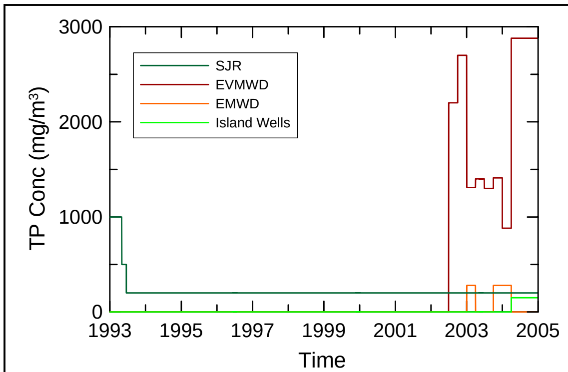

Fig. 9. Total P concentrations of different inflows over time.

Results

Simulation Results: 1993-2004

The differential equations describing the change in TP concentration (eq 11) and lake volume (eq 12) were solved used a daily timestep and the meteorological, flow and other data (Figs. 8 and 9). Predicted lake volumes were converted to predicted lake elevations using the hypsographic data in Table 3.

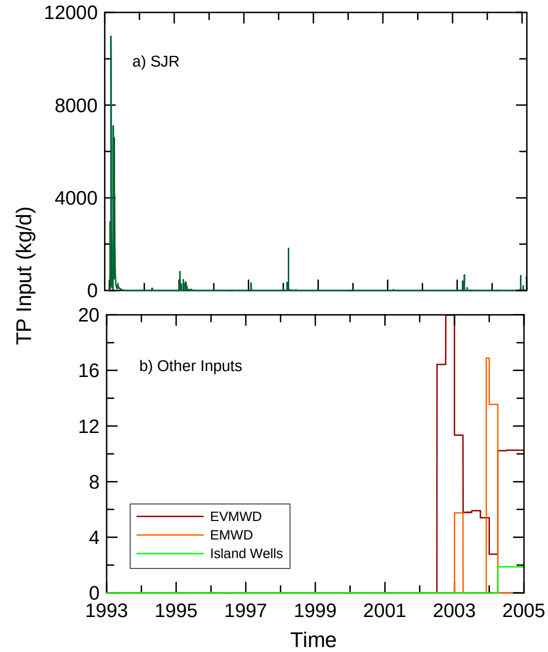

Fig. 10. Daily TP inputs: a) San Jacinto R. and b) other inputs.

The hydrologic model generally did a very good job of reproducing the observed changes in lake elevation (Fig. 11a); the only strong deviation between the predicted and observed values occurred in the summer-fall of 1994, where the model predicted lower surface elevations (<1250') than those recorded (1253'). The reason for this deviation is not clear, but given the good agreement found throughout the remainder of the simulation period, no further efforts were made to reduce this variance.

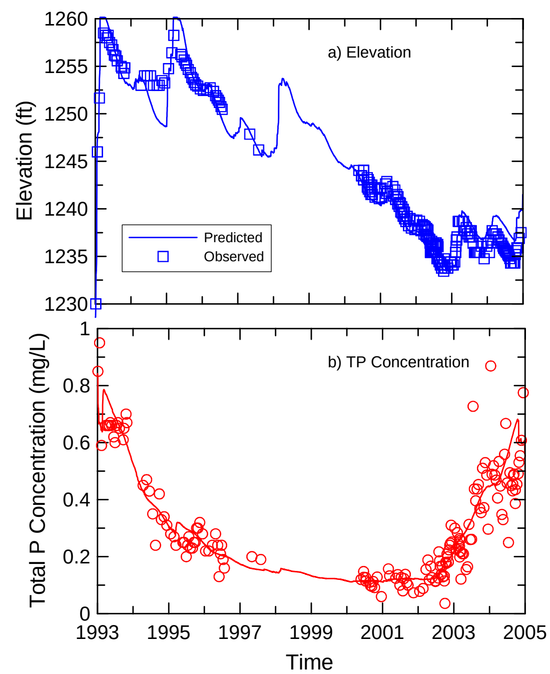

Fig. 11. Predicted and observed: a) lake surface elevation and b) TP concentration.

Total phosphorus concentrations were also quite well described by the model (Fig. 11b). It correctly reproduced the relatively rapid loss in TP over the 1993-1997 period, the relatively stable water quality in 2000-2001, and the increase in TP that began in the spring of 2002 (prior to recycled water addition). As noted in other reports, this increase in TP concentrations coincided with the lowest lake elevations found since January 1993.

Recycled water inputs (Fig. 10b) and continued low lake levels resulted in increases in TP concentrations in the lake through 2004 (Fig. 11b). The development of a strong, year-round dominance of the algal community by the buoyant blue-green alga *Oscillatoria* that began in 2002 may have also contributed to the subsequent increased TP levels by reducing the algal settling velocity (eq 11). In fact, it was necessary to reduce by 4% the settling velocity (v) for 2003 and 2004 to fit the TP levels found at this time (Fig. 11). In the absence of this correction for reduced settling velocity for the buoyant *Oscillatoria*, the predicted TP concentration plateaued near 0.4 mg/L and did not reach the 0.6 mg/L concentrations measured at the lake.

The contributions from each of the 4 types of inputs (external loading, internal loading, wind-driven sediment resuspension and resuspension due to carp foraging, i.e., bioturbation), normalized to the lake surface area, are shown in Fig. 12. External inputs due to storm runoff varied in direct response to runoff volumes, while internal loading varied more slowly over time (Fig. 12a). Nonetheless, even in large runoff years, internal loading dominated the annual predicted nutrient budget, accounting for 82.3 – 97.1% of the total P loading to Lake Elsinore (Table 4). Since internal recycling is directly coupled in this model with water column P concentrations, even very large external loading events, as found in 1993, play a seemingly modest role in the overall nutrient budget, although it is the influence of external loading on P concentration in the lake, combined with the slow relative rate of net sedimentation, that triggers subsequent high internal loading rates. Nutrient loading due to sediment resuspension from wind and bioturbation were predicted to be only minor sources of P to the water column (Table 4). Wind resuspension accounted for <1% of the predicted annual P budget except during those years of very low lake level and low external inputs (Table 4). The amount of P input into the water column from bioturbation was assumed to

