APPENDIX B

LAKE ELSINORE WATER QUALITY MODEL

M. ANDERSON (2003)

Note: This model was used to calculate the total phosphorus load capacity for Lake Elsinore. The model is currently being updated and refined. The update includes sensitivity analysis to address the uncertainties in the parameters in the model. A nitrogen model is also under development.

Water Quality in Lake Elsinore: Model Development and Results

Michael Anderson

Summary

Water quality data over the past decade were analyzed and used to develop a simple dynamic model for Lake Elsinore. The model was able to reproduce water quality in the lake from 1993-present. The model was then used to predict water quality under a variety of scenarios, including continued declines in lake elevation, stabilized lake levels with recycled water of varying nutrient contents, and selected in-lake management techniques.

Analysis of Lake Elsinore Data: 1993 - 2002

Data available for Lake Elsinore for the period 1993-1997 and 2000-2002 were provided by the SARWQCB. This data included water quality measurements made by Black and Veatch, Montgomery-Watson, EVMWD, SARWQCB and others. Recent data collected by UCR have also been added.

Lake elevation has changed quite dramatically over the past decade (Fig. 1). A single measurement made November 23, 1992 was not included (lake surface elevation of 1229 ft), but reflects the low water conditions also present on January 5, 1993 (Fig. 1). Lake elevation increased substantially over the next 3 months, and reached a maximum elevation of 1258.5 ft. Surface elevation then declined to approximately 1253 ft before increasing again in the winter storms of 1995. Limited rainfall and runoff during winter of 1996 and 1997 resulted in continued reductions in lake elevation over this period, with the lake elevation declining below 1250 ft. The available surface elevation (and related water quality) record stops and then picks up again in 2000-present (Fig. 1).

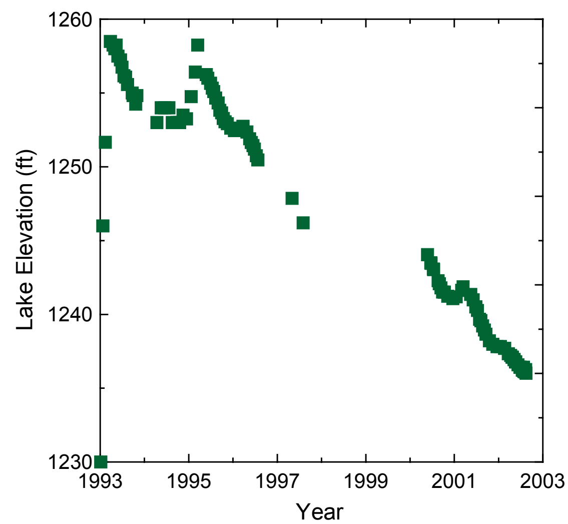

Fig. 1.

The storage-elevation data from Black and Veatch (1995) was used to develop an empirical equation allowing elevation data to be converted to lake volumes (Fig. 2a). The data could be quite reasonably described using a 2nd-order polynomial (R^2=0.999) of the form:

$$Vol\ (af) = 51,200,209 - 85,457.1*Elev\ (ft) + 35.63*Elev\ (ft)^2 \qquad (1)$$

Equation (1) was then used to predict lake volumes over time (Fig. 2b).