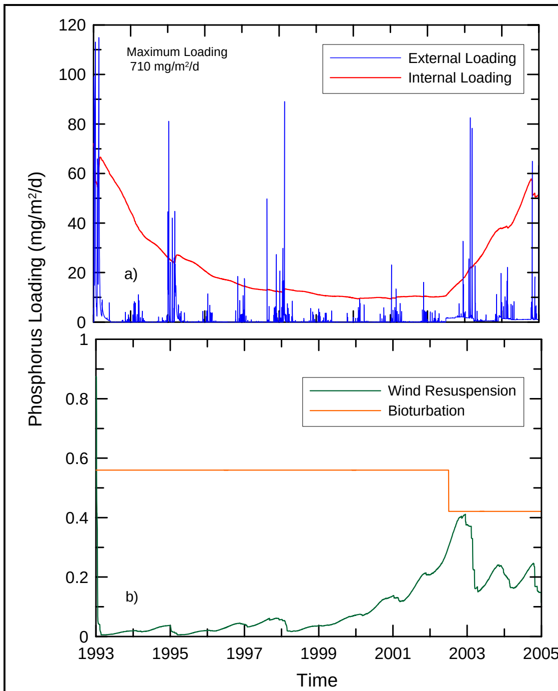

Fig. 12. Phosphorus loading normalized to lake surface area from: a) external loading, b) internal loading, c) wind resuspension and d) bioturbation.

be constant through 2001 since the carp population was treated as at carrying capacity (fish/ha) until carp removal began in 2002.

Table 4. Predicted annual loading (kg) of P to Lake Elsinore for the period 1993-2004.

Year	Total Loading (kg)	External Loading (kg)	Internal Loading (kg)	Wind (kg)	Bioturbation (kg)
1993	588,839	99,487 (16.9%)	484,411 (82.3%)	179 (<0.1%)	4,762 (0.8%)
1994	274,162	2,948 (1.1%)	266,644 (97.3%)	193 (0.1%)	4,377 (1.6%)
1995	244,019	22,257 (9.1%)	216,724 (88.8%)	110 (<0.1%)	4,928 (2.0%)
1996	137,965	2,455 (1.8%)	131,093 (95.0%)	226 (0.2%)	4,191 (3.0%)
1997	96,744	3,816 (3.9%)	88,885 (91.9%)	317 (0.3%)	3,726 (3.9%)
1998	108,252	9,107 (8.4%)	94,679 (87.5%)	215 (0.2%)	4,250 (3.9%)
1999	74,356	1,207 (1.6%)	69,143 (93.0%)	325 (0.4%)	3,681 (5.0%)
2000	56,535	1,191 (2.1%)	51,840 (91.7%)	521 (0.9%)	2,984 (5.3%)
2001	52,696	2,330 (4.4%)	47,070 (89.3%)	727 (1.4%)	2,569 (4.9%)
2002	53,699	4,339 (8.1%)	46,349 (86.3%)	1,142 (2.1%)	1,869 (3.5%)
2003	130,359	8,593 (6.6%)	119,105 (91.4%)	909 (0.7%)	1,751 (1.3%)
2004	212,278	9,897 (4.7%)	199,753 (94.1%)	814 (0.4%)	1,814 (0.9%)

Model Sensitivity Analysis

The solution to the differential and steady-state forms of the lake TP model (eqs 11 and 14, respectively) requires definition of a number of parameters. A sensitivity analysis of the model parameters was thus conducted in which parameters were systematically varied by 5% from their 2001 mean values and compared to the concentration predicted using the steady-state form of the model (eq 14) for the conditions present in 2001 (0.1217331 mg/L). The sensitivity analysis demonstrated that the model parameter with the highest sensitivity was the particle settling velocity (ν), with a relative sensitivity of –95.2% (indicating that predicted concentrations vary quite strongly and inversely with changes in ν). That is, a 5% *increase* in ν resulted in a 4.76% *decrease* in predicted concentration or a relative sensitivity of –95.2%. The factors associated with internal loading (*i* and *O*) bore the next highest relative sensitivities at 74.5%, with lower sensitivity of the model to the terms related to bioturbation (*f*, *P*, *M* and *B*) with relative sensitivities of 23.0-23.6%%. The model was relatively insensitive to the remaining terms (Table 5).

Table 5. Results from model sensitivity analysis. Parameter values represent 2001 conditions.

Parameter	**Description**	**Units**	**Value**	**Predicted TP (mg/L)**	**Sensitivity (%)**
H	Mean Depth	m	4	0.12188357	2.5
Pptn	Precipitation	m/yr	0.25	0.1218457	1.8
R	Runoff Coefficient	m/m	0.37	0.1218457	1.8
A_w	Watershed area	m2	5.40×10^7	0.1218457	1.8
C_w	TP conc of local runoff	mg/m^3	0.2	0.1218457	1.8
V	Volume	m3	4.86×10^7	0.1216258	-1.8
I	Internal loading factor	$mg/m^2/d$ per mg/L	84.8	0.1262699	74.5
O	Aeration factor		1	0.1262699	74.5
F	Fish Resusp Rate	$g/m^2/d$ per kg/ha	0.24	0.1231325	23.0
P	Fish Population	fish/ha	904	0.1231325	23.0
M	Mass of Fish	kg/fish	0.86	0.1231325	23.0
B	Bioavailable P	mg/g	0.015	0.1231703	23.6
W	Wind Resusp Rate	mg/L per %	0.97	0.1217709	0.6
A_r	Area of Resuspension	%	1.3	0.1217709	0.6
ν	Settling Velocity	m/d	0.1	0.11593629	-95.2

Model Uncertainty Analysis

The sensitivity analysis (Table 5) underscored the importance of accurately defining the settling velocity, the factors describing internal loading and, to a somewhat lesser extent, bioturbation. In light of this, a formal uncertainty analysis was conducted which quantified the influence of known uncertainties in these 4 main factors (ν, *I, P* and *B)* on the uncertainty in the predicted TP concentration. A normal distribution about the mean values for each of these 4 parameters was assumed; the standard deviation for each parameter was calculated from available data. That is, for *i*, the internal loading rate term, the standard deviation was calculated from the fitted 95% confidence interval values (Fig. 6); for the carp population, *P*, it was derived from population estimates (EIP, 2004); and for *B*, the labile P concentration in the sediments, it was taken from available sediment data (Anderson, 2001). The standard deviation for the settling velocity, *v*, was set at 10% of the nominal value. The means and standard deviations are presented in Table 6; 1000 simulations using the steady-state form of the model were then conducted in which parameter values were randomly and independently drawn from normal distributions about their mean values using the Monte Carlo technique. The volume and mean depth of the lake were set at those found in 2001, when the lake was near an apparent steady-state with respect to water quality (Fig. 11b). The predicted TP concentration was compared with the observed mean concentration found in 2001.