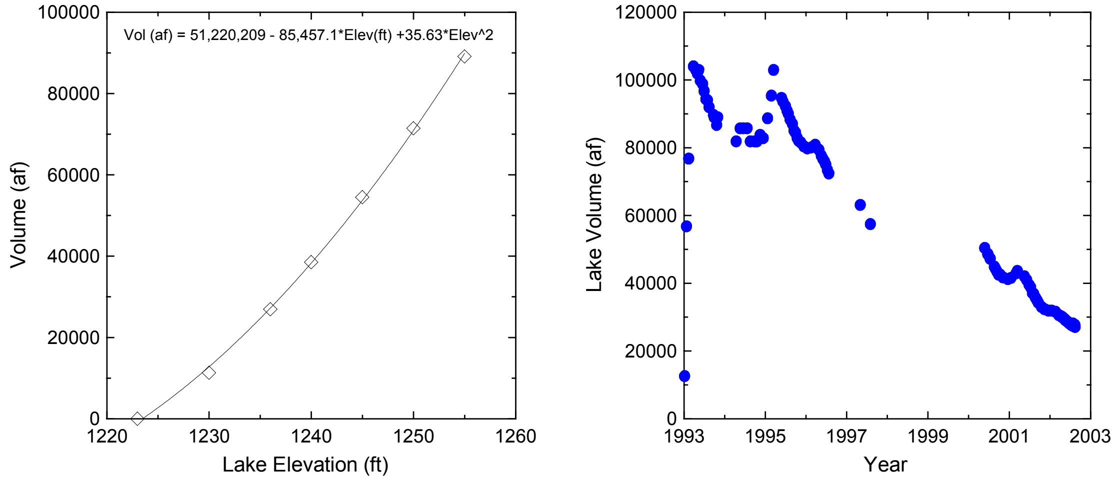

Fig. 2.

Water quality changed substantially over this time period as well (*e.g.*, Fig. 3). Early 1993 was characterized by very high total P concentrations (>0.6 mg/L) but relatively low total N levels (2-3 mg/L). Thus, as previously recognized, the lake was rather strongly N-limited during this time. Total P concentrations decreased significantly over the next 4 years, while total N increased slightly (Fig. 3). The intersection of the data coincides with at TN:TP ratio of 10:1, so the lake was relatively balanced with respect to nutrients in 1995-1996. More recently, TN concentrations have exceeded 4 mg/L while TP levels have typically averaged about 0.15 mg/L, so the lake is now more typically P-limited, especially during the summer (Fig. 3).

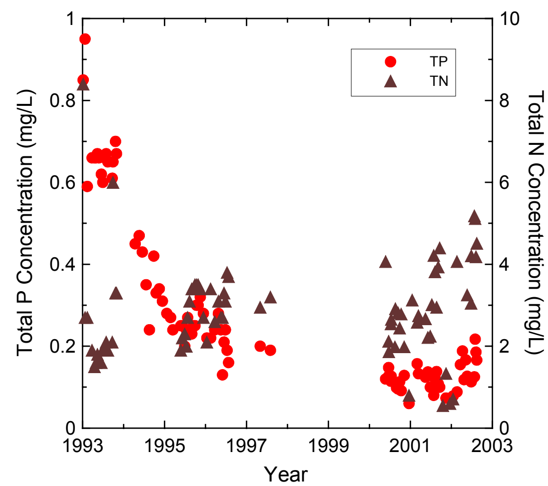

Fig. 3.

Chlorophyll levels and TN:TP ratios varied significantly over this time period as well (Fig. 4). High chlorophyll levels were found during the later summer following the large inputs of P of the winter, 1993. As noted, however, the lake was strongly N-limited (Fig. 4). Summer chlorophyll levels decreased over the next several years. More recently, summer chlorophyll levels have increased from about 100 to 150 and >300 µg/L over the past 3 summers (2000-2002) (Fig. 4). Winter chlorophyll levels (Fig.4) have been generally quite low and typically coincide with the highest TP levels in the lake (Fig. 3).

Plotting TP and TN concentrations as a function of lake elevation provides some interesting results (Fig. 4). Specifically, high TP concentrations (>0.5 mg/L) have, in fact, been noted both at very low lake elevations (~1230 ft) and very high surface elevations (Fig. 4).

The cluster of data near 1240 ft elevation represents recent conditions (2000-present), where lake levels have declined and TP (and TN) levels have begun increasing. The red and brown lines are 2nd-order polynomial and linear fits to the TP and TN data, respectively. These lines are principally used to help highlight general trends in N and P concentrations with lake elevation and do not have any particular significance, especially at high lake levels. Nevertheless, the figure suggests that the recent increases in TP and TN concentrations, coinciding with lake elevation decreases from about 1242 to 1236 are potentially on pace to return to the very high levels found in late 1992 and early 1993 during the lowest recent recorded lake levels.

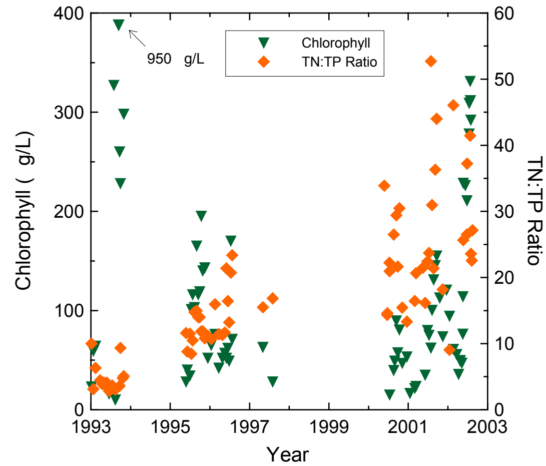

Fig. 4.

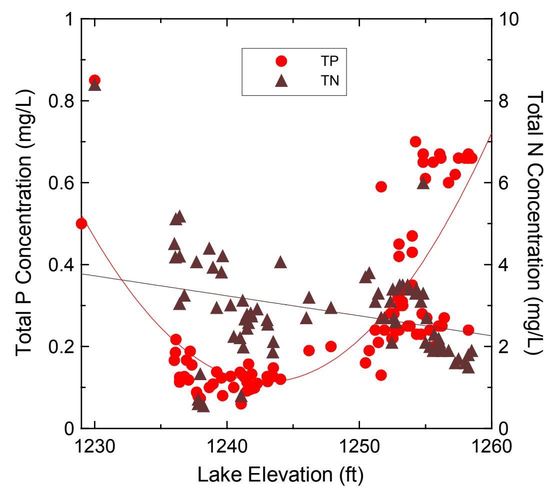

Fig. 5.

Modeling Nutrient Concentrations

The rapid decline in TP concentrations from 1993-1996 (Fig. 3) suggests a first-order loss process. Since the lake volume changed over this time period due to evaporative losses and inflows (relatively little water was lost as outflows; Montgomery-Watson, 1997), with corresponding external loading, a simple coupled water and P-balance model was developed.

The change in nutrient mass within the lake was calculated as:

$$\frac{dM}{dt} = \frac{dVC}{dt} = V\frac{dC}{dt} + C\frac{dV}{dt} \quad (2)$$

where M is the total mass in the system (kg), V is the lake volume (m^3), C is the concentration (mg/m^3) and t is time. The 1st term on the right-hand side of the equation can be written as:

$$V\frac{dC}{dt} = Q_{in}C_{in} - Q_{out}C_{out} - \frac{v}{H}CV \quad (3)$$

where Q_{in} is the flow entering the lake (m^3/yr), Q_{out} is the flow exiting the lake (m$_3$/yr), C_{in} and C_{out} are the influent and effluent concentrations, respectively, v is the net settling velocity (m/yr) and H is the mean lake depth (m).

A number of factors influence the net settling velocity in a lake, including the rate of particle settling and the rates of internal loading and resuspension. It can be shown that the net sedimentation rate, v, is related to the settling rate, v_s (m/yr), the internal loading rate parameter, k (m/yr), and a resuspension velocity, r (m/yr) by:

$$v = v_s - \frac{C_{sed}(k + r)}{C} \tag{4}$$

where C_{sed} is the volumetric sediment P concentration (mg/m^3) and C, as defined above, is the water column TP concentration (mg/m^3). Thus, the net sedimentation rate will be dependent upon the rate in internal loading (taken here to include both dissolved and particulate P, with dissolved being converted to algal forms within the water column), the rate of resuspension, the P concentration in the sediments (assumed to be constant), and the concentration of TP in the water column (eq 4).