Table 6. Parameter values used in uncertainty analysis for 2001.		
Parameter	**Units**	**Mean ± s.d.**
ν	m/d	0.10±0.01
i	mg/m^2/d/mg/L	84.8±10.8
P	Fish/ha	904±168
B	mg/g	0.015±0.008

1000 simulations for the 2001 condition using randomly selected parameter sets (Table 6), combined with fixed values for the remaining parameters (Table 5) yielded a mean TP concentration of 0.123±0.021 mg/L. Chlorophyll a concentrations were predicted from TP levels using the regression equation developed from monitoring data for Lake Elsinore (Veiga-Nascimento and Anderson, 2004). The predicted mean TP and chlorophyll concentrations were in reasonable accord with the observed annual average values (e.g., observed mean TP concentration of 0.112±0.029) (Table 7). It should be pointed out that the model was not calibrated to this data set, so the agreement between the predicted and observed values is encouraging. The comparatively large standard deviations for the observed data result from both spatial variability across the 3 monitoring sites at the lake and seasonal differences (this is especially true for chlorophyll, when levels are typically much lower in the winter relative to the summer).

Table 7. Comparison of measured and model-predicted water quality: 2001.		
Water Quality Parameter	**Predicted**	**Observed**
Total P Concentration (mg/L)	0.123 ± 0.021	0.112 ± 0.029
Chlorophyll Concentration (μg/L)	78.6 ± 19.6	80.4 ± 45.8

The results from the 1000 simulations are also presented graphically as probability density functions (pdf) for TP (Fig. 13a) and chlorophyll (Fig. 13b). The predicted concentrations do reasonably conform to a normal distribution, although there is a relatively wide spread to the tails of the distribution (e.g., the 90% confidence interval about the predicted mean TP concentration ranged from 0.089 – 0.166 mg/L). Predicted chlorophyll concentrations also showed a fairly wide spread in their distribution; thus the uncertainty in the key model parameters (Table 6) is propagated through the model to yield predictions with significant uncertainty in average water column properties.

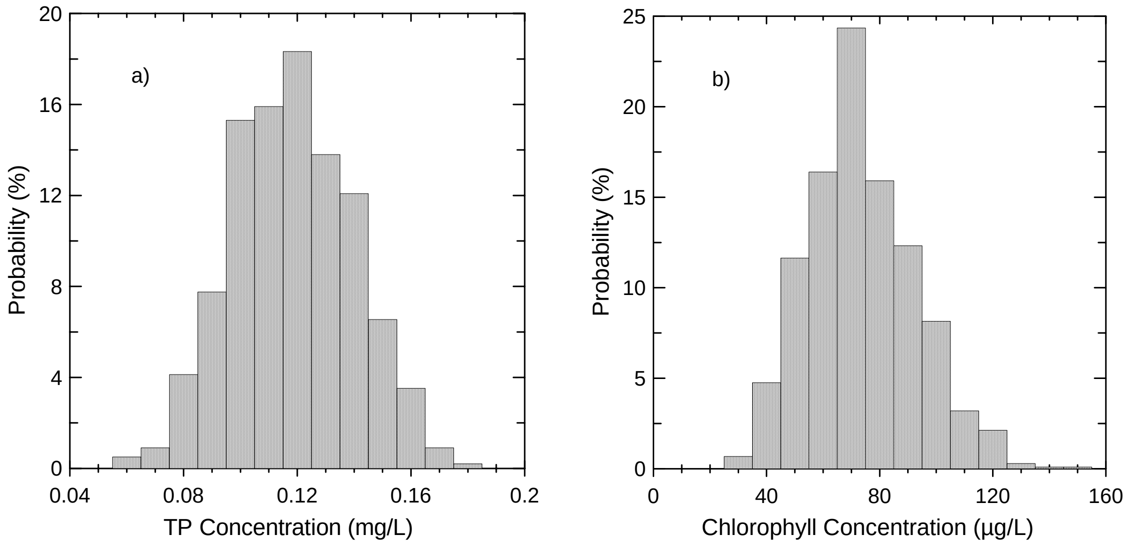

Fig. 13. Model-predicted a) TP and b) chlorophyll concentrations for 2001.

Evaluation of Lake Management Activities: Steady-State Analysis

The deterministic, time-dependent model (eqs 11-12) reasonably reproduced lake level and TP concentrations over the past decade (Fig. 11), while the steady-state form of the model (eq 14) yielded good agreement with measured average TP and chlorophyll concentrations in 2001 (Table 7). Monte Carlo analysis also allowed for the estimation of the uncertainty in the predicted concentrations using values for key parameter with some inherent uncertainty (Table 6).

The steady-state form of the model will now be used to evaluate the influence of recycled water addition, aeration, carp removal, lake level, and island well groundwater inputs on predicted water quality in Lake Elsinore. For this analysis, a Monte Carlo analysis similar to that described above was conducted. That is, 1000 simulations were conducted for each of a range of scenarios using unique parameter sets that involve key model parameter values (υ, I P, and B) drawn from normal distributions about their mean values (Table 6). Other model parameter values were assumed to be fixed (Table 5). Given the low relative sensitivity of the model to most of these parameters, this assumption is not expected to adversely affect the results and conclusions from this analysis.

(i) Effects of Supplemental Flows on Lake Level and Water Quality

An analysis of the proposed lake management activities necessarily requires a reference condition. The reference for this analysis was taken as the theoretical condition that would exist if the lake consistently received the measured mean annual rainfall and mean annual runoff from the San Jacinto River. Analysis of precipitation data from the UCR weather station for the period 1986-2005 as reported by California Irrigation Management Information System (CIMIS) yields a mean annual precipitation rate of 0.25 m/yr. This rate is consistent with values reported for Lake Elsinore and Canyon Lake in other sources. An annual evaporation rate for the region of 1.43 m/yr was similarly derived from CIMIS (2005). Flow data reported by the USGS (http://waterdata.usgs.gov/ca/nwis) for gage #11070500 (San Jacinto River near Lake Elsinore, CA) for the period 1916-2003 yielded a geometric mean annual flow of 558 af/yr. This value was used for the annual natural inflows there, with a mean TP concentration of this inflow of 0.2 mg/L. (Initially, rainfall and runoff to the lake were also modeled with uncertainty, although the result was lake levels that could readily vary from a dry-lakebed to widespread flooding - conditions present over the past 50 years at the lake. Against this backdrop of widely variable hydrologic (and water quality) conditions, it was not possible to infer changes to Lake Elsinore from any given management action.)

Under conditions of mean annual rainfall and runoff, the model predicts a theoretical steady-state lake surface elevation of 1222.7 ft above MSL, a surface area of 1190.3 acres, and a mean depth of 0.96 m (Fig. 14). The lake level was predicted to increase

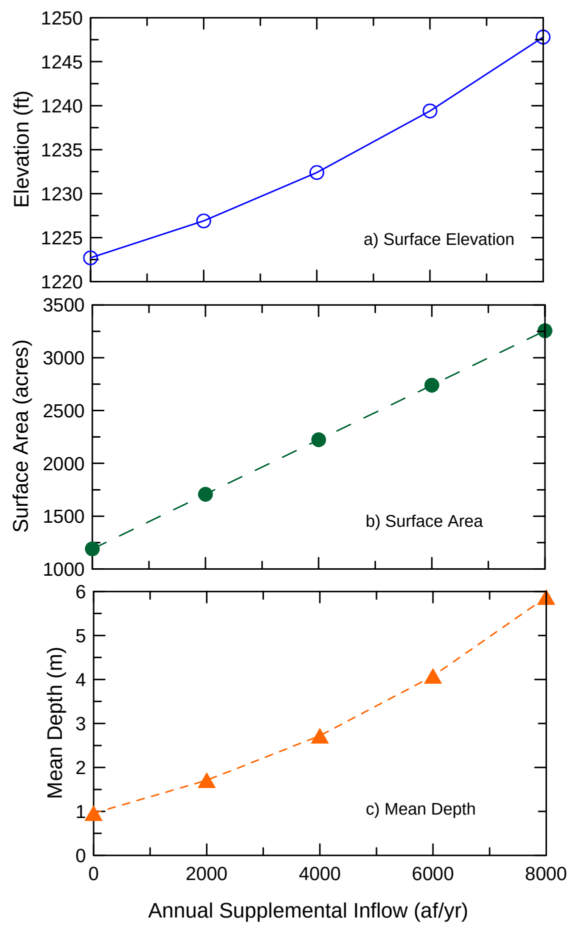

Fig. 14. Influence of supplemental flows on predicted steady-state: a) lake surface elevation, b) surface area, and c) mean depth. A theoretical reference condition was chosen where the lake received mean annual rainfall and runoff, resulting in a very shallow eutrophic condition.

nonlinearly with supplemental inflows, reaching 1240 ft with just over 6000 af/yr and 1247.8 ft with 8000 af/yr of additional water added to the lake (Fig. 14a). The lake surface area increased approximately linearly with increasing supplemental inflow (Fig. 14b); mean depth also increased with inflows, from 0.96 m with no supplementation to 5.87 m with 8000 af/yr of supplemented water (Fig. 14c).

The hydrologic condition of the lake is not influenced by different sources of water, although the predicted water quality in the lake was affected (Fig. 15). At the low lake level reference condition that results from assumption of mean annual rainfall and the (limited) mean runoff, very poor water quality was predicted. The median predicted TP concentration for the reference condition was 0.962 mg/L (Fig. 15a), resulting in very high predicted chlorophyll concentrations (1536 μg/L) (Fig. 15b) and very low Secchi depths (0.04 m) (Fig. 15c). We do not have TP concentration data at this low a lake level, although high TP concentrations (0.5-0.85 mg/L) were present in late 2002-early 2003 just prior to the very strong winter storms that winter when the lake level was <1230 ft (Fig. 11b). Against the poor water quality condition predicted for very low lake levels, addition of supplemental water from any source (EVMWD, EMWD or island wells) was predicted to lower TP and chlorophyll concentrations and increase transparency (Fig.15). The extent of the predicted reduction in TP and chlorophyll levels differed between the 4 water sources, however. A volumetrically equivalent input of recycled water from EVMWD, with a TP concentration of 2.0 mg/L, resulted in higher predicted TP and chlorophyll concentrations than those predicted for an equivalent volume of EMWD

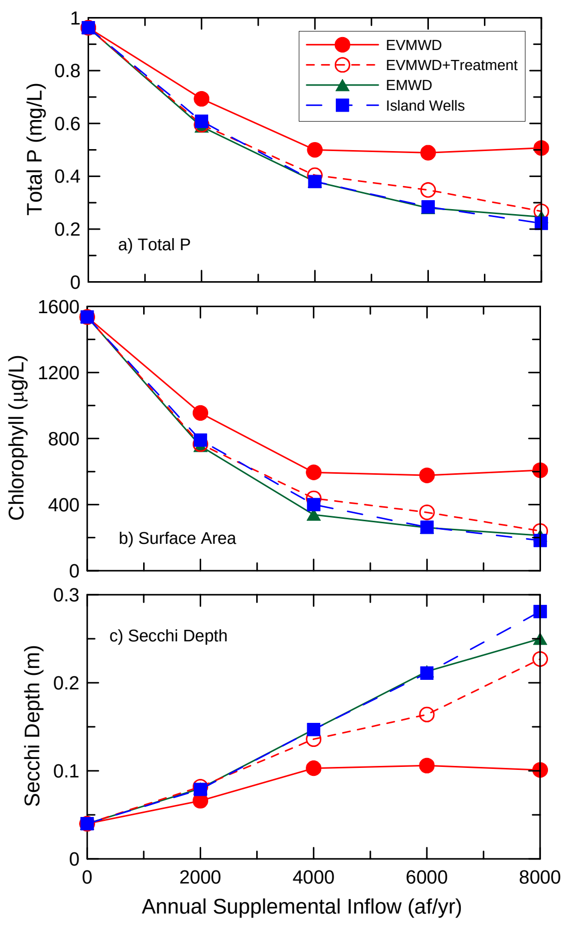

Fig. 15. Influence of supplemental flows on predicted steady-state: a) total P, b) chlorophyll, and c) Secchi depth. A theoretical reference condition was chosen where the lake received mean annual rainfall and runoff, resulting in a very shallow eutrophic condition.