The internal loading rate parameter, k, is effectively then a velocity term describing the rate at which TP is released from the sediments. This parameter is expected to vary depending upon the conditions at the lake. For example, higher TP concentrations in the water column would result in greater delivery of particulate P to the bottom sediments, which could, in turn, result in higher recycling rates. To assess this, average P internal loading rates, either measured (Anderson, 2001) or estimated (Mongtomery-Watson, 1997), were plotted as a function of annual average TP concentratons (Fig. 6). Since the late summer P flux rates were previously found to be comparable to the annual average flux rates (Anderson, 2001), recent core-flux results from a site within the high-speed zone (12.3 mg/m^2/d) were used as an estimate for 2002. A linear relationship was found between average TP concentration and average P internal loading rate (Fig. 6).

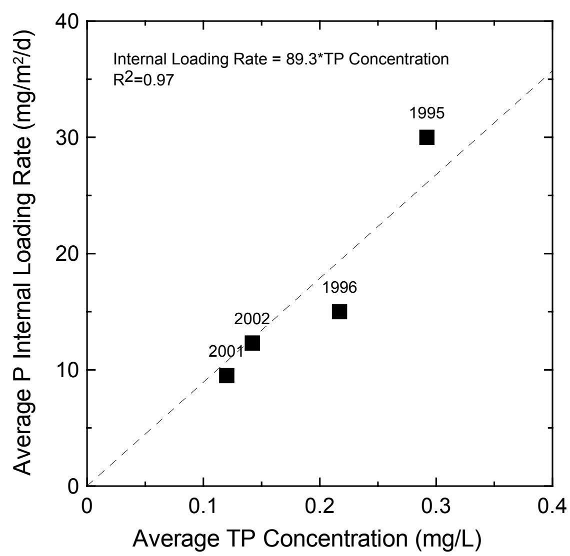

Fig. 6.

Sediment resuspension rates are also expected to vary depending upon lake conditions, in this case, specifically as a function of lake elevation. The potential for resuspension at Lake Elsinore is high given its shallow depth, relatively long fetch and periodic strong winds. 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:

$$L = \frac{gT^2}{2\pi} \tag{5}$$

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:

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

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 a wind speeds of 2.5, 5 and 10 m/s assuming a 3 km fetch. These lower wind speeds represent the range in typical daily average wind speeds often found at the lake, while this 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. 7.

Table 1. Predicted wave properties as a function of windspeed for a 3 km fetch.

Windspeed (m/s)	Wave Period (s)	Wavelength (m)	Critical 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

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 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

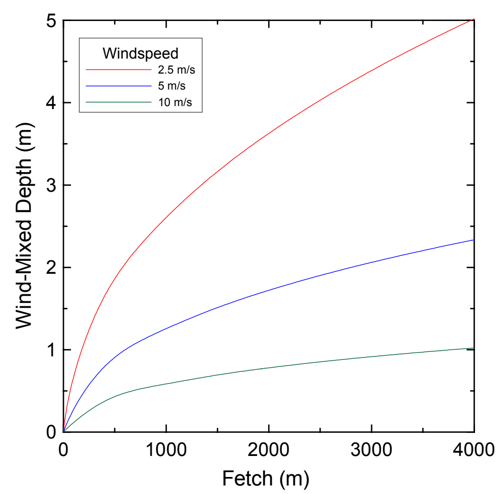

Fig. 7.

lake bottom comprised of type II or III sediments that may be actively resuspended (*i.e.*, occur within the wind-mixed region) (Fig. 8). The percent area of bottom sediments potentially resuspended will vary strongly depending upon lake elevation, with relatively little resuspension at high elevations and extensive resuspension potential at low surface elevations (Fig. 8).