recycled water (at 0.22 mg/L), island well water (at 0.12 mg/L), or EVMWD water assuming chemical P removal to 0.5 mg/L (Fig. 15a). The relative benefit of supplementation decreased within increased inflow, however (i.e., the slope of the plots decreased with increased inflow). This was especially apparent for supplementation with EVMWD water beyond 4000 af/yr, where, although increasing lake level and mean depth (Fig. 14), further inputs did not result in further reductions in TP and chlorophyll. In fact, these water quality parameters were predicted to increase slightly above 6000 af/yr inputs of regular EVMWD recycled water (Fig. 15). Supplementation with groundwater from the island wells would be preferable, since it has the lowest TP concentration, although use of water from EMWD or EVMWD water with chemical P removal would all yield predicted median chlorophyll concentrations below about 200 μg/L at annual inputs of 8000 af.

The predicted loading to the lake under the different source and supplementation scenarios can be apportioned to external sources, internal recycling, wind resuspension and bioturbation from carp (Fig. 16). For the reference case (supplementation of 0 af/yr), an areally averaged median loading rate near 100 mg/m^2/d was predicted, with <1% due to external sources, 85% due to internal recycling, 11.4% due to wind resuspension, and 2.9% due to resuspension from benthic foraging by carp (Fig. 16). Addition of supplemental water lowered the areally-averaged

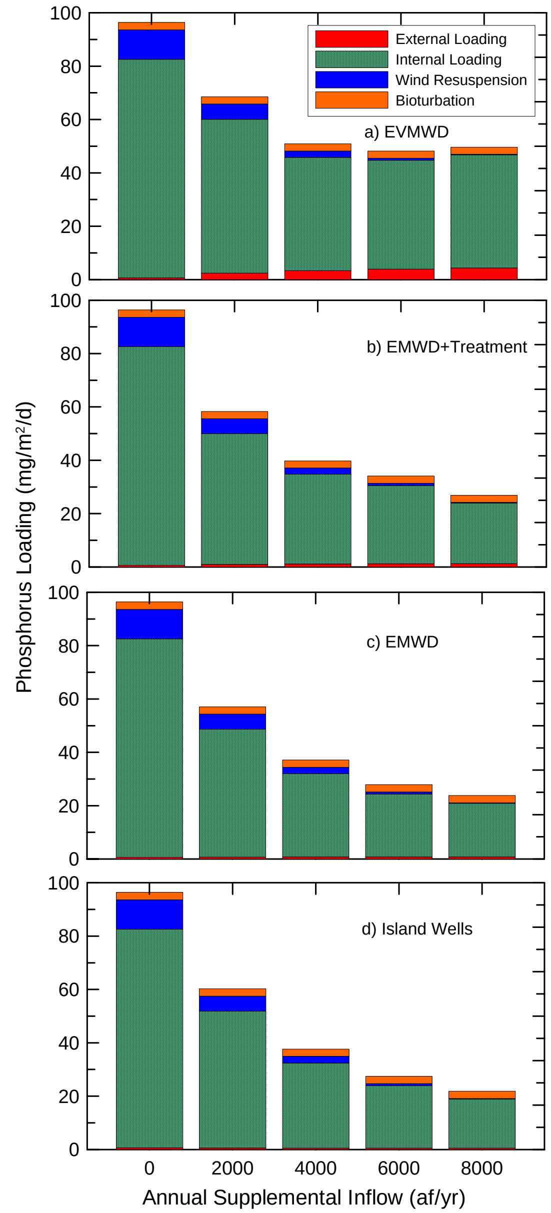

Fig. 16. Influence of supplemental flows on predicted steady-state loading from: a) EVMWD,, b) EVMWD+treatment, c) EMWD, & d) island wells. A theoretical reference condition was chosen where the lake received mean annual rainfall and runoff, resulting in a very shallow eutrophic condition.

loading rates through reductions in wind resuspension and internal recycling. Wind resuspension became relatively unimportant at higher annual flows, accounting for <3% of the total loading (Fig. 16a) when lake levels exceeded about 1239 ft (Fig. 14a). Internal loading rates were reduced through the feedback between water column concentration and J_{iL} (eq 10), since nutrient release into a greater volume of water lowers the relative concentration, further lowering the predicted internal recycling rate. External loading rates remained low even at the highest supplementation at the highest TP concentrations, although it is again necessary to keep in mind the feedback between water column concentration and internal recycling; thus input of higher TP EVMWD recycled water results in higher internal recycling rates than an equivalent volume of EMWD or island well water.

(ii) Effects of Aeration and Carp Removal on Water Quality

The effects of aeration and carp removal on TP levels, chlorophyll concentrations and the steady-state loading budgets were compared against a different baseline condition than that for the analysis of source water impacts on water quality. Here, the reference was a lake level maintained at 1240 ft by annual addition of 4000 af of EVMWD recycled water and 2,171 af of island well water at TP concentrations o f 2.0 and 0.12 mg/L to offset evaporative losses. Under this baseline condition, the model predicted a median TP concentration of 0.377 mg/L, a chlorophyll concentration of 395 μg/L and 0.15 m transparency (Table 8). Under this scenario, external loading (normalized to lake surface area) contributed 2.8 mg/m^2/d (or 7.4% of the predicted total loading of 37.7 mg/m^2/d), while carp bioturbation contributed 2.6 mg/m^2/d (6.9% of total), and wind resuspension contributed 0.7 mg/m^2/d (1.9%) (Table 8). As was the case for the nutrient budget developed for 2000-2001, internal loading dominated the total input of TP to the water column (31.6 mg/m^2/d or 83.8%) (Table 8).