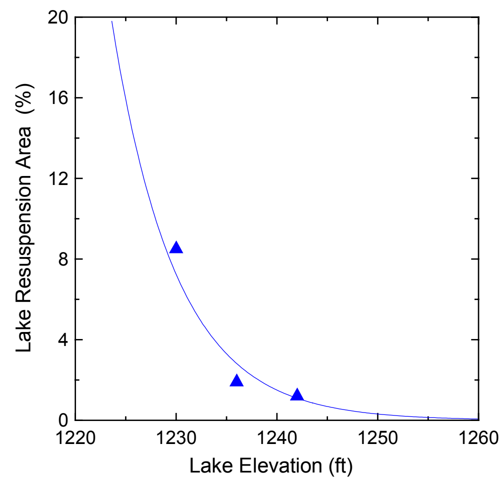

Fig. 8. Area of lake bottom prone to resuspension (type II and III sediments only) as a function of lake elevation.

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 that may be actively resuspended (i.e., occur within the wind-mixed region) (Fig. 8). The percent area of bottom sediments potentially resuspended will vary strongly depending upon lake elevation, with relatively little resuspension at high elevations and extensive resuspension potential at low surface elevations (Fig. 8).

The data in Fig. 8 were used in conjunction with sediment trap and other nutrient budget data (Anderson, 2001) to estimate the elevation-dependent resuspension velocity term (r) in eq 4. Specifically, resuspension of the fine organic (type II and III) sediments was assumed to be very low at high elevations and increase exponentially with decreasing elevation following Fig. 8.

Importantly, then, the net sedimentation rate, v, can be allowed to vary depending upon water quality conditions at the lake.

The change in volume of the lake, dV/dt, can be written as:

$$\frac{dV}{dt} = Q_{in} + PA_s - Q_{out} - EA_s \qquad (7)$$

where P is the precipitation rate (m/yr), A_s is the surface area of the lake (m^2), and E is the evaporation rate (m/yr). Available flow and precipitation data were used, although inflows were estimated for some parts of the simulation period (*e.g.*, 1998-1999). Evaporation rates were assumed to range from 2 ft/yr during the winter-spring months (December-May) to 6.6 ft/yr during the summer months. The above system of ordinary differential equations (eqs 2, 3 and 7) was solved using a forward difference scheme and a timestep of 0.05 yr. Hydrologic balance calculations yielded reasonable agreement between measured and observed lake volumes (Fig. 9).

Solution for TP concentrations in the above coupled ordinary differential equations requires an estimate of the net settling velocity (eq 4). The net settling velocity, v, was initially estimated by simply fitting an exponential to the 1993-1996 data, which yielded a 1st-order constant m of 0.38 yr^{-1}. Since $m=v/H$, and the average depth of the lake over this time period was about 7.5 m, v was first set to 2.85 m/yr. Somewhat better agreement was found when v was increased to 3.8 m/yr (or 0.01 m/d) to offset for evaporative concentration effects. This empricially-derived estimate of *v* (3.8 m/yr) from available water quality data (Fig. 3) is within the range of net settling velocities generally found for lakes (3-30 m/yr) (Thomann and Mueller, 1984). A net settling velocity on the low side of the above range in *v* values is consistent with the shallow and well-mixed condition of Lake Elsinore.

Since C_{sed} was assumed to be constant, *k* allowed to vary with water column TP concentrations (Fig. 6) following the empirical relationship 1.3×10^{-4} x C (mg/m^3) (m/yr), and *r* was taken to be very small (0.002 m/yr), eq 4 was rearranged and solved for v_s, the TP settling rate (m/yr). Doing this, one calculates a settling rate of 37 m/yr or ~0.1m/d. This settling rate is thought to represent largely algal TP. Encouragingly enough, this settling velocity comes in right at typical algal settling rates of 0.1-0.3 m/d measured in *in situ* experiments and used in other modeling studies (Thomann and Mueller, 1984), including the work conducted on Lake Elsinore by Montgomery-Watson (Montgomery-Watson,1997).

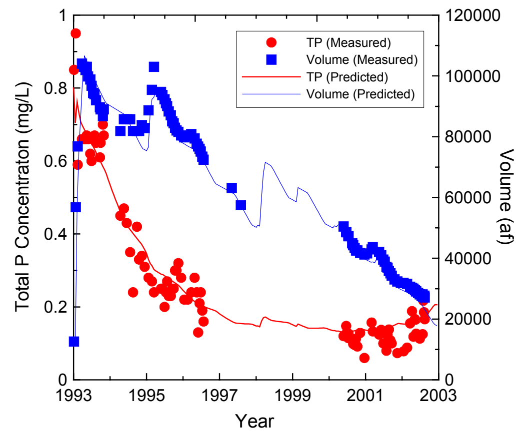

Fig. 9.

Using these independently-derived model parameters, one notes that the model actually does quite a reasonable job of reproducing measured TP concentrations over this time period (Fig. 9). The model correctly predicts a dramatic decline it TP concentrations over the period 1993-1996, from levels >0.6 mg/L to approximately 0.2 mg/L. The model then predicts TP levels to decline more slowly over the next couple of years, reaching a minimum concentration of 0.138 mg/L in May 2000, followed by an increase in TP levels to 0.19 mg/L by the end of 2002 (Fig. 9). It should again be noted that some assumptions about precipitation, runoff volumes and runoff TP concentrations were made for 1998-1999, so the predictions for this time period are tentative. Moreover, no lake elevation, volume or water quality data for the period 1998-1999 are readily available, so it is not possible to compare predicted and measured values for this period.

With some evidence supporting the applicability of the model for predicting water quality in Lake Elsinore, it is instructive to forecast predicted water quality following implementation of different restoration activities. Under controlled conditions, *e.g.*, regular addition of recycled water sufficient to maintain a lake elevation near 1242 ft, steady-state approximations are appropriate. Under steady-state conditions (*i.e.*, dV/dt=0 and dC/dt=0) and assuming Q_{out} is 0, eq 3, following substitution of eq 4 for *v*, reduces to:

$$C_{ss} = \frac{\left(\frac{Q_{in}C_{in}H}{V} + (k+r)C_{sed}\right)}{v_s} \qquad (8)$$

When C_{in} is 0, eq 6 simply reduces to:

$$C_{ss} = \frac{(k + r)C_{sed}}{v_s} \qquad (9)$$

Using the nutrient data developed for 2000-2001 (Anderson, 2001), one estimates an internal loading rate constant, *k* of 0.0156 m/yr, a resuspension velocity of 0.0021 m/yr, a volumetric sediment TP concentration of 247,000 mg/m^3 and a settling rate, v_s, of 37.4 m/yr. Substituting these values into eq 9, one estimates a steady-state TP concentration of 0.117 mg/L. This value is in excellent agreement with the annual average TP concentration of 0.119 mg/L reported by the RWQCB for the 2000-2001 period.

The water quality associated with this TP concentration in the lake was predicted using empirical relationships. The relationship of Dillon and Rigler (1974) was used to predict lake chlorophyll levels, where:

$$\log chl\ (\mu g/L) = 1.449 \log TP\ (ug/L) - 1.136 \qquad (10)$$

The Lake Elsinore-derived relationship between chlorophyll and Secchi depth (m) (Anderson, 2002), here rearranged to solve for Secchi depth, was used to estimate lake transparencies:

$$Secchi\ Depth\ (m) = 67.16 / (Chlorophyll\ (\mu g/L) + 55.98) \qquad (11)$$

The predicted chlorophyll level for the lake at a stable 1242 ft elevation (without external loads) is 73 µg/L, which is expected to produce a transparency of 0.52 m. These values are in reasonable agreement with previously reported measured values for chlorophyll and Secchi depth of 52 µg/L and 0.62 m, respectively.