Aeration very favorably reduced the predicted steady-state TP and chlorophyll levels in the lake; for example, assuming aeration reduces the TP internal recycling rate by 30%, as found in sediment core-flux measurements (Anderson, 2004), steady-state water column TP concentrations were predicted to decline from 0.377 mg/L for the baseline case to 0.148 mg/L (Table 8). Similarly, chlorophyll levels declined from 395 to 103 μg/L and Secchi depths increased from 0.15 m to 0.28 m (Table 8). Although external loading remained the same (2.8 mg/m^2/d) in all scenarios (since flows and concentrations were fixed in these simulations), the steady-state internal recycling rate was lowered from 31.6 mg/m^2/d in the baseline case to 8.7 mg/m^2/d assuming 30% reduction due to aeration (Table 8). As a result of the lowered internal loading rate, the total loading was reduced to a predicted median value of 14.7 mg/m^2/d, with external loading, wind resuspension and bioturbation contributing larger relative amounts (4.8 – 19%).

Table 8. Predicted median water quality and phosphorus loading under different lake management techniques.

Scenario	**Water Quality Variables**			**Phosphorus Loading (mg/m^2/d)**				
	TP (mg/L)	**Chl (ug/L)**	**Zsd (m)**	**External**	**Internal**	**Wind**	**Carp**	**Total**
Baseline[a]	0.377	395	0.15	2.8 (7.4%)	31.6 (83.8%)	0.7 (1.9%)	2.6 (6.9%)	37.7
Aeration								
15%	0.224	186	0.28	2.8 (12.6%)	15.9 (71.6%)	0.7 (3.2%)	2.8 (12.6%)	22.2
30%	0.148	102	0.42	2.8 (19.0%)	8.7 (59.2%)	0.7 (4.8%)	2.5 (17.0%)	14.7
50%	0.108	65	0.56	2.8 (25.9%)	4.5 (41.7%)	0.7 (6.5%)	2.8 (25.9%)	10.8
Carp Removal								
50%	0.331	327	0.18	2.8 (8.5%)	27.9 (85.1%)	0.7 (2.1%)	1.4 (4.3%)	32.8
75%	0.260	231	0.23	2.8 (10.9%)	21.6 (83.7%)	0.7 (2.7%)	0.7 (2.7%)	25.8
Aeration+Carp								
30%+50%	0.121	76	0.51	2.8 (23.5%)	7.0 (58.8%)	0.7 (5.9%)	1.4 (11.8%)	11.9
50%+75%	0.070	25	0.74	2.8 (39.4%)	2.9 (40.8%)	0.7 (9.9%)	0.7 (9.9%)	7.1

[a]*Baseline condition taken as: EVMWD recycled water and island well pumping to maintain 1240 ft surface elevation (4000 and 2171 af, respectively) assuming mean annual values for rainfall (0.25 m/yr) and San Jacinto River flow (558 af/yr). Total P concentrations were as used elsewhere (2.0, 0.12 and 0.22 mg/L for recycled water, island wells groundwater and local runoff, respectively).*

Reducing the internal recycling rate by 30% is equivalent to the assumption that a larger fraction of particulate P reaching the sediments is buried and permanently lost from the system. This greater burial efficiency results in less P being recycled to the water column; with less TP in the water column, the sedimentation flux is reduced (eq 9), and so even less is subsequently recycled (and so on). Thus, this feedback loop exists wherein even quite modest reductions in the internal recycling rate (e.g., 15%) result in 50% net reduction in the steady-state internal loading of P (Table 8). Not unexpectedly then, a 50% reduction in the internal recycling rate had the greatest benefit to predicted water quality, yielding a predicted steady-state TP concentration of 0.108 mg/L, 65 μg/L chlorophyll and Secchi depths exceeding 0.5 m (Table 8).

The removal of carp had less dramatic effects on predicted water quality; for example, a 50% reduction in the carp biomass in the lake only lowered the predicted median TP concentration from 0.377 mg/L to 0.331 mg/L (Table 8). A 75% reduction in carp biomass had a greater beneficial effect, lowering predicted TP levels to 0.260 mg/L. The lower relative benefit from carp removal compared with aeration is consistent with the results of the model sensitivity analysis that found the model to be 3x more sensitive to the internal loading parameters relative to those describing bioturbation (Table 5).

To achieve the greatest improvements in water quality, it is necessary to apply multiple management actions. Maintaining the lake near 1240 ft surface elevation (or above) will help minimize wind-driven sediment resuspension, while aeration and carp removal help to lower these sources of nutrients to the water column. A 50% reduction in internal loading rate coupled with a 75% reduction in carp biomass achieved a predicted steady-state TP concentration 0.070 mg/L, with a corresponding chlorophyll level of 25 μg/L and transparencies exceeding 0.7 m (Table 8). Based on this analysis, it appears that efforts to improve water quality should principally be directed at reducing external loading to the lake combined with measures to lower internal recycling.

Non-Steady State Analysis of Lake Restoration Techniques

The above analyses evaluated the steady-state conditions that would result from implementation of various management actions at the lake. The steady-state analysis offered the clearest way to assess the relative water quality impacts resulting from different supplemental inflows to the lake, aeration and carp removal. Of course, Lake Elsinore is often far from any steady-state condition. For this analysis, then, the time-varying model for the lake (eqs 11, 12) was used to simulate a 1993-2004 period subject to the following conditions: (i) recorded rainfall, runoff and evaporation will be used as model inputs as done previously (Fig. 8a-c); (ii) the actual supplemental flows added to the lake that commenced in 2002 will be excluded (Fig. 8d); (iii) supplemental water will be added to the lake whenever the lake level drops below 1247 ft; (iv) when supplemental water is needed, recycled water from EVMWD will be supplied at daily rates of 10.96 af/day (up to 4000 af/yr), while island well water will be pumped and delivered to the lake when the level drops below 1240 ft (at a rate of 7.5 af/day). Thus, up to 18.46 af/day could be added when the lake levels drops below 1240 ft. Due to the uncertainty in the availability of EMWD recycled water, it was excluded in this analysis. The TP concentrations in these sources were set at those used previously (2.0 and 0.12

mg/L, respectively). Additional simulations were also conducted assuming chemical P removal at the EVMWD plant yielded effluent concentrations of 0.5 mg/L, and assuming aeration (15, 30 and 50% reductions in internal recycling rates) and carp removal (50, 75 and 90% removal) were implemented.

The predicted elevation of Lake Elsinore using only natural precipitation, local runoff and measured San Jacinto River flows varied over the 1993-2004 period in a similar manner to that observed (and predicted) (Fig. 11), even without the supplemental flows initiated in 2002 (Fig. 17a, black line).That is, the modest inputs from recycled water and the island wells did not substantially alter the predicted lake level. If however, recycled water was supplied whenever the lake dropped below 1247 ft (e.g., in the summer of 1997 and beginning again in the summer of 1999), the lake level was predicted to be noticeably elevated (Fig. 17a, black line) relative to the no-supplementation case (Fig. 17a, red line). Further flow augmentation with the island wells when the lake level declined below 1240 ft (Fig. 17a, green line) resulted in only a very slight deviation from that predicted with only EVMWD recycled water was added. This occurred because of the limited time that the lake was below 1240 ft with EVMWD supplementation (Fig. 17a; Table 9).

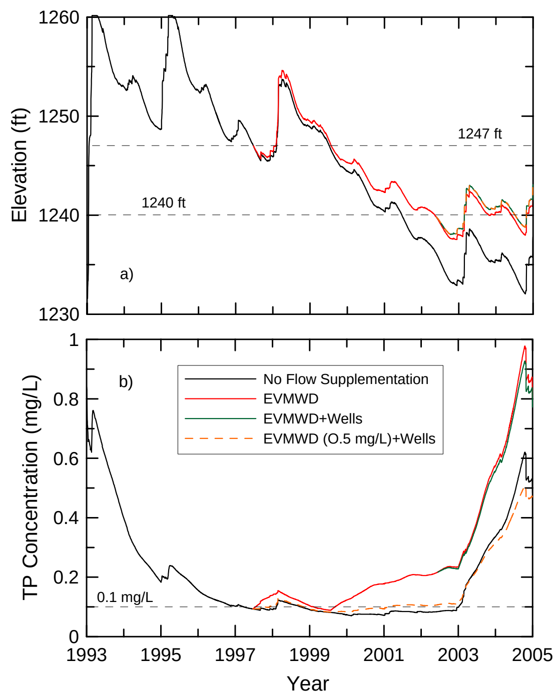

Fig. 17. Influence of flow supplementation on: a) lake elevation and b) TP concentration.

Table 9. Influence of supplemental flows on the frequency of days from 1993-2004 that the lake level declined below 1240 ft and when TP levels exceeded 0.1 mg/L.

Supplementation Scenario	**# Days<1240 ft**	**# Days TP >0.1 mg/L**
No Flow Supplementation	1303 (29.7%)[a]	2516 (57.4%)
EVMWD	465 (10.6%)	4060 (92.7%)
EVMWD+Wells	415 (9.5%)	4060 (92.7%)
EVMWD (0.5 mg/L)+Wells	415 (9.5%)	3324 (75.9%)

[a]*Total number of days in simulation period = 4381*

That is, addition of EVMWD recycled water lowered the number of days during this simulation period that the lake dropped below 1240 ft from 1303 days (29.7% of the time) to 465 days (10.6% of the time). Further supplementation with groundwater from the island wells lowered the number of days to 415 (9.5% of the time) (Table 9).

As previously noted, the TP concentration in the lake varied strongly over this period. With natural inputs from precipitation, local runoff and San Jacinto River flows, the TP concentration was predicted to decline strongly from the very high levels present in early 1993 to <0.1 mg/L in 1999, and remain low for the next 4 yrs (Fig. 17b) due in part to the very limited runoff during this period (Fig. 8c). Concentrations were then predicted (and observed) to increase due increased external loading in the winter of 2003 that resulted in subsequent greater internal loading, with TP levels predicted to reach 0.6 mg/L at the minimum lake level that year before declining due in part to reduced wind resuspension and dilution with the early winter rains at the end of 2004. Under this baseline condition of no flow supplementation, 2516 days out of 4381 (57.4%) the predicted TP concentration was greater than 0.1 mg/L.

As indicated above, supplementation with EVMWD recycled water lowered the number of days when the lake was below 1240 ft (from 29.7 to 10.6% of the time), (Fig. 17a; Table 9), but increased rather markedly the predicted TP concentrations (Fig. 17b; Table 9). For example, the TP concentration near the end of the simulation period was predicted to increase from about 0.6 mg/L to almost 1 mg/L (Fig. 17b). Moreover, the period of comparatively good predicted (and observed) water quality from 1999-2002 under the baseline condition (no EVMWD inputs) with TP levels <0.1 mg/L would be replaced with TP concentrations exceeding 2x that level (>0.2 mg/L) (Fig. 17b, red line). The number of days in which the TP concentration exceeded 0.1 mg/L increased from 2516 days (57.4% of the time) to 4060 days (92.7% of the time) (Table 9). Further supplementation with island well water when the lake level dropped below 1240 ft had little effect on predicted TP concentrations; a small dilution lowered the predicted TP in the lake slightly, but was not predicted to change the number of days when the lake exceeded 0.1 mg/L, since island well supplementation occurred when the lake TP levels were already substantially elevated due to recycled water inputs (Fig. 17b; Table 9). Supplementation with reduced TP (0.5 mg/L) EVMWD water + island well water lowered quite substantially the predicted TP concentration in the lake (Fig. 17b, dashed orange line) relative to the level predicted for EVMWD + island well water (Fig. 17b, red line). The predicted TP concentration actually followed fairly closely the TP concentration predicted under the baseline case (Fig. 17b) while favorably increasing the lake level (Fig. 17a). Addition of 0.5 mg/L EVMWD water + island well water lowered the fraction of time when the TP concentration was below 0.1 mg/L from 92.7% to 75.9%, although it did remain above the 0.1 mg/L level more frequently than the base case (57.4%).

The effects of aeration and carp removal on predicted TP levels for this period were also evaluated. For these analyses, the reference was taken as EVMWD + island wells condition as described above (i.e., where EVMWD recycled water was added at 10.96 af/day whenever the lake level dropped below 1247 ft and island well water was further added at 7.5 af/day when the lake level declined below 1240 ft). The effect of aeration was dramatic; even a modest (15%) reduction in the recycling rate greatly lowered the predicted TP levels in the lake (Fig. 18a), and reduced the frequency of TP levels in the lake >0.1 mg/L from 92.7% to 49.6% of the time (Table 10). Further improvements were predicted when aeration reduced the internal recycling rate by 30% or 50% (i.e., increased the rate of net P burial in the sediments); here, the exceedance frequencies declined further, to 24.7 and 6.4%, respectively (Table 10).