The model was then used to predict steady-state TP, chlorophyll and transparency values for the lake subject to recycled water addition at different TP inlet concentrations (Table 1). A preliminary sensitivity analysis showed that the internal loading (*k*) and resuspension (*r*) terms drive the predicted TP levels in the lake; thus two different model parameterizations were used to estimate the likely range (*i.e.*, uncertainty) in predicted steady-state water quality in the lake.

Table 1. Predicted lake water quality resulting from addition of 15,000 af/yr at different influent P concentrations.

Influent P Conc (mg/L)	Lake TP Conc (mg/L)	Chlorophyll (µg/L)	Secchi Depth (m)
0	0.100 - 0.123	58 - 78	0.50 - 0.59
0.05	0.113 - 0.131	69 - 85	0.48 - 0.54
0.1	0.127 – 0.140	82 - 94	0.45 - 0.49
0.5	0.208 - 0.236	167 - 202	0.26 - 0.30
1.0	0.293 - 0.374	274 - 391	0.15 - 0.20

[a]assuming 15,000af/yr as some mix of recycled water, runoff, groundwater and other sources

Adding the equivalent of 15,000 af of water with no P, the predicted TP and chlorophyll concentrations in the lake were in good agreement with those values predicted using eq 9. The difference appears to be due to numerical dispersion and rounding errors in the model simulation. Notwithstanding such differences, it can be seen that low concentrations of P in the recycled water are predicted to have modest impacts on lake water quality; for example, 15,000 af of water at a P concentration of 0.05 mg/L is expected to only increase the TP by 0.008 – 0.013 mg/L and raise the chlorophyll concentrations by 7 - 11 µg/L. Such an increase in chlorophyll is expected to lower the Secchi depth by 0.02 - 0.05 m, to about 0.5 m. Increasing the influent P concentration to 0.1 mg/L is expected to increase average TP levels in the lake by up to 27%, while the chlorophyll level is expected to increase to 82 - 94 µg/L. Higher concentrations of P in the influent are expected to have more substantial effects on the water quality (Table 1). It should be noted that up to 3 simulation years were required before a steady-state condition in the lake was approached.

While the above projections were made assuming internal loading and resuspension rates were unchanged from the natural conditions in the lake, it is instructive to evaluate recycled water inputs to the lake with simultaneous internal load reductions. For these calculations, a reduction of 30% in the internal loading rate was assumed (Table 2).

Table 2. Predicted lake water quality resulting from addition of 15,000 af/yr at different influent P concentrations, with 30 % reduction in internal loading rate.

Influent P Conc (mg/L)	Lake TP Conc (mg/L)	Chlorophyll (µg/L)	Secchi Depth (m)
0	0.036 – 0.076	12.8 – 38.9	0.71 - 0.98
0.05	0.040 – 0.079	15.4 – 41.1	0.69 - 0.94
0.1	0.045 – 0.082	18.2 – 43.5	0.68 - 0.90
0.5	0.084 – 0.108	45.2 – 65.9	0.56 - 0.67
1.0	0.133 – 0.152	87.1 – 105.6	0.42 - 0.47

A 30% reduction in internal loading rate is predicted to yield a 38 - 61% reduction in lake TP concentration, with correspondingly low chlorophyll levels and high transparencies (Table 2) relative to the natural condition reflected in Table 1. This (unintuitive) lake response to internal load reductions comes from the coupling of the internal loading rate and the water column concentration (Fig. 6). That is, a reduction in the internal loading rate by some amount (e.g., 30%), results in a lowering of the water column concentration; this lower water column concentration, in turn, supports a still lower subsequent internal loading rate. Thus, up to a 2x net reduction in steady-state TP concentrations in the lake may be achieved for a given internal loading rate reduction.

A consequence of this is that internal load reductions, *e.g.*, through aeration or other control strategies, appear to allow relatively high levels of P in recycled water to be added to the lake (Table 2). Ongoing refinements to the model should improve its predictive power, especially at low influent P concentrations where relatively large uncertainties exist.

Discussion

Conclusions

References

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