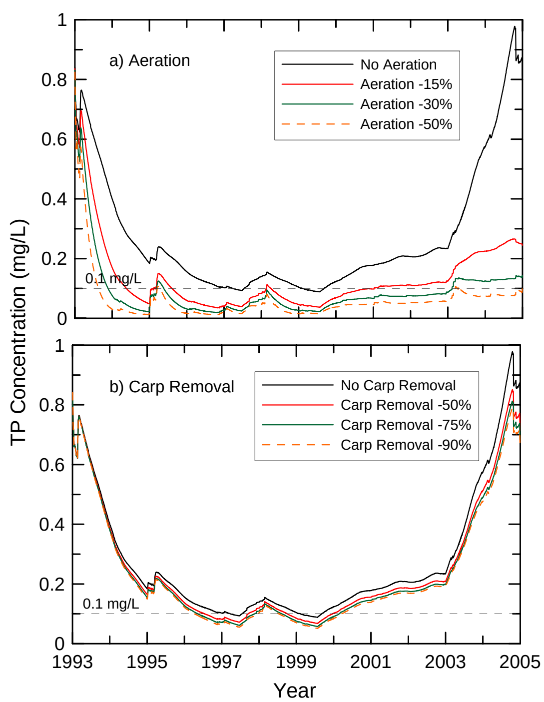

Fig. 18. Predicted TP concentrations over time with recycled water and well inputs. Effects of: a) aeration and b) carp removal.

Table 10. Effect of aeration and carp removal on frequency when TP>0.1 mg/L.

Action	**Reduction**	**#Days >0.1 mg/L**
Aeration	None	4060 (92.7%)
	15%	2172 (49.6%)
	30%	1083 (24.7%)
	50%	280 (6.4%)
Carp Removal	None	4060 (92.7%)
	50%	3509 (80.1%)
	75%	3267 (74.6%)
	90%	3163 (72.2%)
Combined Treatment	None	4060 (92.7%)
0.5 mg/L	None	3324 (75.9%)
0.5 mg/L+ Aeration	30%	394 (9.0%)
0.5 mg/L+ Aeration+Carp	30% +75%	360 (8.2%)

Carp removal was much less effective at reducing predicted TP levels than aeration, however (Fig. 18b). Carp removal lowered predicted TP levels only modestly, but did reduce the percentage of time predicted TP concentrations exceeded 0.1 mg/L (Table 10). Chemical P removal at the EVMWD plant to 0.5 mg/L favorably lowered predicted TP levels in the lake relative to the effluent without P removal (Fig. 17b).

Aeration that achieves 30% reduction in the internal recycling rate further improved predicted water quality, lowering the frequency that the lake exceeded 0.1 mg/L from 75.9 to 9.0% of the time (Table 10). With 75% carp removal, the exceedance frequency declined to 8.2% of the time (Table 10).

Key Findings and Conclusions

A model that incorporated external loading, internal recycling, wind resuspension and carp bioturbation was able to reproduce observed lake water quality over the period from 1993-2004. The differential and steady-state forms of the model were then used to assess the water quality impacts from recycled water and groundwater inputs, aeration and carp removal. A number of conclusions can be drawn from these analyses. First of all, the process of internal recycling dominates the annual loading of phosphorus to the water column of Lake Elsinore, even during years with very strong inputs from the San Jacinto River. Model results indicate that external loading contributed on average only 5.7% of the total annual P loading for the period from 1993-2004, while wind resuspension and bioturbation from carp typically accounted for 0.6% and 3%, respectively (Table 4). Even during the extreme runoff conditions of 1993, external loading was predicted to account for only 16.9% of the total annual P loading to Lake Elsinore. These statistics belie the impact of external loading on P concentrations and lake water quality, however, since externally loaded P is recycled a number of times and potentially over several years before it is finally buried. The lake does have some capacity to sequester P in the sediments, with about 15% of the P settling to the sediments thought to remain there. This capacity to remove P from the water column can be seen from the monitoring data collected from 1993 – 1997, where the TP concentration in the lake declined from almost 1 mg/L to about 0.2 mg/L (Fig. 11b); total annual internal loading was predicted to decline more than 80%, from over 484 metric tons to 89 metric tons of P for this period (Table 4).

A steady-state form of the model was used to assess the impacts of recycled water and groundwater inputs to the lake using a theoretical reference condition based upon the lake receiving mean annual rainfall and runoff. Under such a condition, the lake level was predicted to drop to 1222.7 ft above MSL, have a mean depth <1 m (Fig. 14), and TP concentrations near 1 mg/L (Fig. 15a). Against this shallow, hypereutrophic lake condition, addition of supplemental water increased surface elevation, mean depth and decreased predicted TP concentrations. About 6000 af/yr was predicted to increase the lake level to 1240 ft (Fig. 14a). The source of the supplemental water affected the predicted TP levels; inputs from EVMWD, EMWD or the island wells yielded median predicted TP concentrations of 0.49, 0.28 and 0.28 mg/L, respectively (Fig. 15a). Chemical phosphorus removal at the EVMWD plant to 0.5 mg/L lowered the median predicted TP concentration in Lake Elsinore to 0.35 mg/L. As noted above, even under conditions of high annual inflows with high TP concentrations, internal loading dominated the predicted P budget for the lake due in large measure to the recycling of externally loaded phosphorus.

Other restoration efforts also affected the predicted steady-state water quality in the lake. Aeration was found to very favorably reduce median predicted TP and chlorophyll concentrations; for example, aeration sufficient to reduce the internal recycling rate by 30% was predicted to lower TP from 0.377 mg/L to 0.148 mg/L and reduce chlorophyll levels from 395 μg/L to 102 μg/L (Table 8). Carp removal was less effective than aeration at improving water quality (e.g., a 75% reduction in carp biomass lowered the median predicted TP concentration to 0.26 mg/L). It appears that reducing external phosphorus loadings to the maximum extent practical, coupled with concerted efforts to reduce internal phosphorus loading rates through in-lake treatments is the best approach to consistently achieve water quality objectives.

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